RESTRICTED - COMMERCIAL
AEAT-3773: Issue 3

Options to Reduce Methane Emissions (Final Report)

A report produced for DGXI

November 1998

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Contents

1. Introduction

This report is one of the final reports under a study completed by AEA Technology Environment for DGXI on the control and reduction of greenhouse gases and ozone precursors. Four gases were included in the study, the two direct greenhouse gases, methane and nitrous oxide, and the ozone precursors, nitrogen oxides (NO_x) and non-methane volatile organic compounds. In the initial phase of the study, inventories of these gases for all Member States were reviewed and updated. In the second phase of the study, measures to control and reduce emissions of these gases were identified, their technical feasibility examined, and wherever sufficient cost and performance data was available, the cost-effectiveness of the measures (in terms of ECU (1995) per tonne of pollutant) is also estimated.

This report analyses methane emissions and strategies to control them. Section 2 describes the global sources of methane emissions, the significance of methane as a greenhouse gas, and the level of methane emissions in the EU. These emissions are discussed in context against emissions of the two other direct greenhouse gases (carbon dioxide and nitrous oxide) and the important emissions sources within the EU are identified.

Sections 3-7 of the report consider options for the reduction of emissions from agriculture, waste, coal mining and the oil and gas industry. In each case the sources of emissions, the mitigation options available, and their costs are discussed. The costs have been calculated using an annualised cost methodology. All costs have been calculated based on an 8% discount rate and are expressed in 1995 ECUs to ensure consistency with previous work on the cost of carbon dioxide reduction options. Full details of the costing methodology, exchange rates, deflators and other factors used are given in Appendix 1. Finally each section includes an analysis of the applicability of the considered reduction options under future scenarios and projections of future emissions.

Projections of more minor emissions sources are considered in Section 8 together with a discussion of the potential impact of CO_2 reduction measures on methane emissions arising from fuel combustion. Section 9 of the report examines possible variations in of measures (in all sectors) between Member States.

A summary of the report is contained in Section 10, where projections of emissions and estimates of achievable reductions from all sectors are combined to give an EU wide projection of total methane emissions if a mitigation strategy were implemented. This section also contains a cost--effectiveness curve for all measures.

2. Methane Emissions in the EU

2.1 METHANE AS A GREENHOUSE GAS

Methane (CH_4) is an important greenhouse gas whose concentration in the atmosphere has more than doubled since pre-industrial times, rising from a pre-industrial concentration of about 700 ppbv to a concentration in 1994 of 1720 ppbv. Over the last 20 years there has been a decline in the growth rate of this methane concentration. In the late 1970s, the concentration was growing at a rate of about 20 ppbv/yr, but this fell to a growth rate of 9 to 13 ppbv/yr in the 1980s. Around the middle of 1992, methane concentrations briefly stabilised, but since 1993, the global growth rate has returned to about 8 ppbv/yr.

Methane is a more potent greenhouse gas than carbon dioxide, with a global warming potential (over a 100 year time horizon) 21 times greater than carbon dioxide. However, due to its shorter atmospheric lifetime (of 12 years) it is estimated that global emissions would only need to be reduced by about 8% from current levels to stabilise methane concentrations at today's levels (IPCC, 1996). This is a much smaller percentage reduction than those required to stabilise atmospheric concentrations of the other major greenhouse gases, carbon dioxide (CO_2) and nitrous oxide (N_2O).

An estimate of the global sources and sinks of CH_4 is given in Table 2.1. The sources and sinks of CH_4 are not well quantified and there are many uncertainties. Anthropogenic emissions of methane in the EU are estimated at 22 Mt in 1994, about 6% of global anthropogenic emissions.

Emissions of the three main greenhouse gases in the EU in 1994 are shown in Table 2.2. After allowing for the global warming potential of the gases (100 year), it is clear that methane is a significant contributor to greenhouse gas emissions in the EU, with emissions in 1994 equivalent to 14% of CO_2 emissions.

Table 2.2 Anthropogenic Emissions of CO_2, CH_4 and N_2O in the EU in 1994

Direct GHG	Emissions (kt) in 1994	GWP (100 years)	Global Warming Equivalence - emissions GWP (equivalent kt of CO_2)
CO_2	3 215 558	1	3 215 558
CH_4	21 930	21	460 530
N_2O	1049	310	325 190

Source: Member States and EU Second Communications to the FCCC.

Table 2.1 Estimated Sources and Sinks of Methane (Mt CH_4 per year)

	Range	Best Estimate
Observed atmospheric increase	**35 - 40**	**37**
Sinks		
Reaction with OH in the troposphere	360 - 530	445
Reactions with OH, Cl and O in the stratosphere	32 - 48	40
Removal by soils	15 - 45	30
Total sinks	**430 - 600**	**515**
Implied total sources (atmospheric increase + total sinks)	**465 - 640**	**552**
Identified Sources		
Natural		
Natural wetlands (swamps, marshes, tundra etc.)	55 - 150	115
Termites	10 - 50	20
Oceans	5 - 50	10
Other	10 - 40	15
Total identified natural sources	***110 - 210***	***160***
Anthropogenic		
Fossil fuel related:		
Natural gas	25 - 50	40
Coal mines	15 - 45	30
Petroleum industry	5 - 30	15
Coal combustion	1 - 30	?
Total fossil fuel related	*70 - 120*	*100*
Biospheric carbon:		
Enteric Fermentation	65 - 100	85
Rice fields	20 - 100	60
Biomass burning	20 - 80	40
Landfills	20 - 70	40
Animal wastes	20 - 30	25
Domestic sewage	15 - 80	25
Total biospheric	*200 - 350*	*275*
Total identified anthropogenic sources	***300 - 450***	***375***
TOTAL IDENTIFIED SOURCES	**410 - 660**	**535**

Source: IPCC, 1995.

2.2 CURRENT METHANE EMISSIONS IN THE EU

Only anthropogenic emissions of methane are considered in this report. Estimated anthropogenic emissions of methane by sector in the EU in 1990 and 1994 are shown by country in Table 2.3 and for 1994 for the EU15, in Table 2.4 and Figure 2.1.

Table 2.3a Estimated Emissions of CH_4 in EU Member States in 1990 (kt)

	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
Enteric fermentation	**146**	**198***	**167**	**90**	**1430**	**1430**	**142**	**551**	**643**	**16**	**402**	**124**	**346**	**188**	**1005**	**6878**
Manure management	**27**	**176***	**162**	**11**	**168**	**614**	**23**	**52**	**192**	**2**	**103**	**68**	**465**	**12**	**125**	**2199**
Solid waste disposal	**193**	**173**	**71**	**126**	**758**	**1777**	**102**	**136**	**302**	**4**	**562**	**493**	**472**	**85**	**1890**	**7144**
Fugitive emissions - coal	**0**	**15**	**3**	**0**	**206**	**1230**	**43**	**0**	**15**	**0**	**0**	**3**	**613**	**0**	**818**	**2946**
Fugitive emissions - oil & gas	**4**	**39**	**9**	**0**	**126**	**333**	**0**	**10**	**304**	**2**	**179**	**1**	**74**	**0**	**480**	**1560**
Other sources:	**217**	**36**	**10**	**19**	**328**	**298**	**132**	**62**	**947**	**1**	**44**	**127**	**213**	**39**	**146**	**2619**
- fuel combustion	21	16	10	15	163	205	15	5	97	0	35	22	76	39	98	816
- industrial processes	0	4	0	4	3	0	0	0	4	0	3	0	2	0	0	20
- other agriculture	36	15	0	0	28	0	107	37	67	0	0	19	115	0	12	436
- land use change activities	127	0	0	0	93	0	0	20	179	0	0	0	0	0	0	418
- wastewater	14	0	0	NE	16	93	6	0	511	0	6	85	0	0	33	764
- other	20	2	0	0	27	0	4	0	89	1	0	0	20	0	2	165
Total Anthropogenic Emissions	**587**	**635**	**422**	**246**	**3017**	**5682**	**443**	**811**	**2403**	**25**	**1289**	**816**	**2181**	**324**	**4464**	**23346**

Table 2.3b Estimated Emissions of CH_4 in EU Member States in 1994 (kt)

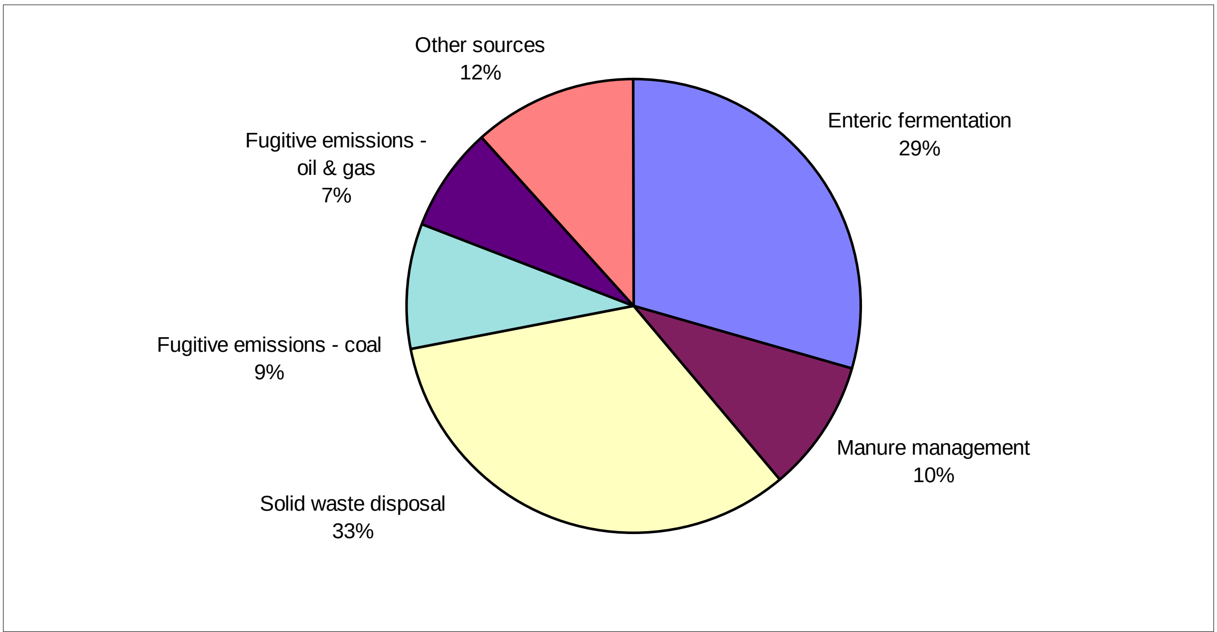
	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
Enteric fermentation	**146**	**199***	**155**	**83**	**1354**	**1162**	**143**	**548**	**607**	**21**	**382**	**119**	**352**	**184**	**991**	**6443**
Manure management	**27**	**176***	**171**	**10**	**170**	**498**	**26**	**56**	**182**	**1**	**101**	**63**	**481**	**19**	**125**	**2105**
Solid waste disposal	**187**	**184**	**72**	**122**	**667**	**1780**	**105**	**136**	**427**	**2**	**505**	**528**	**658**	**61**	**1790**	**7223**
Fugitive emissions - coal	**0**	**0**	**6**	**0**	**213**	**850**	**48**	**0**	**7**	**0**	**0**	**2**	**525**	**0**	**327**	**1978**
Fugitive emissions - oil & gas	**5**	**45**	**11**	**0**	**121**	**320**	**0**	**11**	**349**	**2**	**168**	**1**	**93**	**0**	**484**	**1610**
Other sources:	**216**	**34**	**13**	**30**	**336**	**239**	**135**	**57**	**966**	**1**	**44**	**121**	**205**	**40**	**131**	**2569**
- fuel combustion	19	14	10	16	161	119	15	4	106	1	34	22	74	40	93	729
- industrial processes	0	3	1	4	2	0	0	0	5	0	5	0	2	0	0	23
- other agriculture	36	14	0	0	32	0	109	30	73	0	0	13	101	0	0	408
- land use change activities	127	0	0	0	93	0	0	23	190	0	0	0	0	0	0	433
- wastewater	14	0	2	10	17	120	6	0	511	0	5	85	0	0	36	807
- other	20	2	0	0	31	0	4	0	82	0	0	0	28	0	2	169
Total Anthropogenic Emissions	**581**	**637**	**428**	**245**	**2860**	**4849**	**457**	**807**	**2538**	**27**	**1199**	**834**	**2314**	**303**	**3848**	**21928**

** Values derived by splitting value for agriculture given in Belgian Second National Communication according to data reported in CORINAIR*

Source: Member States and EU Second Communications under the FCCC.

The largest source is the agricultural sector, which was responsible for 41% of anthropogenic emissions. These emissions arise principally from enteric fermentation in the digestive tract of ruminant livestock (cattle and sheep), but also from the anaerobic decomposition of livestock manure. The other major source is landfills, where the anaerobic decomposition of organic waste in the landfill leads to the release of landfill gas, which is a mix of CO_2 and CH_4. Coal mining and gas production and distribution are smaller, but still significant, contributions.

Figure 2.1 Methane Emissions by Source for EU in 1994 (Total 21.9 Mt)



Source: Member States and EU Second Communications to the FCCC.

Table 2.4 Methane Emissions by Source for EU in 1994

Sector	Emissions	% of total	Of which:	
Agriculture	9.0 Mt	40.8%	Enteric fermentation	29.4%
			Livestock manure	9.6%
			Other agriculture	1.9%
Waste	8.2 Mt	37.4%	Solid waste disposal	32.9%
			Wastewater treatment	3.7%
			Waste incineration	0.8%
Energy sector	4.3 Mt	19.6%	Coal mining, transport and storage	9.0%
			Gas production and distribution	7.3%
			Combustion (including transport)	3.3%
Other	0.4 Mt	2.1%	Land use changes	2.0%
			Other sources	0.1%
Total	**21.9 Mt**	**100%**		

Source: Member States and EU Second Communications under the FCCC.

There is significant uncertainty attached to some of the emissions estimates, particularly those arising from landfill sites. As an example, the uncertainties estimated for each source sector in the UK emissions inventory (Salway, 1997) are shown in Table 2.5. A Monte Carlo analysis suggested that overall, the maximum uncertainty in the total emissions estimate is around 19%.

Table 2.5 Estimated Uncertainties in the UK Methane Inventory

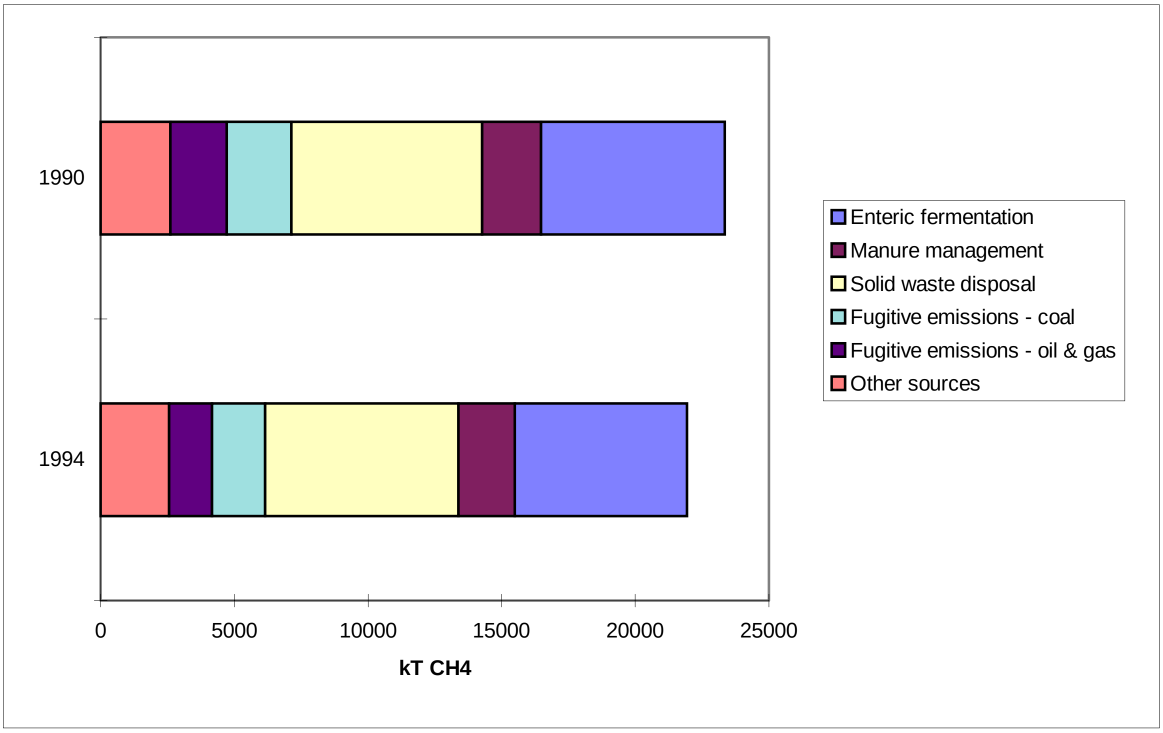
Source	Source Uncertainty %
Fuel Combustion	**50**
Landfill	**~39**[1]
Livestock: enteric	**20**
Livestock: wastes	**30**
Coal Mining	**13**
Gas Leakage	**17-75**[2]
Offshore	**28**
Sewage Sludge	**50**
Total	**19**

1 Skewed distribution
2 Various uncertainties for different types of main service
Source: Salway, 1997

2.3 TRENDS IN METHANE EMISSIONS 1990-1994

Methane emissions in 1990 were 23.3 Mt (Table 2.3) and had thus fallen by 6% by 1994 (Figure 2.2).

Figure 2.2 Trends in EU Methane Emissions 1990-1994



Source: Member States and EU Second Communications to the FCCC.

Trends in individual sectors were as follows:

- The greatest reduction in emissions was from the ***extraction and distribution of fossil fuels*** (1 Mt), with emissions from both coal mining and the oil and gas industry falling significantly (by 18% and 23% respectively). Reduction in emissions from coal mining were primarily a result of a decline in deep coal mining in the EU over this period, although improved methane capture is also thought to have contributed to this trend. Emissions from the oil and gas sector have fallen due to the replacement of gas distribution pipework which has reduced fugitive emissions, the major source of emissions in this sector.

- Emissions from the ***agricultural*** sector fell by about 6% (0.5 Mt), with emissions from enteric fermentation falling by 6% and emissions from manure by 4%. This reduction can be attributed to changes in livestock numbers and improvements in manure management methods over this period. Between 1990 and 1993, the number of dairy cows fell by 5.8% and all cattle by 1.6%, although pig numbers increased by about 9% (Eurostat 1995). Thus enteric fermentation emissions, which come mainly from ruminants and are highest for dairy cows, fell more sharply than emissions from manure.

- Emissions from waste disposal (landfill) increased by about 2% over the period 1990-1994 in spite of measures by most EU countries to limit the amount of waste sent to landfill and to capture methane from new and existing landfill sites. This increase can be explained by the long term nature of emissions from landfills, i.e. waste deposited several years ago will still be releasing methane, so that the impact of reductions in the amount of waste disposed to landfill take several years to be fully realised. Similarly, measures to improve the capture and combustion of methane will be generally targeted at new sites and will not have an impact on the significant proportion of emissions arising from older sites.

2.4 COMPARISON OF EMISSION INVENTORY DATA

Estimates of methane emissions in the EU are available from two sources:

CORINAIR - the Europe wide emissions inventory compiled by the European Environment Agency from national inventories supplied by countries. Data is available for 1990 and 1994, and is presently being compiled for 1995 and 1996.

FCCC submissions - the IPCC has issued guidelines for the compilation and reporting of inventory data by counties to the Framework Convention on Climate Change. Data for the EU Member States compiled for this purpose is available for 1990 and 1994 and in some cases also for intervening years and for 1995.

In general, emissions data compiled for the purposes of reporting to the FCCC has been used as the basis of analysis and reporting in this report, since it is the basis on which Member States and the EU's compliance with the Kyoto targets under the FCCC will be agreed. In some cases, however, a more detailed breakdown of sources is available from the CORINAIR inventory and this has been used to aid analysis; these cases are clearly indicated in the text.

Figure 2.3 shows the breakdown of EU methane emissions by source for 1990 and 1994 given by CORINAIR and by the submissions to the FCCC. A more detailed breakdown of the CORINAIR data for 1990 and 1994 is given in Appendix 2.

The two inventories are broadly similar in that they give similar values for total EU emissions, particularly in 1990 and they agree on the relative contributions of the major emission sources. However, they also show the following differences.

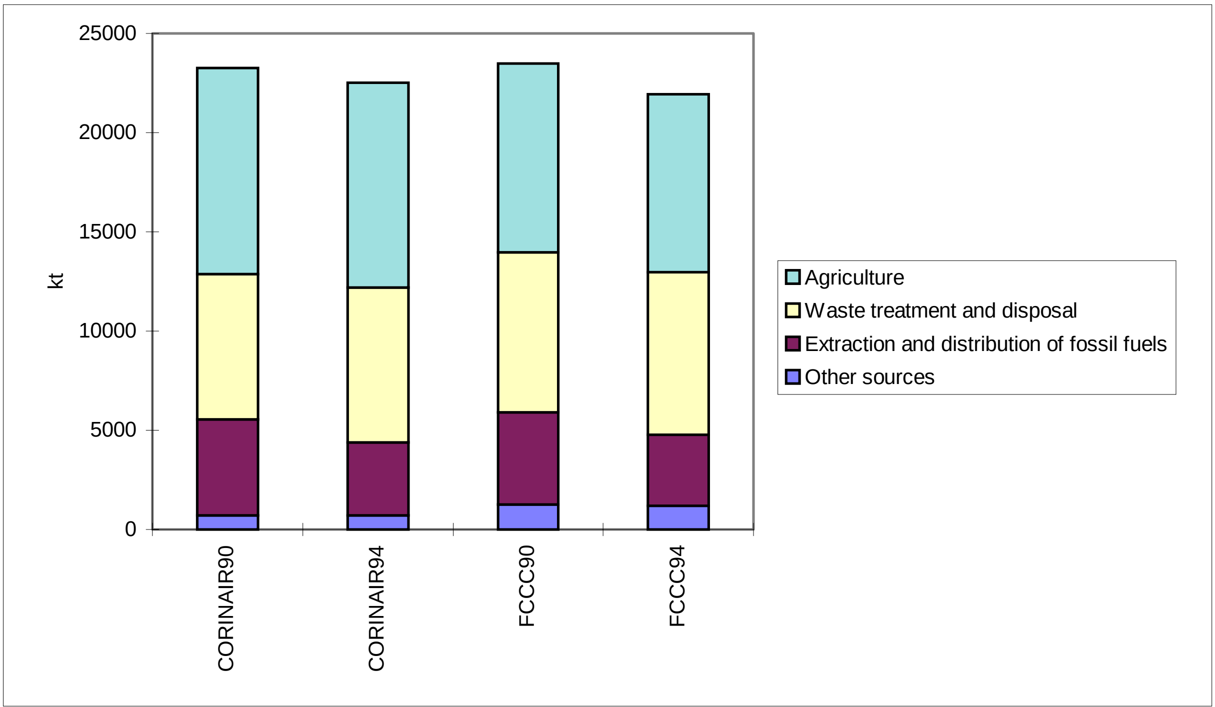
- The estimate of EU emissions in 1990 derived from countries' FCCC submissions is very slightly higher (23,346 kt) than the CORINAIR estimate (23,261 kt). However 1994 emissions in CORINAIR are almost 3% higher than the FCCC submissions and the reduction between 1990 and 1994 is therefore smaller according to the CORINAIR inventory (3% compared to 6%).

- The FCCC submissions include a category for land use changes which is not included in the CORINAIR database.

- Emissions from wastewater treatment are estimated to be much higher by the national and EU submissions to the FCCC (about 3% of total) than CORINAIR (about 1% of total).

The EEA, through its Air Emissions Topic Centre and the UNECE Emission Inventory Task Force are actively working on improving the reporting of emissions in Europe and resolving the differences between the two datasets.

Figure 2.3 Comparison of EU Methane Emissions by Source for 1990 and 1994



Source: CORINAIR90 and CORINAIR94 and Member States and EU Second Communications to the FCCC.

3. Options to Reduce Emissions from Enteric Fermentation

3.1 METHANE EMISSIONS FROM ENTERIC FERMENTATION

3.1.1 Mechanism of enteric fermentation

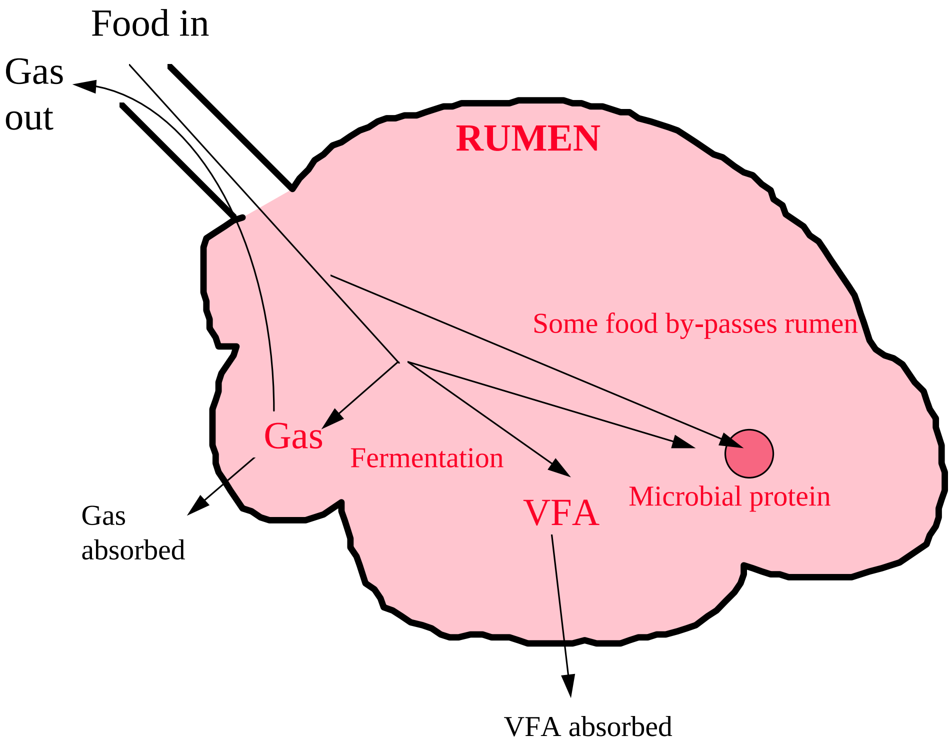
Enteric fermentation is the anaerobic fermentation of polysaccharides and other feed components in the gut of animals. Methane is produced as a waste product of this fermentation process.

In humans and other non-ruminant mammals, digestion is achieved by the action of enzymes in the gastric juices of the stomach. Food has a relatively short residence time in the gut and so there is little fermentation and associated methane production. In ruminant animals however, such as cows and sheep, plant polymers in the feed cannot be digested by host enzymes alone. Instead, ruminant animals have an expanded gut (retrulo-rumen, generally termed the rumen) in which food is broken down by fermentation prior to gastric digestion in the abomasum. In an adult ruminant, the rumen represents about 85% of the total stomach capacity and contains digesta equal to about 10% to 20% of the animal's weight (Campling et al, 1961). Coarse feedstuffs are retained in the rumen for a considerable period of time in the presence of a large and diverse microbial population, allowing extensive fermentation.

The mechanism of enteric fermentation in the rumen is shown in Figure 3.1. Food enters the rumen where it is fermented to volatile fatty acids (VFA), carbon dioxide and methane. 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 of the host; they may also be directly utilised by the host as building blocks for synthesis of cell material. Fermentation is also coupled to microbial growth and the microbial cell protein synthesised forms the major source of protein for the animal. The gaseous waste products of the fermentation, carbon monoxide and methane, are mainly removed from the rumen by eructation (Dougherty et al, 1965). A small proportion of methane is absorbed in the blood and is eliminated through the lungs.

The efficiency of fermentation in the rumen, and hence methane emissions, depends on the diversity, size and activity of the microbial population in the rumen, which are largely determined by diet. An extensive grazing ruminant will feed on forage, the term given to fibrous material, usually involving the whole plant. A more intensively reared ruminant may feed on forage for part of the time and concentrates for the remainder. (Concentrates are feedstuffs that are mainly derived from the seeds of plants or by-products of the seeds after processing.) In the wild and extensive grazing situation, rumen efficiency will vary with seasonal and climatic differences as they affect the availability, composition and variety of vegetation available. In domesticated ruminants, where conditions are less variable, changes in diet composition, physical form and amount offered are largely responsible for changes in the microbial population (Thorley et al, 1968; Mackie et al, 1978).

Figure 3.1 Flow diagram of the rumen



Bacteria are the principal micro-organisms that ferment carbohydrates in the rumen (Hungate, 1966). The type of bacteria required depends on the animal's diet. For an animal fed on forage and concentrates, both cellulolytic and amylolytic bacteria must be present to maximise rumen efficiency. 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 (Wolin, 1974). 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).

3.1.2 Methane emissions from livestock

Methane emissions from livestock 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, as discussed above, on rumen efficiency and the quality (digestibility and energy value) of the feed. Methane emissions per head for typical livestock in Europe (IPCC, 1996) are shown in Table 3.1, and are combined with estimates of livestock numbers to give an indication of enteric emissions by animal type in the EU. Non-dairy cattle produce about half as much methane per head as dairy cows, but are responsible for just over half of enteric emissions.

Table 3.1 Methane emissions from different types of livestock in the EU

Animal type	Emission factor (kg CH_4/head/yr)	Animals in EU in 1990 (millions)	CH_4 emissions (kt)
Dairy cows	100	25.0	2,498
Non-dairy cattle	48	87.8	4,215
Sheep	8	98.1	785
Pigs	1.5	113.9	171
Total			**7,669**

Source: Derived from IPCC, 1997 and Eurostat, 1995.

3.2 MITIGATION OPTIONS

The most direct approach to reduce enteric emissions is to increase rumen efficiency and reduce the amount of methane produced for a given amount of feed intake; a number of options for achieving this are discussed below.

A more indirect approach is to improve animal productivity, that is, to increase the amount of product (milk or meat) produced per unit of feed intake. However this will only have the desired result of reducing total enteric emissions if the amount of product is kept constant, i.e. as productivity increases, animal numbers are decreased so that the total amount of product remains constant. The current effects of the Common Agricultural Policy (CAP) indicate that this trend is already evident. In particular milk production is limited by quotas so that as productivity increases, livestock numbers are decreasing. In the case of beef however, subsidies are paid on a per head basis, so a decline in livestock numbers might not be expected to follow increases in productivity unless policy is reformed.

Finally, emissions could be reduced by reducing the amounts of milk and meat produced in the EU, an option which is inextricably linked with general decisions on EU agricultural policy. Reducing production within the EU is likely to be compensated for by increases in production elsewhere. If this occurs in countries where animal productivity is not as high then there may actually be a net increase in global emissions. For these reasons, this option is not discussed further here.

3.2.1 Options to increase rumen efficiency

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: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.

Feeding ruminants on diets containing high levels of readily fermented non-structural carbohydrate has been shown to minimise methane production by reducing the protozoal population and lowering rumen pH. However, this can give rise to an overall depressed ruminal fermentation, which may lower the conversion of feed energy into animal product and may be detrimental to the animal's health. Using diets with extreme nutrient compositions is therefore not considered likely to be a successful or sustainable method to control methane emissions from ruminants.

A number of possible options have been identified for increasing rumen efficiency without threatening animal health:

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

These options are summarised in Table 3.2 and briefly described below. Further details are given in Annex A.

3.2.1.1 Hexose partitioning

As already discussed, during rumen fermentation, feedstuffs are converted into short-chain volatile fatty acids (VFAs), ammonia, methane, carbon dioxide, cell material and heat. Animal performance is dependent on the balance of these products and this balance is ultimately controlled by the types and activities of micro-organisms in the rumen. The VFAs are used by the animal as an energy source while the microbes serve as an important source of amino acids for protein synthesis. Ammonia, methane and heat by contrast represent a loss of either nitrogen or energy unavailable to the animal.

By varying diet, it may be possible to manipulate the amount of the feed carbohydrate going directly into microbial growth as opposed to fermentation (hexose partitioning). Theoretical studies have shown that increasing the quantity of microbial cells leaving the rumen per unit of carbohydrate consumed may have a large effect on the overall methane production (up to a 35% reduction; Beever, 1993). Further experimental research is required to investigate, *in vitro*, carbohydrate sources that provide improved hexose partitioning and to use this information to design diets with enhanced hexose partitioning for testing *in vivo* to determine the impact on methane emissions.

Theoretically this technology should also enhance protein utilisation and hence reduce ammonia emissions. This option, if proved in practise to reduce methane emissions, could be implemented in all Member States though it would be more readily implemented for livestock receiving supplementary concentrates, and less applicable to grazing/free-ranging livestock. The cost of implementing the option is likely to be minimal as the overall effect would be increased productivity which would offset any additional feed costs associated with the option. However, no reliable cost or performance data are available at present.

Table 3.2 Options to increase rumen efficiency

Option	Status	Available by	Reduction offered (on evidence to date)	Applicability		Cost	Comments	Actions required
				Types of animals	Countries			
Hexose partitioning (via diet changes)	Some theoretical evidence - needs research (some ongoing in UK + Germany?)	2010 onwards	Theo. max. 35-40%. In field 15-20%	Livestock receiving concentrates i.e. mostly cattle	Wherever supplements are used	Cost neutral due to gains in productivity	Possible added benefit of reducing ammonia emissions	More R&D
Propionate precursors (via additives or via adapted plants)	Evidence from *in vitro* and some *in vivo* research. Needs confirmation *in vivo* + must assess effectiveness with different diets.	2005 onwards (additives) 2015 onwards (plants)	Up to 25% in field.	Livestock receiving concentrates for additive route. Wider applicability via plants.	Wherever supplements are used for additives. Wider applicability via plants.	Supplements ~80g/day for feedlot cattle @ 2200 ECU/tonne, excluding productivity gains.	Possible other benefits: improved feed degradation and reduced acidosis.	More R&D Better dialogue needed between animal nutritionists and plant breeders.
Direct fed microbials – Acetogens	Subject of research in Belgium, France, Spain and UK. Several 'hurdles' to clear.	2010 onwards	Potentially high level of reduction but many uncertainties	Livestock receiving concentrates i.e. mostly cattle	Wherever supplements are used	Not known at this stage	Possible added benefit of reducing ammonia emissions. No "side-effects" expected.	
Direct fed microbials – Methane oxidisers	Some ongoing research and *in vitro* evidence. Needs further R&D for practical application.	2010 onwards	~8% on limited evidence so far	Livestock receiving concentrates i.e. mostly cattle	Wherever supplements are used	Not known at this stage	Less promising approach than acetogens.	
Genetic engineering	Very little R&D since 1980's. Unlikely to be acceptable (see comments)	2020 onwards (if at all)	-	-	-	-	Genetically modified organisms unlikely to be acceptable to EU or member states.	
Immunogenic approach	Patented by Australian group. Applications uncertain without further information.	?	up to 70%	All ruminants	All Member States	Not known	Very promising but uncertain.	Set up collaborative research.

3.2.1.2 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.

It is estimated that if successful, the option could reduce methane emissions by up to 25%, (ADAS, 1998), and that there could be other benefits to the livestock industry such as improved feed degradation which would be likely to reduce feed costs. Another possible benefit would be a reduced incidence of acidosis (a digestive disorder) in high producing dairy cows and intensively reared cattle, which could lead to considerable cost savings. As propionate precursors naturally occur in the rumen, they are likely to be more readily acceptable than antibiotic or chemical additives.

3.2.1.3 Direct fed microbials

Certain microbes in the rumen are known to promote reactions that minimise methane production and it may be possible to introduce such microbes directly as feed supplements. Such microbes include acetogens and methane oxidisers.

Acetogens

Acetogens are bacteria that produce acetic acid by the reduction of carbon dioxide with hydrogen, thus reducing the hydrogen available for reaction to produce methane (methanogenisis) (Demeyer and de Graeve, 1991). Although this reaction is theoretically possible in the rumen, populations of acetogens in the rumen of adult ruminants are low and a methane producing reaction tends to dominate. Research groups are currently investigating these reactions with the aim of devising practical solutions for the survival of acetogenic bacteria in the rumen and hence the displacement of methanogenic bacteria. This would not only decrease methane production, but would also increase the efficiency of ruminant production.

An alternative approach would be to screen a range of acetogenic bacteria for their activity in rumen fluid and to introduce the acetogens into the rumen as a feed supplement on a daily basis. If successful, this option has the potential to eliminate or reduce to a minimum methane emissions from ruminants. Emissions of ammonia may also be reduced as a result of more efficient carbohydrate fermentation which requires nitrogen. The option would again be applicable to all ruminants receiving supplements on a controlled and regular basis. The costs associated with isolating, growing and preparing this type of micro-organism are not clear, but some of these costs would inevitably be offset by improved rumen efficiency.

Methane oxidisers

Methane oxidisers could also be introduced as direct-fed microbial preparations. The oxidation reaction would compete with the production of methane, which is a strictly anaerobic process. Methane oxidisers from gut and non-gut sources could be screened for their activity in rumen fluid *in vitro* and then selected methane oxidisers could be introduced into the rumen on a daily basis in a manner analogous with current feed supplements.

If successful, this option has the potential to reduce methane production in the rumen by a minimum of 8% (ADAS, 1998). As for acetogens, the option would be available to all ruminants receiving supplements on a controlled and regular basis but the costs associated with isolating, growing and the preparation of the micro-organism are not clear.

3.2.1.4 Genetic Engineering

Recombinant deoxyribonucleic acid (DNA) technology could potentially be used to modify the fermentation characteristics of rumen micro-organisms (Armstrong and Gilbert, 1985). Examples of application include an enhanced cellulolytic activity in the rumen biomass for forage fed animals to increase their supply of VFAs and amino acids, and a reduction in methanogenesis accompanied by an alternative hydrogen sink through increasing propionate production.

This method however, is likely to be unacceptable to most EU Member States as there is considerable opposition to the increased release of genetically engineered organisms into the environment.

3.2.1.5 Immunogenic approach

A team of researchers at CSIRO in Australia have made an application for a world wide patent for a method of improving the productivity of a ruminant animal by administering to the animal an immunogenic preparation effective to invoke an immune response to at least one rumen protozoan. The removal of one species of protozoan from the rumen will invoke the improvements in productivity associated with defaunation. It is also believed that by modifying the activity of the rumen protozoan, there will be an indirect effect on the activity of methanogens, due to their commensal relationship with rumen protozoa.

Data from this work are not yet published but it is anticipated that methane production could be reduced by as much as 70%. The long term prospects of this approach are not yet available but areas to be considered are the longevity of the immunisation and whether other species of protozoa and methanogens will increase their populations to compensate for those species where immunisation has taken place.

If this option develops successfully, it could be applied to the whole ruminant population. The costs associated with the approach could be high initially due to the monopoly associated with patents. The increased protein utilisation associated with defaunation would mean reduced emissions of ammonia and increased animal productivity.

3.2.2 Options to increase animal productivity

As discussed earlier, success in reduce methane emissions from ruminants by increasing animal productivity depends on keeping overall production levels constant. That is, a reduction in total emissions of methane would only result if total output levels (e.g. total milk or beef produced) remained constant and the advantages gained from increased productivity were realised by reducing livestock numbers.

Possible options for increasing ruminant productivity are discussed in the following sections. These options are summarised in Table 3.3 and discussed in more detail in Annex A

3.2.2.1 Probiotics

Probiotics are microbial feed additives containing live cells and a growth medium. They are already widely available in the EU, and are used to improve animal productivity. From an analysis of published results from more than 1000 cows, Wallace and Newbold (1993) calculated that probiotics stimulated milk yield by 7.8%, and from 16 trials using growing cattle, they showed an average increase in liveweight gain of 7.5%. Further research is required to confirm whether there is any additional effect on methane production *per se*. Even without a direct effect on methane production, there would be a reduction in methane production per unit of production (e.g. per litre of milk).

3.2.2.2 Ionophores

Ionophores are chemical feed additives which increase productivity (weight gain per unit of feed intake) by adjusting several fermentation pathways. On average, an 8% increase in feed conversion efficiency has been observed (Chalupa, 1988). Reductions in methane production (of up to 25%) have been observed (Van Nevel and Demeyer, 1992), but the persistence of this reduction is unproven. Ionophores are fed to beef animals only in the EU. Their use in dairy cows is not permitted because a withdrawal period is required before human consumption.

The use of chemicals and antibiotics to increase animal productivity is increasingly becoming unpopular to the consumers of animal products. It is therefore considered that the use of ionophores to reduce methane production is not a viable option.

3.2.2.3 Bovine somatotropin

Bovine somatotropin (BST) is a genetically engineered metabolic modifier approved for use in some countries to enhance milk production from dairy cows. Again, this is not a popular consumer choice for enhancing animal productivity and its use now banned by all EU Member States. Again this is therefore not considered a viable option.

3.2.2.4 Forage type and supplementation

Supplementing forages whether of low or high quality, with energy and protein supplements, is a well documented method of increasing microbial growth efficiency and digestibility, and thus increasing milk and meat productivity. The direct effect on methanogenesis is variable and unclear.

Research has shown that increasing the level of non-structural carbohydrate in the diet (by 25%) would reduce methane production by as much as 20%, but this may result in detrimental health effects e.g. acidosis, fertility problems (Moss, 1994). Also with the implementation of quotas for milk production in the EU, many producers are optimising milk production from home-grown forages in order to reduce feed costs. Supplementing poor quality forages and chemically upgrading them may be a good option for increasing productivity and in turn reducing methane emissions per unit product.

Feeding of ruminants to optimise rumen and animal efficiency is a developing area and the efficient deployment of appropriate nutritional information to all livestock producers would benefit the environment in terms of both methane and nitrogen emissions.

Table 3.3 Options to increase animal productivity*

Option	Status	Available by	Reduction offered (on evidence to date)	Applicability		Cost	Comments	Actions required
				Types of animals	Countries			
Probiotics	Widely available but effect on productivity not established.	Now	7.5% per unit output	Livestock receiving supplements	Wherever supplements are used	Supplements ~50g/day for adult cattle @ 2100 ECU/tonne.	Less opposition than for other additives.	
Ionophores	Already in use	Now	~8% per unit output + more if CH_4 reduced too	Livestock receiving supplements (beef only)	Wherever supplements are used	?	Consumer acceptability a major issue.	
Bovine somatropin	Already in use	Now	~15%	Livestock receiving supplements (dairy only)	Wherever supplements are used	?	Currently not permitted in the EU. Consumer acceptability a major issue.	
Forage type and supplementation	Already in use	By 2000?	~5%					
High genetic merit dairy cows	Current trend	Now	~20-30%	All dairy cattle.	All EU members states	Cost effective	High genetic merit cows can have problems with fertility, lameness, mastitis and metabolic disorders.	

* Only valid if the overall production levels remain the same, i.e. livestock numbers are reduced.

3.2.2.5 High genetic merit dairy cows

Improving the genetic merit of dairy cows has escalated in the last decade with the import of Holstein genetic material from US and Canada for use on the EU native dairy breeds. As a result, average national yields have increased. For example, the UK dairy herd has increased its average yield by 8.8% from 1995 to 1997 (ADAS, 1998). One of the major improvements is the ability of the cow to partition nutrients into milk preferentially to maintenance and/or growth. This has undoubtedly resulted in increased efficiency.

The genetic merit of livestock within the EU is rapidly improving and this will undoubtedly bring with it increased efficiency, and potential reductions in methane emissions of 20 to 30%. However, the management of these high genetic merit cows will also become more complex and overall implementation of this approach may be stalled by animal welfare implications. High genetic merit cows can have increased problems with fertility, lameness, mastitis and metabolic disorders, and all these issues will have to be addressed if genetic progress is to be successfully continued.

3.3 COST OF OPTIONS

With respect to enteric fermentation, enough cost and performance data is available to calculate the cost-effectiveness of two options:

- propionate precursors;
- probiotics.

Increasing rumen efficiency by hexose partitioning or the immunogenic approach and increasing productivity by forage supplementation or introducing high merit cows are also promising mitigation options but there is insufficient cost and performance data to adequately assess their cost-effectiveness at present.

The methodology used to calculate the annualised cost (in 1995 ECU) of the measures, and details of exchange rates and conversions from different years are described in Appendix 1. A discount rate of 8% has been used in annualising costs.

3.3.1 Cost Assumptions

The following assumptions have been made in calculating the cost of these two options to reduce methane emissions from enteric fermentation.

- The average methane emission for dairy cows is 100 kg/head/year and for non-dairy cattle is 48 kg/head/year (IPCC, 1996).
- Propionate precursors cost £1628/tonne and probiotics cost £1554/tonne in 1997 (ADAS, 1998). Capital costs are zero.
- Both additives are given to animals as daily supplements. These supplements are generally only given to larger herds (see below). Supplements are given to dairy cows year-round but non-dairy cattle can only be fed with supplements when they are housed inside in the winter. It is estimated that suckler cows can take supplements for 30-40% of the year and beef cattle for 40-50% of the year. For these calculations the two categories are taken together as non-dairy cattle and it is assumed that supplements are taken for 40% of the year.

- 80g/day of propionate precursor is required per head of cattle to give a reduction in methane emissions of 25% (ADAS, 1998). Assuming the propionate precursor is used throughout the year, the total cost of additive per head per year is therefore:

  $$\text{Cost/head/yr} = \frac{1628}{1000} \text{ £/kg} \times 0.08 \text{ kg/day} \times 365 \text{ day/yr}$$

  $$= \text{£}47.5\text{/head/yr}$$

  This gives an emissions reduction of 25% of the total annual emissions = 25 kg CH_4/head/yr for dairy cows. For non-dairy cattle, costs are 40% of £47.5, and reduction is 4.8 kg CH_4/head/yr.

- For probiotics, 50g/day/head is required to give an emissions reduction of 7.5% (ADAS, 1998). Using the same calculation as above, the cost of probiotics is £28.4/head/year and the emissions reduction is 7.5 kg/head/yr for dairy cows and £11.36 and 1.44 kg/head/yr for non-dairy cattle.

3.3.2 Cost-effectiveness of measures

Table 3.4 summarises the cost-effectiveness of measures to reduce emissions of CH_4 from enteric fermentation. It indicates that the use of propionate precursors for dairy cows is the most cost effective option. The same option for non-dairy cattle is less effective because non-dairy cattle emit less than half the methane of dairy cows. Probiotics are less cost-effective because they are expected to give a smaller percentage reduction in methane emissions than propionate precursors. It is important to note that the emissions reductions offered by these options are still very uncertain and further research is required to confirm the data. In addition the estimates below do not take into account any savings from the increased productivity which may result from the use of the additives and the costs given below may thus be an overestimate.

Table 3.4 Cost-effectiveness of Measures to Reduce Enteric Fermentation Emissions

Mitigation Measure	ECU/ t CH_4	ECU/t CO2-equiv
Propionate precursors:		
Dairy cows	2,729	130
Non-dairy cattle	5,686	270
Probiotics:		
Dairy cows	5,440	259
Non-dairy cattle	11,332	540

3.4 EMISSIONS PROJECTIONS AND APPLICABILITY OF MEASURES

3.4.1 Base Line Trends

In order to ensure consistency with other modelling work being carried out for DGXI (e.g. on its acidification policy) base line projections (i.e. under a business as usual scenario) of future livestock numbers are taken from policy support studies being carried out by IIASA. These projections, shown in Table 3.5 were compiled by IIASA (Amann et al, 1996), based on national information and a number of other studies. Overall, in the EU-15, numbers of cattle, sheep and pigs are projected to decline, whilst numbers of poultry will increase slightly.

From Table 3.5 it can be seen that countries projected to increase their cattle populations are Belgium, Ireland, Italy, Spain and Sweden. Generally these countries have the lower productivity and may not be self-sufficient in cattle products. In order to sustain the increased population of cattle with limited land resource, productivity will have to increase, which would reduce methane emissions per unit product. It is apparent that Belgium, Luxembourg and Sweden are switching livestock enterprises from pigs and poultry to cattle, whereas the other countries (Ireland, Italy and Spain) are expanding most of their livestock enterprises.

3.4.2 Emissions under a Business-as-Usual Scenario

Emissions under a business-as-usual scenario to 2020 were estimated using the IPCC methodology (IPCC, 1997); IPCC default emissions factors (in kg CH_4 per head per year) were used apart from the UK where a country specific factor was used for emissions from dairy cattle. Animal numbers in 2020 were estimated by assuming the same rate of change between 2010 and 2020 as assumed in the period (2005-2010); the last period for which the IIASA projections are available. It is not known what assumptions were made about changes in animal size and animal productivity in constructing the IIASA animal projections; emissions per head are thus assumed to remain constant. It is assumed that no measures to reduce emissions are introduced.

Overall for the EU, emissions for 1990 estimated using this methodology are in very close agreement with those reported for the EU, although as shown in Table 3.6, there are some significant differences for some countries (Greece, Ireland, Sweden and Spain). In order to ensure consistency with national estimates, emissions for future years are scaled by the difference in the 1990 emissions estimates and take account of country specific factors.

Projections of emissions to 2020 are shown in Table 3.7; overall in the EU emissions are predicted to fall by 9% by 2010 and 10% by 2020.

3.4.3 Existing Policies and Measures

Table 3.8 summarises national policies for reducing methane emissions from enteric fermentation, and expected national trends in livestock numbers as reported in Member States Second National Communications. In general there are few substantive policies on either animal numbers or productivity.

3.4.4 Applicability Of Measures

The total reduction which might be achieved in 2020 if options were fully implemented is summarised in Table 3.9 for options for which effectiveness data is available. Options which are

Table 3.5 Projection of livestock numbers up to the year 2010 for the EU-15 (million animals)

	Cows[1]			Sheep			Pigs[2]			Poultry		
	1990	2010	Change %	1990	2010	Change %	1990	2010	Change %	1990	2010	Change %
Austria	2.6	2.5	- 1	0.323	0.231	-28	3.8	4.5	20	14.0	17.3	23
Belgium	3.0	5.1	68	0.163	0.110	-33	6.4	4.7	- 26	35.3	27.1	- 23
Denmark	2.2	1.7	- 23	0.100	0.080	-20	9.3	11.7	26	16.2	17.1	5
Finland	1.4	0.9	- 34	0.065	0.070	8	1.3	1.2	- 11	6.0	4.5	- 25
France	21.4	20.9	- 3	12.219	11.000	-10	12.4	17.4	41	236.0	279.3	18
Germany	20.3	15.7	- 23	4.213	3.230	-23	34.2	21.2	- 38	125.5	78.6	- 37
Greece	0.6	0.6	- 1	13.994	15.161	8	1.0	1.5	46	27.4	33.0	20
Ireland	5.9	7.7	31	5.791	6.000	4	1.0	1.9	93	8.9	13.6	52
Italy	8.7	9.5	9	12.094	11.716	-3	9.3	10.5	13	161.0	204.1	27
Luxembourg	0.2	0.4	78	0.007	0.005	-29	0.1	0.1	- 33	0.1	0.1	- 28
Netherlands	4.9	4.8	- 2	1.737	1.340	-23	13.4	11.2	- 16	93.8	79.5	- 15
Portugal	1.3	1.2	- 7	6.424	7.355	14	2.5	1.5	- 41	21.9	26.8	22
Spain	5.1	5.3	3	27.700	26.577	-4	16.0	21.4	34	51.0	56.1	10
Sweden	1.7	1.9	10	0.406	0.483	19	2.3	2.1	- 7	12.3	9.0	- 27
UK	11.9	9.9	- 17	29.678	24.160	-19	7.4	4.8	- 34	141.0	120.5	- 15
EU 15	**91.4**	**88.2**	**- 4%**	**114.91**	**107.52**	**-6%**	**120.3**	**115.6**	**- 4%**	**950.5**	**966.5**	**2%**

1. Cows, include dairy cows & other cattle
2. Pigs, include fattening pigs & sows
3. Poultry, include laying hens, broilers and other poultry

Source: Amann et al, 1996

Table 3.6 Comparison of Emissions Estimates for 1990

	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
Emissions estimate based on IPCC methodology (kt)	178	201	162	94	1442	1398	193	405	697	14	369	149	599	123	995	**7020**
Emissions as reported in Second National Comm. (kt)	146	198	167	90	1430	1430	142	551	643	16	402	124	346	188	1005	**6878**
Difference %	22%	2%	-3%	5%	1%	-2%	36%	-27%	8%	-14%	-8%	20%	73%	-35%	-1%	2%
Difference (kt)	32	3	-5	4	12	-32	51	-146	54	-2	-33	25	254	-65	-10	141

Table 3.7 Enteric Emissions under a Business-as-Usual Scenario (kt CH_4 per year)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15	Change from 1990
1990	146	198	167	90	1430	1430	142	551	643	16	402	124	346	188	1005	**6878**	100%
1994	148	209	162	83	1321	1238	146	569	600	16	380	122	333	198	988	**6443**	94%
2000	147	232	151	71	1289	1173	146	602	607	17	375	124	337	201	882	**6355**	92%
2005	147	254	146	65	1321	1156	146	647	623	20	346	124	337	201	861	**6396**	93%
2010	147	286	133	60	1303	1069	146	653	638	24	362	124	337	202	819	**6302**	92%
2020	147	376	111	50	1270	915	145	666	682	37	414	124	337	202	741	**6217**	90%

Table 3.8 National Policies to Reduce Emissions from Enteric Fermentation

Country	Reported Trends and Policies
Austria	None reported
Belgium	Reducing dairy cattle numbers as a result of the EU CAP, but increasing beef production. Improving manure management methods to reduce methane emissions.
Denmark	Reducing cattle numbers but increasing numbers of pigs.
Finland	Gradual reduction in methane emissions from enteric fermentation is expected due to reducing numbers of dairy cows.
France	Expected increases in productivity for milk and beef (from same number of animals). Increased numbers of pigs and poultry. R&D into enteric fermentation mechanisms. Overall expect small increase in emissions from enteric fermentation.
Germany	Improving livestock digestive efficiency through diet and permitted feed additives.
Greece	No measures reported
Ireland	Increase in animal numbers from 1990 to 1995 is expected to stabilise by 2000.
Italy	Second National Communication not available
Luxembourg	Second National Communication not available
Netherlands	Reducing cattle numbers due to the CAP; expect 0.07 Mt/year reduction from cattle numbers
Portugal	No measures reported
Spain	No measures reported
Sweden	Increasing cattle numbers but efforts to increase milk and meat yields will reduce methane.
UK	Decline in animal population due to 1992 CAP reforms and gradual increases in productivity. BSE may have a future impact. R&D into productivity increases through diet.

Table 3.9 Total Potential Reduction Available from Measures

Measure	Redn per head	Total potential reduction in 2020 (kt)			Cost ECU/t CH_4		Available from Year
		Dairy	Non-dairy	Total	Dairy	Non-dairy	
Propionate precursors	25%	69	270	339	2,729	5,686	2005
Hexose partitioning	15%	41	162	203	not known		2010
Methane oxidisers	8%	22	86	108	not known		2010
Probiotics	8%	21	81	102	5,440	11,332	now
High genetic merit	20%	364	-	364	cost neutral		now

unlikely to be acceptable due to consumer opposition have been excluded. For options involving feed additives, it has been assumed that these are only given to cattle already receiving supplements, and that only cattle kept in herds greater than about 100 heads receive supplements. Thus only 46% of all cattle (15% of dairy cows and 58% of non-dairy cattle) in the EU can be fed with these additives, although it should be noted that there are significant variations in the proportion of cattle in large herds across Europe (Table 3.10).

The reduction potentials in Table 3.9 reflect the maximum potential reduction which might be achieved, if the measure is fully implemented; the reductions are not additive, as only one measure would be applied to any particular herd.

Table 3.10 Percentage of Dairy and Other Cattle in Herds greater than 100 Head

	Dairy	Other cattle
Austria	0%	32%
Belgium	4%	30%
Denmark	11%	100%
Finland	0%	27%
France	2%	97%
Germany	19%	97%
Greece	5%	53%
Ireland	9%	72%
Italy	23%	71%
Luxembourg	2%	100%
Netherlands	12%	100%
Portugal	5%	10%
Spain	8%	64%
Sweden	6%	82%
UK	48%	90%
EU15	**15%**	**83%**

Source: Derived from Eurostat, 1995.

For the purposes of the projections the following assumptions are made about the implementation of the measures:

- **High genetic merit dairy cows** are the most cost-effective option, being cost-neutral or even offering cost savings due to the productivity gains which they bring. However there are some associated problems (fertility, lameness, mastistis etc) which may limit their use, and tend to restrict their use to larger dairy farms. It is therefore assumed that by 2020, 60% of cows in large (>100) dairy herds and 20% of cows in smaller herds are high genetic merit cows.
- **Increased rumen efficiency**- it is assumed that from 2005 propionate precursors are used, and that by 2020, 75% of dairy and non-dairy cattle (excluding the high genetic merit cows) receive this (or other supplements which are under development such as acetogens).

The reductions which these measures achieve are shown in Table 3.11 for the EU and Table 3.12 on a country by country basis. Implementation of both measures is assumed to rise linearly. The reduction in 2020 is 5% of total emissions from enteric fermentation in 1990.

Table 3.11 Estimate of Achievable Reductions from Measures

Measure	Reduction (kt CH_4/yr)		
	2000	2010	2020
High genetic merit cows	24	63	95
Improved rumen efficiency	0	83	242
Total	**24**	**146**	**337**

3.4.5 Projection of Emissions under a With-Measures Scenario

Emissions under a business as usual scenario and with the reduction shown in Table 3.11 (a 'with measures' scenario) are shown in Figure 3.2 for the EU and on a country by country basis in Table 3.13. Emissions in the with measures scenario are 11% below 1990 levels by 2010 and 15% below 1990 levels by 2020.

Figure 3.2 Projection of Enteric Fermentation Emissions in the EU under a Business as Usual and With Measures Scenario.

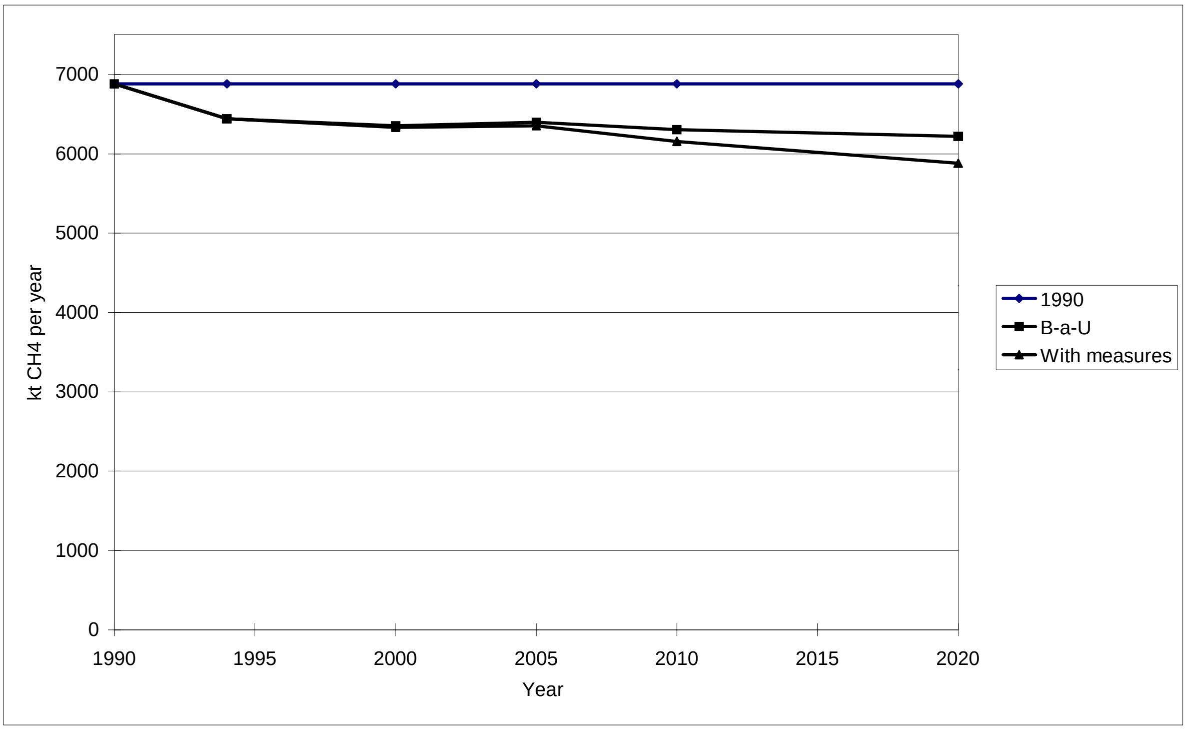


Table 3.12 Estimate of Achievable Reductions in Enteric Fermentation Emissions (kt of CH_4/year)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
1990	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	**0**
1994	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	**0**
2000	1	1	1	0	3	5	0	1	3	0	2	0	1	1	4	**24**
2005	1	1	1	1	6	10	0	2	5	0	3	1	2	1	8	**44**
2010	4	7	3	1	30	29	1	11	16	1	8	2	8	3	21	**146**
2020	10	25	6	3	76	54	2	28	39	2	23	5	20	7	38	**337**

Table 3.13 Enteric Fermentation Emissions under a With Measures Implemented Scenario (kt of CH_4/year)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15	Change from 1990
1990	146	198	167	90	1430	1430	142	551	643	16	402	124	346	188	1005	**6878**	
1994	148	209	162	83	1321	1238	146	569	600	16	380	122	333	198	988	**6513**	-5%
2000	146	231	151	71	1286	1167	146	601	604	17	374	124	335	201	878	**6331**	-8%
2005	145	252	145	65	1315	1146	146	645	618	20	343	123	334	200	853	**6351**	-8%
2010	143	279	130	58	1273	1040	144	642	622	24	354	122	328	198	798	**6156**	-11%
2020	137	351	105	47	1194	861	142	638	643	35	392	119	317	195	704	**5880**	-15%

4. Options to Reduce Emissions from Manure Management

4.1 METHANE EMISSIONS FROM MANURE

4.1.1 Mechanisms by which methane is released

Animal manures contain organic compounds such as carbohydrates and proteins. These relatively complex compounds are broken down naturally by bacteria. In the presence of oxygen, the action of aerobic bacteria results in the carbon being converted to carbon dioxide and, in the absence of oxygen, anaerobic bacteria transform the carbon to methane. Where carbon dioxide is evolved this is part of the natural cycling of carbon in the environment and results in no overall increase in atmospheric carbon dioxide; the carbon dioxide, originally absorbed from the atmosphere through photosynthesis by the plants which formed the livestock feed, is simply being released. In an anaerobic reaction however the carbon dioxide which was absorbed is being converted into and released as methane, and as methane has a higher global warming potential than carbon dioxide, this results in an overall contribution to the greenhouse effect.

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. There is therefore a need to adopt measures which avoid the evolution of methane or convert evolved methane to carbon dioxide.

4.1.2 Release of nitrous oxide

Animal manures contain nitrogen in the form of various complex compounds. If manures are applied to land, then this nitrogen enters the nitrogen cycle, as various bacteria in the soil break down these nitrogen containing compounds. Under the right conditions this can lead to the evolution of nitrous oxide which 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 which increase nitrous oxide releases.

Nitrous oxide emissions are influenced by nitrogen availability, soil moisture content and temperature. Of these, nitrogen availability is the most important (Colbourn 1993) and the most readily controlled. It is therefore important that, where manures are applied to land, the nitrogen load is matched to the crop demand. This will help to avoid excess soil nitrogen and hence the potential for nitrous oxide releases.

4.1.3 Methane emissions from manures

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) and the climate as 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 again increasing the potential for methane release (for wastes which have been spread).

Emissions from manure were calculated using the IPCC methodology (IPCC, 1997). The quantity of manure arising in each EU country was calculated using the same livestock population figures described in Section 3, and data on manure per head from IPCC guidelines (Table 4.1a). It was assumed that all EU countries have manure management systems usage as described by IPCC for Western Europe (shown in Table 4.1b), apart from the UK where more detailed national data on systems used for dairy cows was available[1]. The IPCC guidelines for estimating emissions defines three climatic regions based on annual average temperature, cool (less than 15^{o}C), temperate (15^{o}C to 25^{o}C inclusive) and warm (greater than 25^{o}C). Using data from the 'worldclimate' internet site (Worldclimate 1997), Greece, Italy, Portugal and Spain fall in the temperate region and all other EU countries the cool region.

Using the above assumptions, total emissions from manures in the EU are estimated as 1983 kt in 1990 which is in good agreement with national estimates. Of the emissions, just over half (Figure 4.1) come from cattle and most of the remainder from pig manure. Of these emissions from cattle and pig manure, almost all (97%) arose due to storage of the manure in anaerobic conditions as a liquid/slurry (cattle) and in a pit (pigs) as shown in Figure 4.2.

4.2 MITIGATION OPTIONS

As already mentioned, emissions of methane from animal manures depends on several factors:

- animal numbers;
- feed intake and digestibility;
- climatic conditions;
- manure management.

The potential for reductions in animal numbers through increases in animal productivity were discussed earlier in Section 3, where it was pointed out that numbers will only be reduced if output levels are held constant. Increased animal productivity will reduce the maintenance overhead associated with each unit of product, hence emissions from manure per unit product will be reduced. As before however, total emissions will only be reduced if the level of output remains constant. As output (meat and milk) is influenced by market demand and the Common Agricultural Policy, reducing animal numbers per se, cannot be considered a realistic mitigation option.

[1] In the UK, it is estimated that 43% of dairy cow manures goes to pasture, 38% to liquid storage, 10% to solid storage and 9% to daily spread.

Table 4.1a Methane Emissions per head under Various Management Methods

Cool climate											
Management method			**Liquid/ slurry**	**Solid storage**	**Drylot**	**Daily spread**	**Digester**	**Other**	**Burned for fuel**	**Pit <1 month**	**Pit >1 month**
Methane conversion factor			10%	1.0%	1.0%	0.1%	5%	1%	10%	5%	10%
	t manure head per year	total methane generation potential kg/head	methane generated per head for given management method(kg/head)								
Dairy cows	17.34	294	29	2.9	2.9	0.3	15	2.9	29	N/A	N/A
Beef cattle	8.67	110	11	1.1	1.1	0.1	6	1.1	11	N/A	N/A
Pigs	1.73	54	N/A	0.5	0.5	0.1	3	0.5	N/A	3	5

Temperate climate											
Management method			**Liquid/ slurry**	**Solid storage**	**Drylot**	**Daily spread**	**Digester**	**Other**	**Burned for fuel**	**Pit <1 month**	**Pit >1 month**
Methane conversion factor			35%	1.5%	1.5%	0.5%	5%	1%	10%	18%	35%
	t manure head per year	total CH4 generation potential kg/head	methane generated per head for given management method(kg/head)								
Dairy cows	17.34	294	103	4	4	1	15	3	29		N/A
Beef cattle	8.67	110	39	2	2	1	6	1	11	N/A	N/A
Pigs	1.73	54	N/A	1	1	0	3	1	N/A	10	19

Source: IPCC (1997)

Table 4.1b Manure management system usage for Western Europe (%)

	Lagoon	**Liquid/ slurry**	**Solid storage**	**Drylot**	**Pasture/ range**	**Daily spread**	**Digester**	**Burned for fuel**	**Pit <1 month**	**Pit >1 month**	**Other**
Dairy cattle	0	40	18	0	19	20	0	2	0	0	1
Beef cattle	0	50	0	2	38	0	0	2	0	0	8
Pigs	0	0	21	2	0	0	0	0	3	73	1

Source: IPCC (1997)

Figure 4.1 Estimated Emissions (in 1990) from Animal Wastes by Animal Type (Total = 1983 kt)

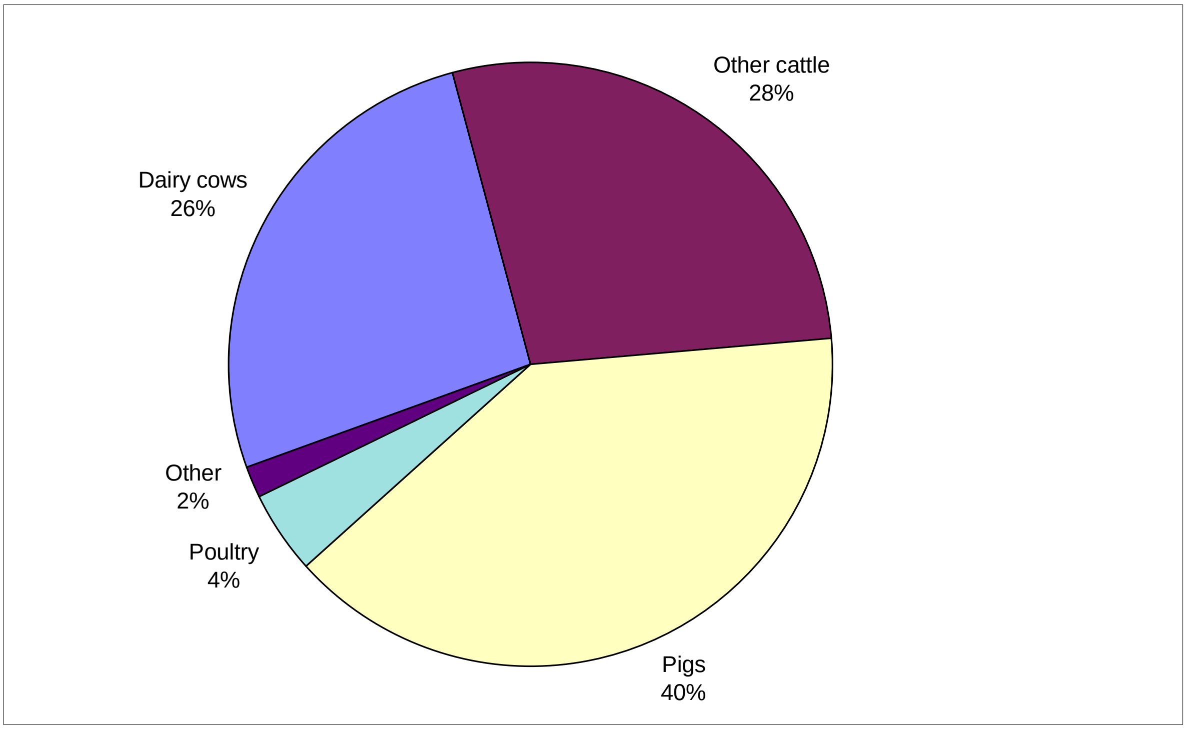
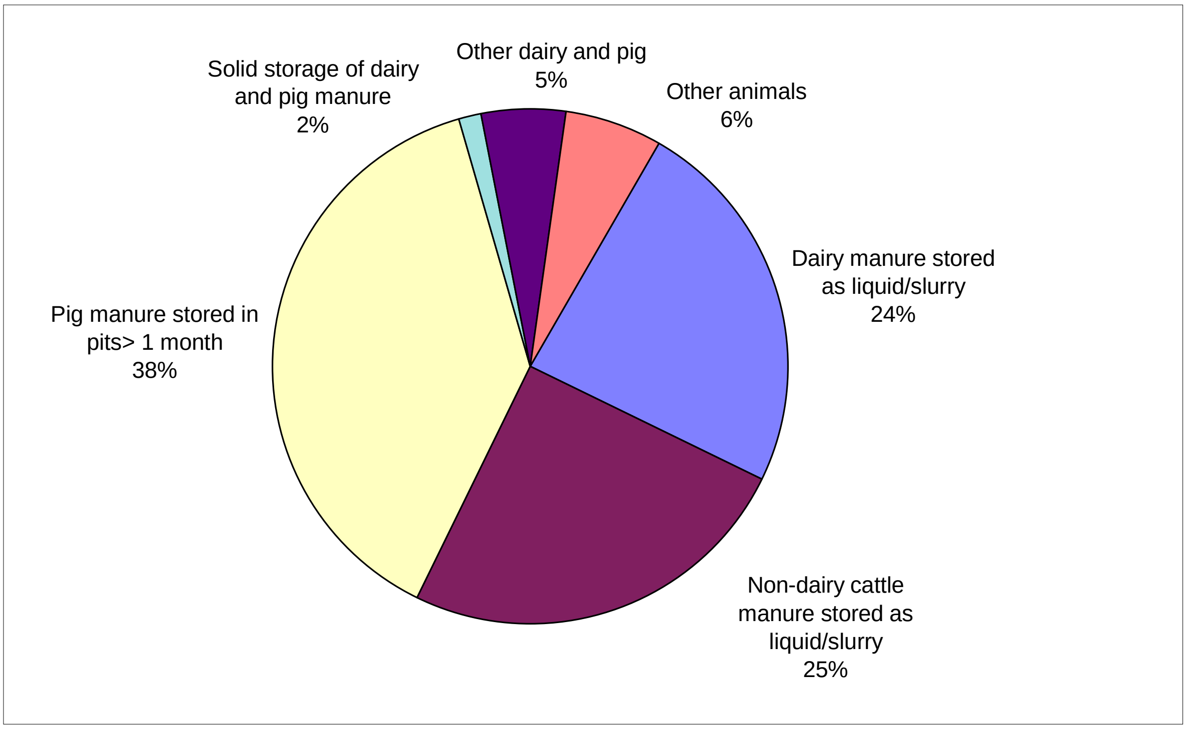


Figure 4.2 Estimated Emissions (in 1990) from Animal Wastes by Management Technique (Total = 1983 kt)



The digestibility of feed given to animals will effect the nature of manure produced and hence its potential for degrading to evolve methane. All of the methods which improve rumen fermentation efficiency which were discussed in Section 3.2, are likely to increase the digestibility of the diet which in turn means that there is less digestible organic matter in the faeces for methanogenic activity. Differences in emission rates between grain and hay fed animals were noted by Lodman et al, (1993); over 7.5 times more methane per kg manure was emitted when animals were fed on grain. Jarvis et al, (1995) showed there was a strong relationship between carbon to nitrogen ratios in the manure and the total amounts of methane emitted (in manure from grazing animals). Methane emissions increased with lower C to N ratios in an exponential fashion. Therefore manure with a high nitrogen content will emit greater levels of methane. Increased rumen fermentation efficiency (hexose partitioning, defaunation, ionophores) will all minimise nitrogen emissions and hence reduce the potential of the manure to emit methane. Research into the composition of livestock manure and its potential methane producing capacity would be beneficial in making improved estimates of methane emissions and to estimate the impact of dietary/efficiency changes on this.

As climate can not be controlled, it can not be considered as a mitigation option; however as warmer climates result in higher potential methane releases, warmer areas may be given higher priority for applying mitigation.

The main strategy for reducing emissions from manure is thus in influencing the choice of manure management methods as these will determine whether the degradation process is predominantly aerobic or anaerobic and hence whether carbon dioxide or methane are evolved. They also determine how much organic carbon gets locked up in the soil. Any methane reduction strategy should therefore seek to avoid uncontrolled releases from anaerobic degradation, either by ensuring aerobic digestion or recovering methane evolved from anaerobic degradation. Consequently, practical measures to reduce methane releases resulting from animal manures fall into two categories:

- measures to ensure aerobic decomposition and avoid methane evolution;
- measures to convert evolved methane to carbon dioxide.

4.2.1 Aerobic decomposition

For Western Europe, 40% of dairy cattle manure is treated as liquid slurry (Table 4.1) releasing 10 to 35% of the manure’s methane potential, depending on annual average temperature, and for beef cattle 50% of manure is treated in this way (IPCC 1997). 73% of pig manure is treated by pit storage for more than 1 month, again releasing 10 to 35% of the manure’s methane potential, depending on annual average temperature. These relatively large emissions are due to anaerobic storage conditions. Farming practices which ensure that manure degrades aerobically can therefore reduce methane emissions from current levels.

4.2.1.1 Livestock management

When manures are spread thinly across the land then decomposition will be largely aerobic, and livestock kept in pastures or rangeland result in manure being naturally widely spread. Within the EU, the main market driver is currently for cheap food products and this has led to intensive farming methods, which normally involve animals being housed rather than pasture based systems. There is some potential for a market led switch to greater use of pasture as a result of public concern for animal welfare and an increasing demand for organically produced foodstuffs. However, for the purposes of this study it is considered that any switches to more extensive farming systems will be small for the foreseeable future and therefore are not considered further as a methane reduction option.

4.2.1.2 Land application management

In EU countries intensive livestock management regimes or cold climates lead to animals being housed for at least part of the year. Under these circumstances high concentrations of manure arise, often on farms with little land, and manure storage is required. Stored manure will degrade anaerobically and the sooner the manure is spread onto the land the less anaerobic decomposition will have taken place and the less methane evolved.

When spreading to land, anaerobic degradation can be minimised by avoiding anaerobic soil condition (e.g. waterlogged soils or compacted soils) and by spreading thinly and evenly. Spreading has the potential to release emissions of both nitrous oxide and ammonia, and their minimisation should also be considered. As already mentioned manure applications should be matched to crop nitrogen needs to avoid excess soil nitrogen and hence nitrous oxides release. Up to 90% of ammonia loss to atmosphere occurs within 12 hours of spreading (Burton et al 1997) hence it is important to incorporate the manure into the soil as effectively as possible to minimise this loss (this equally applies to spreading of manures treated by anaerobic digestion).

According to IPCC (IPCC 1997), in EU climates, daily spreading of manure results in the release of 0.1 to 0.5% of the manure's methane potential. This compares to releases of 10 to 35% of the manure's methane potential from the manure treatment methods most commonly used in the EU, and switching to daily spreading of manure could therefore reduce methane emissions from these sources significantly.

However, there are limitations to the applicability of daily spreading e.g.:

- during times of high rainfall there will be a risk of run-off causing water pollution;
- application of nutrients in the manure can not always be matched to crop requirements hence the risk of nitrous oxide release and eutrophication is increased;
- large numbers of vehicle movements may result in damage to soil structure;
- access to the land may be restricted at certain times of the year;
- increased labour requirement.

The overall applicability of this option is discussed below in Section 4.4.4.

4.2.1.3 Aerobic treatments

Aerobic treatments can be applied to liquid manures through aeration and to solid manures by composting. Aeration involves dissolving sufficient oxygen in the liquid manure to allow bacteria to oxidise the organic carbon. Systems for aeration involve mechanical methods for passing air through the liquid, usually driven by electric motors. When considering methane reduction using this technique, emissions of greenhouse gases from electricity generation should be deducted from any saving. Other factors to consider are that aeration may leave up to 70% of the total organic load (Burton et al 1997) which may subsequently degrade anaerobically if the liquid manure is stored and that losses of 4 to 11% of the total nitrogen as nitrous oxide have been reported from the aeration of liquid pig manure (Burton et al 1993). Therefore there is considerable uncertainty as to the effectiveness of this treatment option.

Solid manures can be aerobically treated by composting. This may require de-watering of liquid manures or addition of other dry organic materials to increase porosity and penetration of air. Also organic material may have to be added to increase the carbon:nitrogen ratio to levels suitable for composting. As with aeration, composting requires energy input to turn the compost ensuring good mixing and air penetration. Greenhouse gas emission attributable to this energy input should again be deducted from any savings in methane emissions. Ammonia releases to atmosphere should also be considered for composting, losses of 3 to 75% of the initial ammoniacal nitrogen content have been reported (Burton et al 1997).

4.2.2 Conversion of methane to carbon dioxide

4.2.2.1 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 nitrogen, sulphur compounds, volatile organic compounds and ammonia. This biogas has a typical calorific value of 17 to 25 MJ/m^3 and can be combusted directly in modified gas boilers, used to run an internal combustion engine or simply flared.

Applying this process to animal manures ensures that most of the carbon is ultimately converted to carbon dioxide before being released to atmosphere. Typically, between 40% and 60% of the organic matter present is converted to biogas. The remainder consists of a relatively odour free residue with an appearance similar to peat, which has some value as a soil conditioner and also, with some systems, a liquid residue which has potential as a fertiliser.

According to IPCC (IPCC 1997), AD releases 5% of the manure’s total methane potential. This release is largely due to further decomposition of the digested material on removal from the digester. These emissions can be minimised through covered storage with gas collection or by reducing storage times. Leakage of biogas is another potential source of methane emissions but this should be minimised by good system design and maintenance. AD plants vary from small ‘low-tech’ on-farm systems to highly engineered centralised plant. In the latter case commercial drivers ensure that plants recover all practicable methane for energy generation and fugitive methane emissions will be minimised. For example covered digestate storage with gas collection is becoming the industry norm for centralised AD plants. These types of plant are likely to operate with less than 5% methane releases, but this figure will be used here as a conservative assumption.

It should be noted that an additional benefit of utilising biogas from AD plants to produce heat and electricity is that this will offset greenhouse gas emissions resulting from fossil fuel energy sources and this should be accounted for when considering the contribution of AD to wider greenhouse gas reduction.

There are some constraints on the applicability of this measure. Any plant needs sufficient feedstock from the surrounding area without incurring excessive transport costs (both financial and environmental). For on-farm plants this is not an issue and there is no real limit to how small a digester can be. In practice, for commercial plant rather then self-built units, about 50m^3 is the smallest plant being installed. Such a plant would be suitable for about 50 cattle or 500 pigs. For centralised plant the maximum catchment area will be determined by local factors such as transport costs and the fuel value of the feedstock. On environmental grounds increased transport leads to increased emissions, including oxides of carbon. Lindboe (1994) calculated that, where the mean distance from farm to plant was 8 km, transport would release 2 kg of carbon dioxide per m^3 of feedstock or about 2% of the net carbon dioxide savings (avoided methane, avoided natural gas use in energy generation and avoided emissions from mineral fertilisers) associated with the plant being studied. Many of the Danish plants serve farms within a 10 km radius or less (Danish Energy Agency 1992). It should be noted that these plants are often for pig slurries which have very low solids content. Thicker slurries can be transported greater distances for the same environmental cost.

A further constraint is that the surrounding land must have sufficient capacity to accept the nutrients in the feedstock. Nitrate loadings are often the limiting factor and applications of digested slurry should match nutrient loadings to crop requirements to reduce the risk of leaching and eutrophication.

In practice the main constraint on the application of AD to date has been economics. Looking at EU countries where AD deployment is most extensive gives a good indication of how reliant the technology has been on financial incentives. Danish plants have received support in the form of capital grants, low cost loans and tax incentives. In Germany and Italy, premium electricity prices have been available. In the UK, capital grants were available for a few years and many of the farm scale digesters were built during this period. All the proposed centralised schemes in the UK have contracts which guarantee the purchase of electricity at a premium price for 15 years. In Austria, capital grants have supported most of the installed digesters.

Market, as opposed to government financial, incentives may also play a role. For example, there is increasing interest in the use of AD digestate as a substitute for mineral fertilisers. Development of this market has the potential for useful income to plant operators who don’t have land on which to spread the digestate.

The level and type of support has influenced the technology as well as the numbers of units installed. Denmark, which is often considered the lead EU country for AD technology, has about 20 centralised digesters installed (FAIR 1997). Capital grants and cheap loans for community based schemes plus a large degree of centralised planning, have contributed to a focus on larger schemes. These plants are typically 500 to 10,000 m^3 and are technically advanced including high levels of automation. In contrast, Germany has over 300 digesters (FAIR 1997) which are typically 100 to 1000 m^3. Many German plants are of simpler design, utilising readily available materials such as modified car engines and old liquid storage tanks. As the main support for German plants is through electricity prices, there is no strong incentive towards centralisation but there is an incentive to minimise capital costs.

As a result of the large degree of financial support available in Denmark, their plants are not typical of those being installed elsewhere in the EU. For example centralised plant being proposed in the UK have lower capital costs and lower degrees of automation. If the technology was to be encouraged as a methane abatement option, greater deployment is likely to be achieved through the lower technology options such as those being installed in Germany, Italy and the UK.

As well as economic factors, other drivers influence the deployment of AD. Some farmers are prepared to operate digesters at a cost, rather than as a source of net income. This is due to other perceived benefits such as reduced odour problems, avoiding prosecution for water pollution and improved manure management.

4.2.2.2 Covered Lagoons

In some cases liquid manures are stored in open pits or lagoons. These lagoons often have an impervious liner to prevent leaching into soils and ground water. Slurry lagoons also undergo anaerobic digestion and hence emit methane. Uncovered lagoons are not used to a significant extent within the EU, however where they are used about 90% of the manure's methane potential can be released to atmosphere (IPCC 1997).

Covered lagoons, which collect and use the methane evolved, can be considered as simple, low cost AD plants and have a similar potential for methane reduction. Methane releases from covered lagoons may be higher than for AD plant due to difficulties in sealing over the large areas concerned. The potential use of covered lagoons will be lower than for AD due to land constraints where intensive farming is carried out. This will be especially the case for pig and poultry farming.

4.3 COST OF OPTIONS

Four options to reduce CH_4 emissions for which some cost data is available, have been identified:

- Daily spread of manure;
- Anaerobic Digestion: centralised plant producing electricity and heat
- Anaerobic Digestion: small-scale plant producing electricity and heat
- Covered lagoon;

In all cases the methane reduction achieved by the use of each measure has been compared to the most common current manure management method. For cattle this is liquid/slurry storage which accounts for 40% of dairy cattle manures and 50% of non-dairy cattle. In liquid storage, 10% of potential methane is released in cool climates and 35% in temperate climates. For pigs the most common management option (73%) is storage in a pit for longer than one month; this results in 10% of potential methane being released in cool climates and 35% in temperate climates.

The methodology used to calculate the annualised cost (in 1995 ECU) of the measures, and details of exchange rates and conversions from different years are described in Appendix 2. A discount rate of 8% has been used in annualising costs.

4.3.1 Cost Assumptions

4.3.1.1 Aerobic Digestion: daily spread of manure

The cost–effectiveness of this measure has been estimated using data collected from the Farm Management Pocketbook (1996). The following assumptions have been made:

- The non-recurring cost is zero.
- The annual recurring costs are (in 1995 prices) £10 per pig per year, £50 per head of beef per year and £100 per dairy cow per year.
- The option has a lifetime of 1 year and requires 0 years to install.

- The maximum methane emission potential per animal is as above. Under 'most common' current management practices 10 per cent (cool climates) to 35 per cent (temperate climates) of the maximum potential methane is released. The daily spread of manure results in the release of only 0.1 per cent (cool climates) to 0.5 per cent (temperate climates) of the maximum potential methane. In cool climates the daily spread of manure thus results in annual savings in methane emissions of 0.0053 tonnes per pig, 0.0109 tonnes per head of beef, and 0.0291 tonnes per dairy cow. For temperate climates, these figures are 0.0186 tonnes per pig, 0.0380 tonnes per head of beef, and 0.1014 tonnes per dairy cow.

4.3.1.2 Anaerobic Digestion: centralised plant

Cost data is available for two centralised plant, one in the UK and one in Denmark. Data for the former comes from the UK DTI's New and Renewable Energy Programme (ETSU, 1997), and for the latter from Danish Energy Agency (1994).

For the UK plant, the following assumptions have been made:

- The non-recurring capital cost is £4 million per plant (in 1996 prices). Purchased equipment costs account for 84.5 per cent of this figure; the remainder is made up of associated costs.
- Annual recurring costs are £0.6 million per plant per year (in 1996 prices). Energy costs, material costs and labour costs account for 3, 58 and 39 per cent of this figure, respectively. The plant produces 8 GWh of electricity and 9.6 GWh of heat per year. The selling price of electricity is £0.021/kWh and the price of fuel oil is 0.829 p/kWh (in 1996 prices). Therefore, the value of the combined heat and power output is £247,584 per plant per year. The sale of digestate by-products results in an additional income of £0.5 million per plant per year.
- The option has a lifetime of 20 years and requires 1 year to install.
- The plant is designed to handle about 200,000 tonnes of animal manure per year. Dairy cows produce 17.3 tonnes of manure per head per year, beef cattle produce 8.7 tonnes of manure per head per year, and pigs produce 1.7 tonnes of manure per head per year. Under current management practices 10 per cent (cool climates) to 35 per cent (temperate climates) of the maximum potential methane is released. Anaerobic digestion results in the release of only 5 per cent of the maximum potential methane. In cool climates the use of this plant therefore results in annual savings in methane emissions of 312 tonnes, 127 tonnes, and 170 tonnes, for pig, beef or dairy manure, respectively. The use of this plant in temperate climates results in annual savings in methane emissions of 1,873 tonnes, 761 tonnes, and 1,017 tonnes, respectively.

For the Danish plant, the following assumptions have been made:

- The non-recurring capital cost is £80 million Danish Kroner (DKK) per plant (in 1996 prices). Purchased equipment costs account for 84.5 per cent of this figure; the remainder is made up of associated costs.
- Annual recurring costs are £6.6 million DKK per plant per year (in 1996 prices). Energy costs, material costs and labour costs account for 3, 58 and 39 per cent of this figure, respectively. The plant produces biogas equivalent to 4.2 million m^3 of natural gas a year. The selling price of the gas is 20.75 DKK per GJ (in 1996 prices). Therefore, the value of the gas sales is 2.98 million DKK per plant per year. The sale of digestate by-products results in an additional income of 1.4 million DKK per plant per year.
- The option has a lifetime of 20 years and requires 1 year to install.

- The plant is designed to handle about 200,000 tonnes of animal manure per year. Dairy cows produce 17.3 tonnes of manure per head per year, beef cattle produce 8.7 tonnes of manure per head per year, and pigs produce 1.7 tonnes of manure per head per year. Under current management practices 10 per cent (cool climates) to 35 per cent (temperate climates) of the maximum potential methane is released. In this type of anaerobic digestion plant, leakage is kept to a minimum and as a result only 3 per cent of the maximum potential methane is released. In cool climates the use of this plant therefore results in annual savings in methane emissions of 437 tonnes, 178 tonnes, and 238 tonnes, for pig, beef and dairy manure, respectively. The use of this plant in temperate climates results in annual savings in methane emissions of 1,998 tonnes, 812 tonnes, and 1,084 tonnes, respectively.

4.3.1.3 Anaerobic Digestion: Small Scale Plant

Data are available for two small scale plant, one in Italy and one in Germany. Data for the Italian plant is drawn from Piccinini, (1996) and for the German plant from Kuatorium fur Technik und Bauwesen in der Landwirtschaft, (1992) and Fachverband Biogas eV (1993).

The following assumptions have been made for the Italian plant:

- The non-recurring capital cost is 176 million Lira per plant (in 1993 prices). Purchased equipment costs account for 84.5 per cent of this figure; the remainder is made up of associated costs.
- The annual recurring costs are 11 million Lira per plant per year (in 1993 prices). Energy costs, material costs and labour costs account for 3, 58 and 39 per cent of this figure, respectively. The plant produces 180 MWh of electricity and 440 MWh of heat per year. The price of electricity is 98 lira/kWh and the price of fuel oil is 137 065 lira/t (in 1993 prices). Therefore, the value of the combined heat and power output is 23 million lira per plant per year.
- The option has a lifetime of 20 years and requires 1 year to install.
- The plant is designed to handle about 22,000 tonnes of animal manure per year. Dairy cows produce 17.3 tonnes of manure per head per year, beef cattle produce 8.7 tonnes of manure per head per year, and pigs produce 1.7 tonnes of manure per head per year. The maximum potential methane emission per animal is as above. Under current management practices 10 per cent (cool climates) to 35 per cent (temperate climates) of the maximum potential methane is released. Anaerobic digestion results in the release of only 5 per cent of the maximum potential methane. In cool climates the use of this plant therefore results in annual savings in methane emissions of 14 tonnes, 6 tonnes, and 8 tonnes, for treating pig, beef or dairy manure, respectively. The use of this plant in temperate climates results in annual methane savings of 86 tonnes, 36 tonnes, and 48 tonnes, respectively.

The following assumptions have been made for the German plant:

- The non-recurring capital cost is 175,000 DM per plant (in 1992 prices). Purchased equipment costs account for 84.5 per cent of this figure; the remainder is made up of associated costs.
- The annual recurring costs are 2,000 DM per plant per year (in 1992 prices). Energy costs, material costs and labour costs account for 3, 58 and 39 per cent of this figure, respectively. The plant produces 380 MWh of electricity and 460 MWh of heat per year. The price of electricity is 11.09 DM/MWh and the price of fuel oil is 17.58 DM/MWh (in 1992 prices). Therefore, the value of the combined heat and power output is 12,300 DM per plant per

year. A second option, where the plant produces heat only is also examined. In this case the capital costs are reduced by 20,000 DM and the available heat is increased from 460 MWh/yr to 740 MWh per year.

- The option has a lifetime of 20 years and requires 1 year to install.
- The plant is designed to handle about 9,000 tonnes of animal manure per year. Dairy cows produce 17.3 tonnes of manure per head per year, beef cattle produce 8.7 tonnes of manure per head per year, and pigs produce 1.7 tonnes of manure per head per year. The maximum potential methane emission per animal is as above. Under current management practices 10 per cent (cool climates) to 35 per cent (temperate climates) of the maximum potential methane is released. Anaerobic digestion results in the release of only 5 per cent of the maximum potential methane. In cool climates the use of this plant therefore results in annual savings in methane emissions of 34 tonnes, 14 tonnes, and 19 tonnes, for treating pig, beef or dairy manure, respectively. The use of this plant in temperate climates results in annual methane savings of 206 tonnes, 84 tonnes, and 112 tonnes, respectively.

4.3.1.4 Anaerobic Digestion: covered lagoon

An indicative calculation has been made based on an American case study (US, 1993). The case study involves a installation designed to treat slurry from a 5,000 pig unit, a 1,000 beef cattle unit, or a 500 dairy cattle unit. The following assumptions have been made:

- The non-recurring capital cost is $US 355,000 (in 1993 prices). This equates to $US 71 per pig, $US 355 per head of beef, and $US 710 per dairy cow.
- The annual recurring costs are $US 11,500 per year (in 1993 prices). That is, $US 2.30 per pig per year, $US 11.50 per head of beef per year, and $US 23.00 per dairy cow per year. It is estimated that 1 GWh of heat can be produced from gas evolved from the lagoon per annum. The price of oil is about $US 0.00996/kWh (in 1993 prices). The value of heat output from the lagoon is thus $US 9,960 per year, or $US 1.99 per pig per year, $US 9.96 per head of beef per year, and $US 19.92 per dairy cow per year.
- The option has a lifetime of 10 years, and requires 1 year to install.
- The maximum potential methane emission per pig, head of beef, and dairy cow is 54 kg/year, 110 kg/year and 294 kg/year, respectively. Under the most common current management practices in the EU 10 per cent (cool climates) to 35 per cent (temperate climates) of the maximum potential methane is released. Anaerobic digestion in covered lagoons results in the release of only 5 per cent of the maximum potential methane. In cool climates the use of a covered lagoon therefore results in annual savings of methane emissions of 0.0027 tonnes per pig, 0.0055 tonnes per head of beef, and 0.0147 tonnes per dairy cow. In temperate climates, these figures are 0.0162 tonnes per pig, 0.033 tonnes per head of beef, and 0.0882 tonnes per dairy cow.

4.3.2 Cost-effectiveness

The cost-effectiveness of the above measures is summarised in Table 4.2 in ECU/t of CH_4 abated; the cost -effectiveness in terms of ECU/t of CO_2- equivalent is given in Table 4.3. Table 4.2 indicates that overall, the most cost-effective measure to reduce methane emissions from the agricultural sector is anaerobic digestion, where the biogas produced is used to generate electricity and or heat. For large scale plant, the Danish plant, which is of a highly engineered type, is five to seven times more expensive than the UK plant, even allowing for the lower ‘leakage’ rate (3% compared to 5%). The small scale plant are more cost effective, due mainly to a simpler design leading to lower capital costs.

Table 4.2 Cost-effectiveness of Measures to Reduce Emissions from Animal Manures (best estimate) (ECU per tonne of CH_4)

	Cool climate			Temperate climate		
	Pigs	Dairy	Beef	Pigs	Dairy	Beef
Daily spread of manure	2264	4124	5505	645	1183	1579
AD - Centralised						
UK	450	828	1106	75	138	184
Denmark	2287	4211	7878	500	921	1723
Small scale						
Germany (CHP)	286	526	703	48	88	117
Germany (heat only)*	95	175	234	16	29	39
Italy (CHP)	181	334	446	30	56	74
Covered lagoon	3503	6448	8617	584	1075	1436

* Due to the relative costs of light fuel oil and electricity in Germany, a greater revenue is achieved by running the plant in heat only mode, thus making it a more cost-effective option. In other countries a heat only plant may be less cost-effective than one run in CHP mode.

Table 4.3 Cost-effectiveness of Measures to Reduce Emissions from Animal Manures (best estimate) (ECU per tonne of CO_2)

	Cool climate			Temperate climate		
	Pigs	Dairy	Beef	Pigs	Dairy	Beef
Daily spread of manure	108	196	262	31	56	75
AD - Centralised					32	
UK	21	39	53	4	7	9
Denmark	109	201	375	24	44	82
Small scale						
Germany (CHP)	14	25	33	2	4	6
Germany (heat only)*	5	8	11	1	1	2
Italy (CHP)	9	16	21	1	3	4
Covered lagoon	167	307	410	28	51	68

* Due to the relative costs of light fuel oil and electricity in Germany, a greater revenue is achieved by running the plant in heat only mode, thus making it a more cost-effective option. In other countries a heat only plant may be less cost-effective than one run in CHP mode.

Leakage rates in the small scale plant, have been assumed to be the same as that at the (UK) large scale plant; it has been suggested that at on-farm small scale plants leakage rates may be higher, as there is not the same incentive as in large scale commercial plant to ensure that as much biogas is collected as possible to generate power and supply income. However, it could be argued that if such plant are to be installed with the primary aim of reducing methane emissions (rather than as a method of waste disposal or controlling odour as has often been the case in the past), then an emphasis should be placed on making the plant as leak tight as possible.

In all cases, costs are based on finding a use for the heat and power produced; and it is possible that with on-farm systems, a suitable heat use may not be available; this lack of income would reduce the cost-effectiveness of the measure. The effect of this depends on the relative prices received for electricity and heat. For example, if the Italian small scale plant was to receive an income only for the electricity produced and not the heat, the cost-effectiveness of the plant would only be marginally increased (to 451 ECU/t CH_4) whereas the cost-effectiveness of the German plant is doubled (to 1457 ECU/t CH_4). Both costs are for treating beef manure in a cool climate. Similarly if there was no market for the digestate produced from large scale plant, their cost effectiveness would be reduced. For example, in the case of the UK plant, if no income was received from the digestate, then the cost of the measure would be almost six times higher at 5932 ECU/t CH_4 for beef cattle in a cool climate; for the Danish plant, where costs are dominated by the capital cost, the cost is increased by only 13%, to 8942 ECU/t CH_4 for beef cattle in a cool climate.

The cost-effectiveness of the covered lagoon is poor, with income for the heat produced insufficient to cover running costs. Similarly the daily spread of manure is not very cost-effective compared to most anaerobic digestion options, but is of the same order as the more expensive Danish plant, and indeed the UK plant if no income were received for digestate.

In all cases, measures implemented in a temperate region are three to six times more cost-effective than the same measure taken in a cool climate, due to the fact that in warmer climatic conditions there is greater potential for methane evolution. Measures utilising pig manure are always more cost-effective than those using beef or dairy manure due to the different methane producing potential (Table 4.1a) of the wastes.

4.4 APPLICABILITY OF MEASURES

4.4.1 Base Line Trends

Methane emissions from animal manures depends on manure management, climatic conditions animal feeding and animal numbers. To predict future trends in emissions it is therefore necessary to examine how these causal factors will develop. This study initially estimates future emissions assuming a baseline where management techniques remain constant, and manure production per head of animal and the methane production potential of manures remain constant. Changes to management techniques are considered as the basis of the mitigation options.

The projections of animal numbers produced by IIASA and given in Section 3 (Table 3.1) are used as the basis for projecting future emissions. Overall, in the EU-15, numbers of cattle, sheep and pigs are projected to decline, whilst numbers of poultry will increase slightly. The IIASA predictions are reinforced regarding cattle numbers by EU predictions. The EU (DG VI, 1997) has stated that "beef consumption is expected to gradually recover from the 1996 shock and return to its long term (declining) trend" and that "total milk production is forecast to decline slightly from 121.6 million tonnes in 1996 to 118.1 million tonnes in 2005". This EU document also suggests that pig meat prices may weaken up to 2006. Weakening prices are likely to lead to reduced animal numbers.

When considering the implications of these changes in animal numbers for methane emissions, it is necessary to look at trends in individual countries. This is because, although the EU aggregate livestock numbers may be decreasing, if this hides an underlying shift in production to warmer climates there may still be increased methane production.

4.4.2 Emissions under a Business-as-Usual Scenario

Emissions under a business-as-usual scenario to 2020 were estimated using the IPCC methodology (IPCC, 1997) and assumptions discussed in Section 4.1.3 As previously (Section 3) animal numbers were estimated by assuming the same rate of change between 2010 and 2020 as assumed in the last period (2005-2010) for which IIASA projections are available.

Overall for the EU, emissions for 1990 estimated using this methodology are about 10% below emissions reported by countries in their Second National Communications. As shown in Table 4.4 there are more significant differences for some countries, with emissions in Belgium, Denmark and Germany underestimated by 45 to 70% compared to national submissions, and emissions for Italy and Sweden overestimated by about 100%. The relevant Second National Communications do not provide enough detail to allow the precise reasons for these differences to be resolved. For Belgium, Denmark and Germany; it is thought that they are due either to different assumptions about the methane producing potential of the manures, the release rates associated with various management techniques, or the proportions of wastes going to different disposal routes. In the case of Italy it is possible that the difference is due to Italy being classed as a cool country rather than the temperate classification assumed in this study; calculating emissions on this basis gives an estimate of 138 kt for 1990, which is 28% lower than the national estimate.

In order to ensure consistency with national estimates, emissions for future years are scaled by the difference in the 1990 emissions estimates; these projections to 2020 are shown in Table 4.5. Overall in the EU emissions are predicted to fall (from 1990 levels) by 5% by 2010 and 9% by 2020. The fall is mainly due to a large reduction (275 kt) in emissions in Germany (the largest single contributor to emissions) and a smaller decrease in the UK (of 38 kt) which are partially compensated by an increase in emissions (of 85 kt) from Spain and smaller increases in Belgium and Italy (35 and 23 kt respectively).

4.4.3 Existing Policies and Measures

Table 4.6 summarises national policies for reducing methane emissions from animal manures as reported in Member States Second National Communications. Predictions of national trends in livestock numbers which will also influence emissions from manures were detailed in Table 3.8. These measures have not been included in the Business as Usual scenario. In some cases e.g. Finland and France, the predicted trend is significantly different from those in the Business as Usual scenario.

4.4.4 Applicability Of Measures

Anaerobic digestion would produce emissions savings if applied to beef and dairy manures currently sorted as liquids/slurries and pig manure kept in pits for longer than a month. The daily spread of manure would bring about savings in all of these cases, and a small reduction for dairy and pig manures kept in solid storage. However this latter option is not considered as due to the smaller reduction in emissions achieved, the cost per tonne abated of this measure would be about five times that shown in Table 4.2.

Table 4.4 Comparison of Estimates of Annual Emissions from Animal Wastes in 1990

	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
Emissions estimate based on IPCC methodology (kt)	39	55	60	18	247	327	43	53	402	2.1	108	78	387	26	138	**1983**
Emissions as reported in Second National Comm. (kt)	27	176	162	11	168	614	23	52	192	2.0	103	68	465	12	125	**2199**
Difference %	49%	-69%	-63%	66%	47%	-47%	85%	2%	109%	6%	5%	15%	-17%	120%	11%	-10%
Difference (kt)	13	-121	-102	7	79	-287	20	1	210	0.1	5	10	-77	14	13	-215

Table 4.5 Emissions from Animal Wastes under a Business-as-Usual Scenario (kt CH_4 per year)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15	Change from 1990
1990	27	176	162	11	168	614	23	52	192	2	103	68	465	12	125	**2199**	-
1994	28	186	182	10	160	521	24	55	178	2	102	61	486	12	122	**2130**	-3%
2000	29	185	188	9	164	481	25	60	185	2	101	54	550	12	105	**2150**	-2%
2005	29	186	191	9	173	475	26	65	196	2	91	54	550	12	105	**2163**	-2%
2010	29	189	176	8	173	423	26	65	201	3	89	54	550	12	98	**2097**	-5%
2020	29	211	149	7	172	339	27	66	215	4	89	54	550	12	87	**2009**	-9%

Table 4.6 National Policies to Reduce Emissions from Animal Manures

Country	Reported Trends and Policies
Austria	Research planned on biogas utilisation from manure storage.
Belgium	Improving manure management methods to reduce methane emissions.
Denmark	Encouraging large scale biogas plants to use methane from stored manure (20 built to date).
Finland	Emissions from manure will not change significantly as the number of pigs, the largest source of emissions, will remain constant.
France	Local environmental protection measures will lead to an increase of 28% in emissions from stored manure.
Germany	Studying improved manure management methods (such as mixing in straw) and utilisation of biogas from manure.
Greece	No measures reported
Ireland	Capital allowances for investment in pollution control facilities introduced recently.
Italy	Second national communication not available
Luxembourg	Second national communication not available
Netherlands	Reducing cattle numbers due to the CAP and the introduction of better manure management methods are expected to lead to a reduction of 35 kt/ year of methane (no specific measures).
Portugal	No measures reported
Spain	No measures reported
Sweden	Measures against ammonia leaching from manure storage expected to reduce methane emissions.
UK	Codes of Good Agricultural Practice for the Protection of Air will encourage the safe and efficient utilisation of animal waste. Anaerobic digestion projects are eligible for support under NFFO.

Source: Second National Communications

Table 4.7 Estimate of Total Technical Potential of Measures in 2020 (kt of CH_4/yr)

	Cool climate			Temperate Climate			Total
	Pigs	**Dairy cattle**	**Non-dairy**	**Pigs**	**Dairy cattle**	**Non-dairy**	
Liquid slurry/pits to daily spread	235	141	266	393	106	176	1318
Liquid slurry/pits to AD (3% leakage)	166	100	188	365	99	163	1081
Liquid slurry/pits to AD (5% leakage)	119	71	134	342	92	153	912

Note: the emissions reductions estimated in Table 4.7 are not additive

The total potential saving which might be achieved if all manures in liquid slurry or pit storage were treated via anaerobic digestion or daily spread is shown in Table 4.7. In order to avoid overestimating potential reductions, estimates for countries where the IPCC methodology provided an emissions estimate greater than the national estimates (see Section 4.4.2) were scaled by the same factor used to scale the emissions projections.In practice, several factors would practically limit the implementation of both anaerobic digestion and daily spreading of manure. In the case of anaerobic digestion, there is a minimum size for small-scale plant (100 dairy cattle, or 200 beef cattle or 1000 pigs). If it is assumed that this must be supplied from one, or at the most two farms, then it is possible to make an estimate of the percentage of animals (and hence manure) generated in farms above this minimum size. This is shown in Table 4.8.

In the case of centralised digestion, enough livestock/farms must be within a relatively small radius (typically 10 km^2) of the plant so that transport costs can be kept low. A rough estimate of livestock density was made by dividing livestock numbers by agricultural area in each Member State. The results were then compared with the livestock density within a 15 km radius of the plant to give an indication of whether the density of livestock is high enough to support centralised plant (Table 4.8). This probably gives conservatively based indications, as centralised AD plant may take more than one type of manure, and livestock holdings are often regionally grouped so that an average livestock density is not a good indication. (A higher than average supply area was assumed to try and at least partially allow for these biases). In terms of the choice between small scale and centralised AD, while the estimates in Table 4.2 indicate that small scale digestion is more cost-effective than centralised digestion (assuming leakage is not higher), existing policies such as a premium price for electricity from centralised facilities, may influence the choice between farm scale and centralised digestion. Existing plant and policies are briefly summarised in Table 4.8.

In the case of daily spreading of manure, a limitation in areas where there is a lot of intensive farming may be the availability of land on which to spread the manure, and also, as discussed previously, considerations over ammonia emissions. Countries where the livestock density may be suitable for centralised digestion may not have as many areas of land suitable for daily spreading.

In order to estimate the reductions which are achievable, an optimistic and pessimistic scenario are examined. It has been assumed in both cases, that small scale digestion is implemented first as the cheapest option; the costs of the Italian plant are taken as typical. In the optimistic case, small scale digestion is implemented in all farms where manure is currently stored as slurry or in pits, and which are over half the size required to supply a small scale digester. In the pessimistic case, small scale digestion is implemented in all farms which are over the size required to supply a small scale digester, and where manure is currently stored as slurry or in pits. It is difficult to estimate accurately the potential for daily spread of the remaining manure, but in order to give an indication of the magnitude of savings which might be achieved, it is assumed that in the optimistic case, 50% of the remaining manure is spread daily, and in the pessimistic case, 25%. Implementation of both measures is assumed to rise linearly between 2000 and 2010. The current mix of management techniques is assumed not to vary with size of farm.

The reductions which these measures achieve in 2010 (by which time the measures are assumed to be fully implemented) are shown in Table 4.9 for the EU and Table 4.10 on a country by country basis. Reductions in 2010 are estimated to be 547 to 820 kt per year; equivalent to 26 to 39% of projected emissions from animal wastes in 2010.

Table 4.8 Factors Affecting Implementation of Measures

Country	% of animals in farms of minimum size or above			High livestock density i.e. suitable for centralised AD			Current practice
	Dairy	Other cattle	Pigs	Dairy	Other cattle	Pigs	
Austria	0%	4%	12%	yes	yes	no	Mainly farm scale- about 50 operating digesters
Belgium	33%	27%	85%	yes	yes	yes	
Denmark	53%	89%	87%	yes	yes	yes	Mainly centralised, often supply district heating schemes. Programme since 1987 encourage development, 20 centralised plants. Farm scale also now receiving encouragement (17 plants)
Finland	1%	4%	20%	no	no	no	6631 pig farms/1.3 M pigs. No problems with excess manure- mostly composted. Some experimental farm scale plant. May be high transport costs for centralised plants, as farms dispersed
France	25%	61%	83%	no	yes	no	About 15 operating plant
Germany	32%	57%	58%	yes	yes	no	Encouraging small scale plant: 380 plants installed
Greece	15%	24%	71%	no	no	no	
Ireland	38%	42%	95%	yes	yes	no	
Italy	42%	54%	82%	no	no	no	Three regions where more manure produced than can be spread. 5 centralised plant and 150 farm scale plant built or under construction. Premium price for electricity produced.
Lux.	22%	90%	54%	yes	yes	no	
Nls	61%	96%	89%	yes	yes	yes	High concentration of pig farms in S and E. All centralised biogas plant now closed for financial reasons. 30 farm scale plant built, but most no longer in operation.
Portugal	11%	0%	52%	no	no	no	Some regions with high concentrations of pig farms. 4 centralised plant and 90 farm scale plant
Spain	18%	48%	70%	no	no	no	
Sweden	25%	42%	62%	no	yes	no	6 farm scale plant
United Kingdom	84%	79%	90%	no	yes	no	40 farm scale plant, but only 20 operational. 18 applications for centralised plant
EU	38%	57%	73%				

Table 4.9 Estimate of Achievable Reductions in EU from Measures in 2010 (kt of CH_4 per year)

	Cool climate			Temperate climate			Total
	Pigs	Dairy	Non-dairy	Pigs	Dairy	Non-dairy	
Optimistic scenario							
AD	104	32	76	239	26	64	**541**
Daily spread	36	47	54	57	42	43	**279**
Total							**820**
Pessimistic scenario							
AD	62	13	38	167	13	32	**325**
Daily spread	39	33	46	49	25	31	**222**
Total							**547**

4.4.5 Cost-Effectiveness Curves

The data on the cost-effectiveness and applicability of the measures is combined to produce cost-effectiveness curves for both scenarios for 2010 and 2020 (Figure 4.3 to 4.6). Measures over a cost of 3500 ECU/t of CH_4 are not shown. A key for the measures in the graphs is given in Table 4.11

Table 4.11 Cost and Scope of Measures to Reduce Emissions from Animal Wastes

Key	Measure	Cost	2010		2020	
		ECU/ tonne	Opt. scenario	Pess. scenario	Opt. scenario	Pess. scenario
1	AD - Temperate - Pigs	30	239	167	243	170
2	AD - Temperate - Dairy	56	26	13	23	12
3	AD - Temperate - Non-dairy	74	64	32	71	36
4	AD - Cool - Pigs	181	104	62	89	54
5	AD - Cool - Dairy	334	32	13	28	11
6	AD - Cool - Non-dairy	446	76	38	77	39
7	Daily spread - Temperate - Pigs	645	57	49	57	50
8	Daily spread - Temperate - Dairy	1183	42	25	40	23
9	Daily spread - Temperate - Non-dairy	1579	43	31	47	34
10	Daily spread - Cool - Pigs	2264	36	39	30	32
	Not included in graph					
	Daily spread - Cool - Dairy	4124	47	33	42	30
	Daily spread - Cool - Non-dairy	5505	54	46	56	47

Figure 4.3 Cost-Effectiveness of Measures to Reduce Emissions from Animal Wastes 2010 - Optimistic Scenario

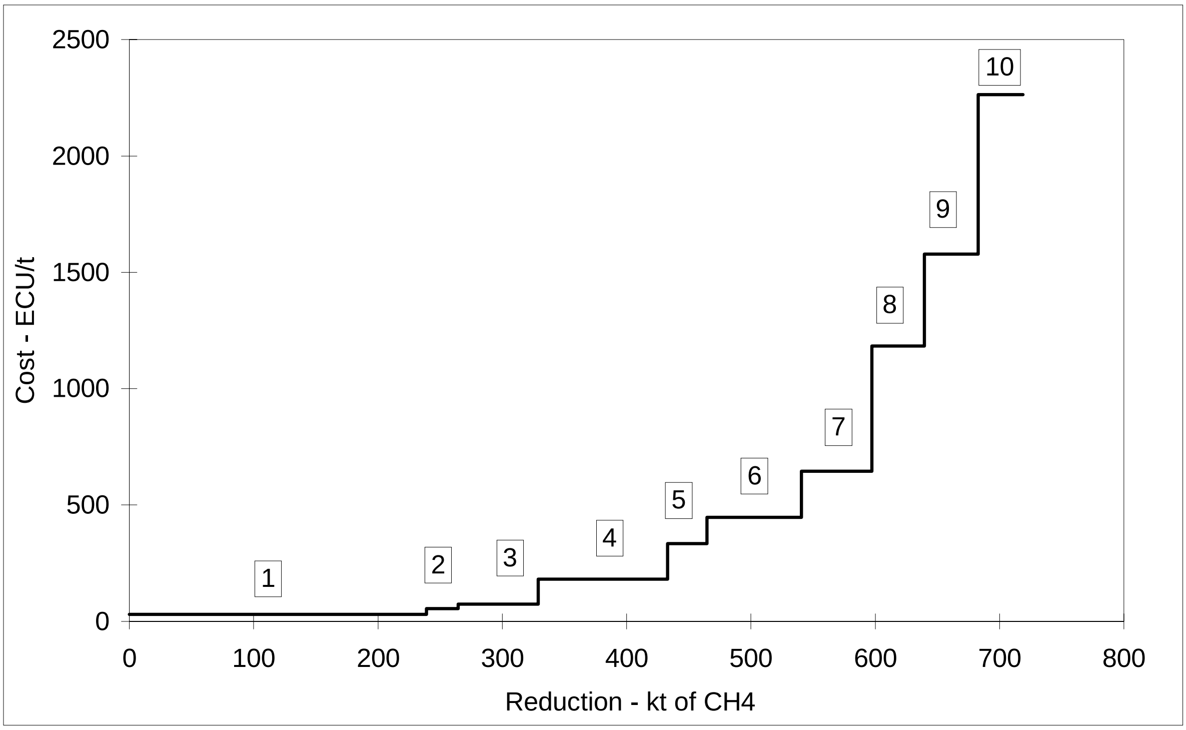


Figure 4.4 Cost-Effectiveness of Measures to Reduce Emissions from Animal Wastes 2010 - Pessimistic Scenario

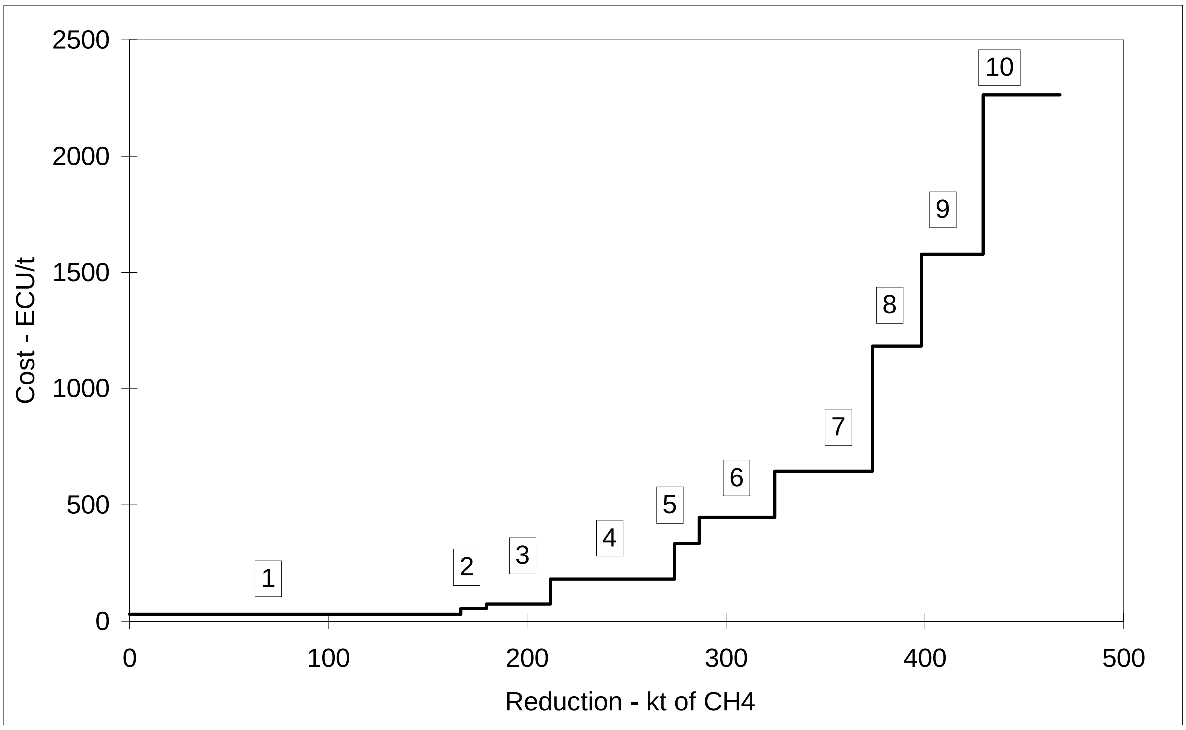


Figure 4.5 Cost-Effectiveness of Measures to Reduce Emissions from Animal Wastes 2020 - Optimistic Scenario

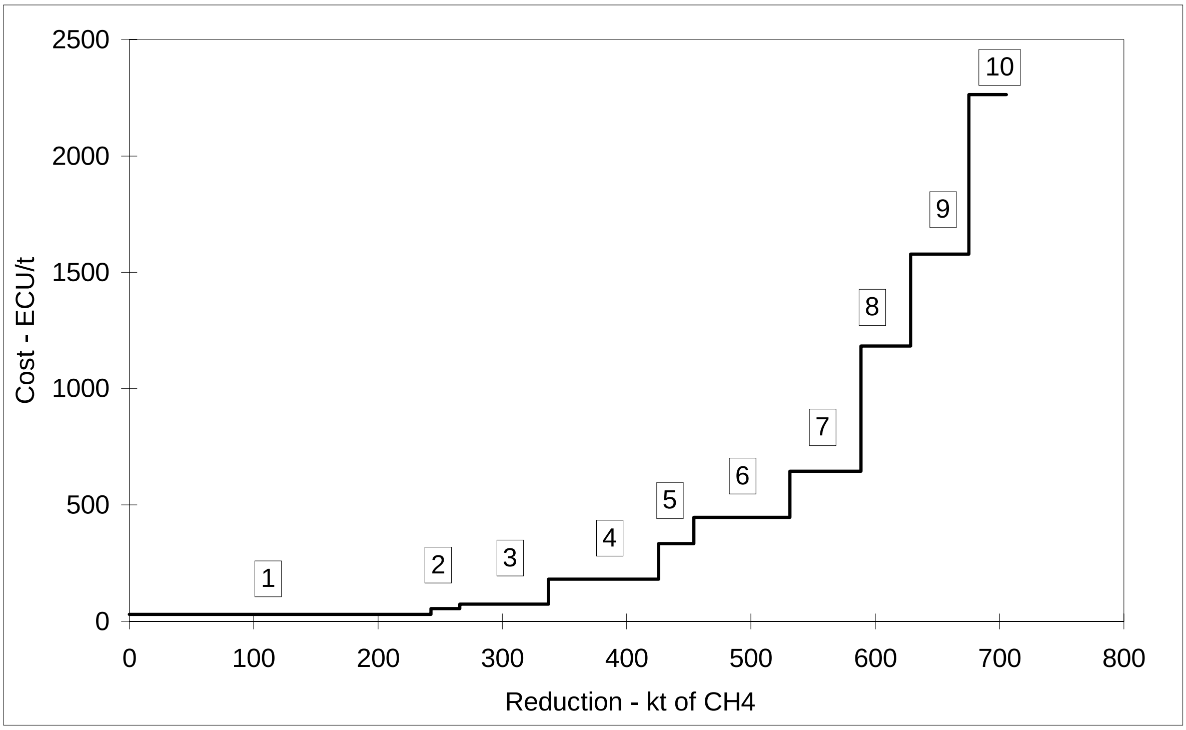


Figure 4.6 Cost-Effectiveness of Measures to Reduce Emissions from Animal Wastes 2020 - Pessimistic Scenario

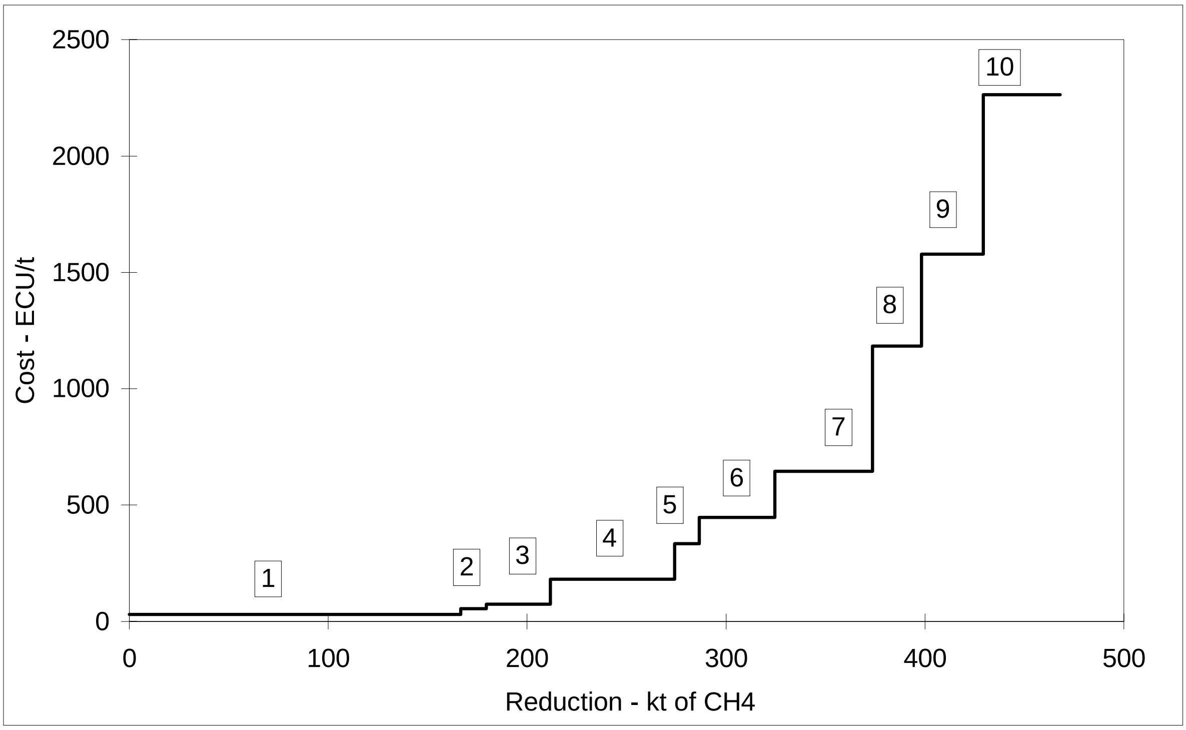


Table 4.10 Estimate of Achievable Reductions in Emissions from Animal Wastes (kt of CH_4/year)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
OPTIMISTIC SCENARIO - ANAEROBIC DIGESTION																
2005	0	7	13	0	21	28	4	7	46	0	16	5	108	1	17	**275**
2010	1	13	25	0	43	50	8	13	95	1	32	9	216	2	32	**541**
2020	1	12	21	0	43	39	8	13	103	1	33	9	216	2	28	**531**
OPTIMISTIC SCENARIO - DAILY SPREADING																
2005	6	6	3	2	14	27	3	7	17	0	3	9	43	1	3	**142**
2010	12	13	5	3	27	48	5	14	34	0	6	17	85	3	7	**279**
2020	12	16	4	3	26	39	5	15	36	0	5	17	85	3	6	**273**
OPTIMISTIC SCENARIO - TOTAL																
2005	6	13	16	2	35	55	7	14	62	1	19	13	151	3	20	**417**
2010	13	26	29	3	70	98	13	28	129	1	38	27	302	5	38	**820**
2020	13	28	25	3	70	78	14	28	139	2	38	27	302	5	33	**804**
PESSIMISTIC SCENARIO - ANAEROBIC DIGESTION																
2005	0	4	8	0	11	13	3	4	31	0	9	3	67	0	10	**164**
2010	0	7	15	0	22	23	6	8	65	0	18	7	134	1	19	**325**
2020	0	6	13	0	23	19	6	7	70	1	18	7	134	1	17	**320**
PESSIMISTIC SCENARIO - DAILY SPREADING																
2005	3	4	4	1	12	21	2	5	12	0	5	5	33	1	5	**114**
2010	6	9	7	2	24	37	3	10	26	0	10	9	66	2	9	**222**
2020	6	11	6	2	23	30	3	10	27	1	10	9	66	2	8	**216**
PESSIMISTIC SCENARIO - TOTAL																
2005	3	8	12	1	23	34	5	9	44	0	14	8	100	2	15	**278**
2010	6	16	22	2	46	61	9	18	90	1	28	16	200	3	29	**547**
2020	6	17	19	2	46	48	9	18	97	1	28	16	200	3	25	**536**

4.4.6 Projection of Emissions under a With-Measures Scenario

Emissions for the 'business as usual' scenario and for the 'with measures' scenarios (assuming reductions shown in Table 4.10) are shown in Figure 4.7 for the EU and on a country by country basis in Table 4.12. Emissions in the 'optimistic with measures scenario' are 58% of 1990 levels by 2010 and 55% of 1990 levels by 2020; in the pessimistic scenario, emissions are 70% and 67% of 1990 levels in 2010 and 2020 respectively.

Figure 4.7 Projection of Emissions from Animal Wastes in the EU under Business-as-Usual and 'With Measures' Scenarios.

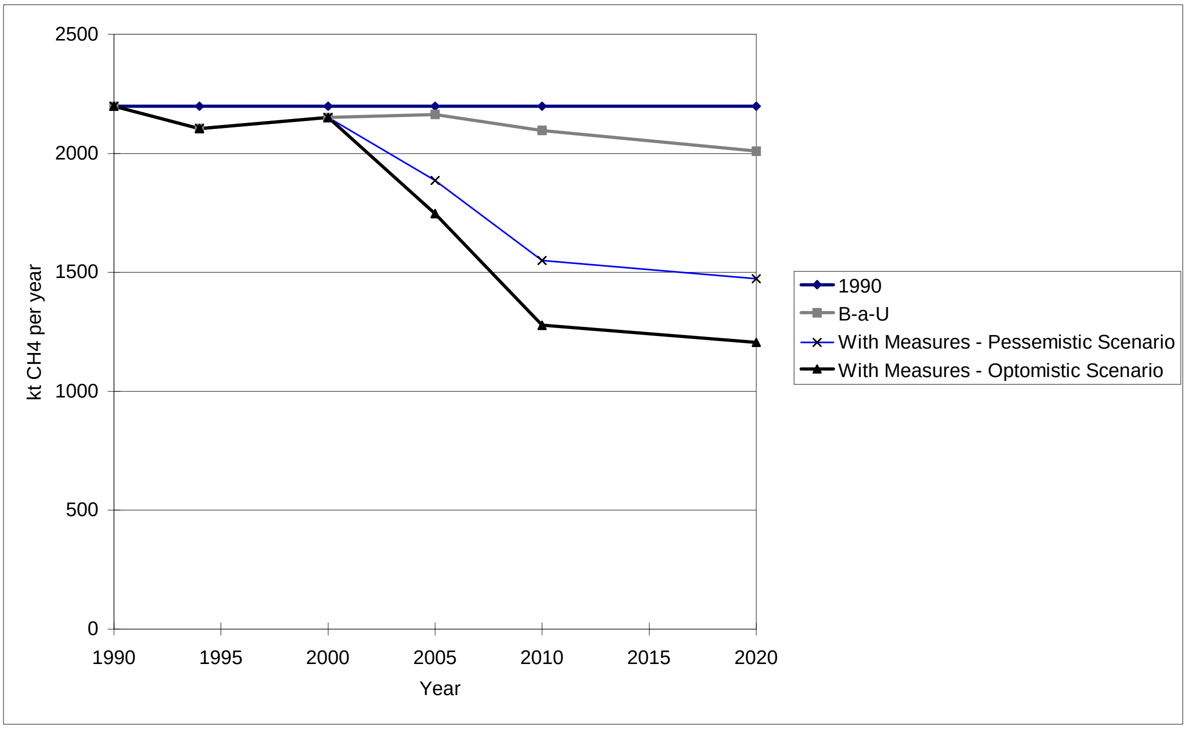


Table 4.12 Projections of Emissions from Animal Wastes under 'With Measures Implemented' Scenarios (kt of CH_4/year)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15	Change from 1990
OPTIMISTIC SCENARIO																	
1990	27	176	162	11	168	614	23	52	192	2	103	68	465	12	125	**2199**	-
1994	28	186	182	10	160	521	24	55	178	2	102	61	486	12	122	**2130**	-3%
2000	29	185	188	9	164	481	25	60	185	2	101	54	550	12	105	**2150**	-2%
2005	23	174	175	7	138	420	20	51	133	2	71	40	399	9	84	**1746**	-21%
2010	16	164	146	5	103	325	13	37	72	2	51	27	248	7	60	**1277**	-42%
2020	16	182	124	4	102	261	13	38	77	2	51	27	248	7	54	**1205**	-45%
PESSIMISTIC SCENARIO																	
1990	27	176	162	11	168	614	23	52	192	2	103	68	465	12	125	**2199**	-
1994	28	186	182	10	160	521	24	55	178	2	102	61	486	12	122	**2130**	-3%
2000	29	185	188	9	164	481	25	60	185	2	101	54	550	12	105	**2150**	-2%
2005	26	178	179	8	151	441	22	56	152	2	77	45	450	10	89	**1885**	-14%
2010	23	173	153	6	127	363	17	48	111	2	62	37	349	9	70	**1550**	-30%
2020	23	193	130	6	126	291	17	48	118	3	61	37	349	9	62	**1473**	-33%

5. Options to Reduce Emissions from Waste

5.1 METHANE EMISSIONS FROM LANDFILLS

5.1.1 Landfill gas generation and composition

Waste comprises a mixed mass of material whose composition varies according to source and with time. When deposited in a landfill (and often during transport to the landfill), a proportion of the organic waste fraction will begin to degrade through biological and chemical action. Degradation results in biochemical breakdown products and the liberation of landfill gas which is a mixture of methane and carbon dioxide. Waste components that contain significant biodegradable fractions are:

- food waste;
- waste from animals;
- garden waste
- paper and cardboard.

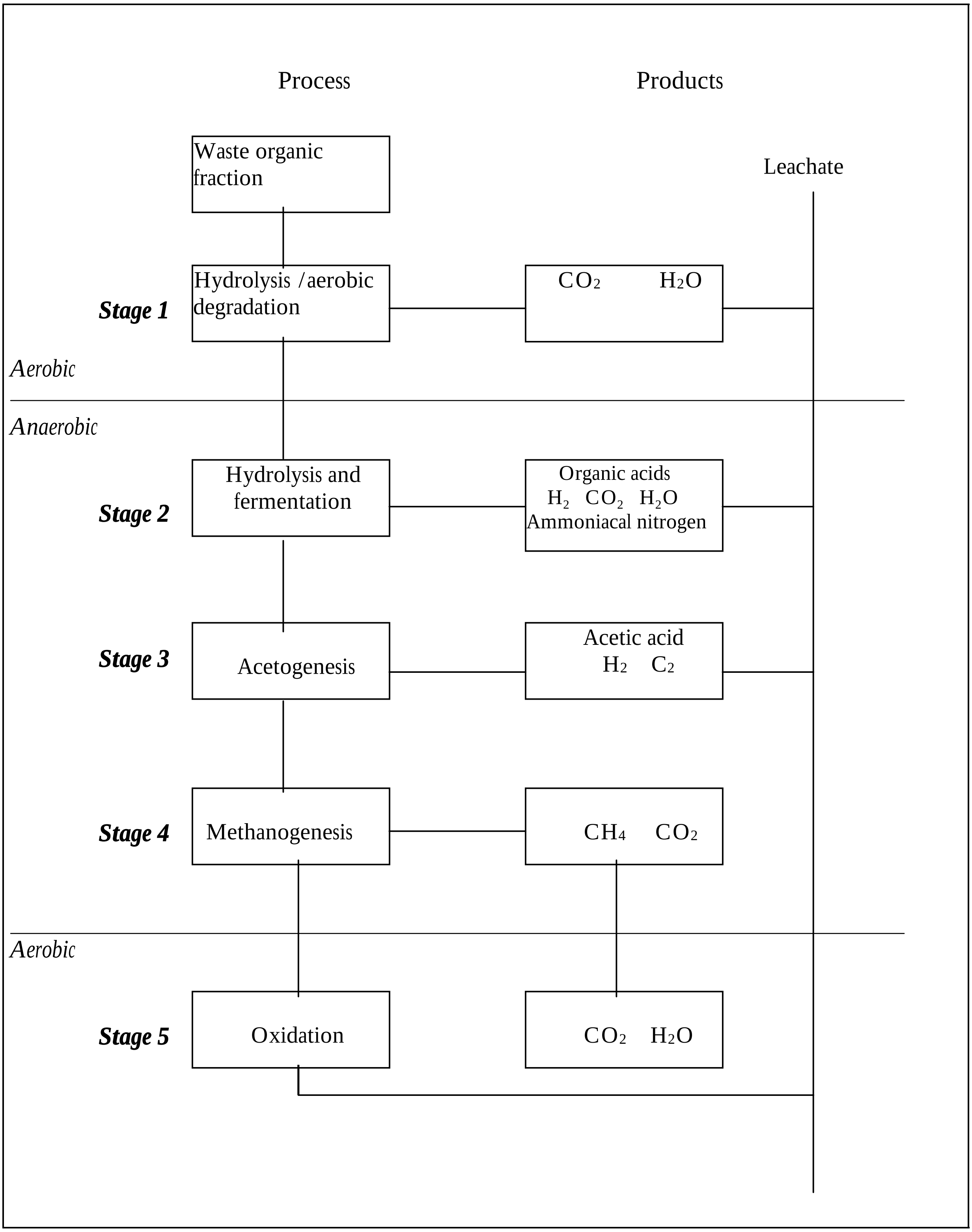
Degradation of such wastes is generally represented as a five stage process which is illustrated in Figure 5.1. It is important to note that methane is only produced under anaerobic conditions, i.e. in the absence of oxygen or air.

Waste degradation is a dynamic and ever changing process, and within the landfill site or open dump, all the stages of degradation may take place at different rates at any one time. This is the result of different wastes having different fractions of biodegradable material and the spatial variability in the physical and chemical environment of the waste materials.

The major factors affecting the production of landfill gas are:

- waste composition;
- moisture content of the waste;
- presence / absence of growth promoting and growth inhibiting compounds;
- temperature;
- pH;
- oxygen content;
- depth of waste in the landfill site;
- age of the landfill;
- presence and operation of a landfill gas collection system;
- site management practices.

Figure 5.1 Major Stages of Waste Degradation in Landfill



Source: Department of Environment, 1995.

These factors can also influence the composition of landfill gas. Landfills may take between 80 and 500 days to reach reasonably steady state conditions and then approximately equal amounts by volume of methane and carbon dioxide are produced (Aitchison *et al.*, 1996). The methanogenic phase can last 10 to 20 years with a slow decrease in landfill gas production during this time as the more degradable materials are broken down (Gendebien *et al.*, 1992). Typical concentrations of the major components of landfill gas are given in Table 5.1, although the concentrations of these gases vary greatly between landfills.

Table 5.1 Typical Concentrations of Major Components of Landfill Gas

Components	Typical Concentration (% by volume)	Maximum Concentration (% by volume)
Methane	40-60	33-88
Carbon dioxide	60-40	35-89
Oxygen	0.16	21
Nitrogen	2.4-5	87

A range of minor components are also present in landfill gas, generally in trace quantities. They are formed by the wide range of microbial, chemical and physical processes that occur within the wastes. There are typically between 50 and 200 minor components present in landfill gas (Gendebien *et al.*, 1992; Rudd, 1995), the majority of which are organic. In general, all minor components combined make up less than 0.15% by volume of a typical landfill gas.

Paradoxically, the measures introduced to improve the standards of landfilling such as liner systems, compaction of the wastes and capping have increased the generation of methane in engineered landfill sites. By engineering barriers to inhibit leachate and gas movement away from the site, air is also prevented from entering the waste mass creating the anaerobic conditions required for the generation of methane. Compacting the wastes reduces the amount of air trapped within the waste mass, whilst the presence of a cap prevents the diffusion of air into the wastes, helping to maintain anaerobic conditions within the landfill.

5.1.2 Factors affecting methane emissions from landfills

Once landfill gas production has started, the fate of the methane it contains depends on the type of waste disposal site. In the case of open dumps or uncontrolled landfill sites the methane may:

- be emitted directly to the atmosphere;
- be released through lateral migration from the landfill site;
- be oxidised, usually to carbon dioxide, as it passes through aerobic regions near the surface of the waste;
- be stored within the waste mass.

In the case of engineered landfill sites, methane may:

- be emitted from the landfill, through pipes and vents to the atmosphere;

- be oxidised, usually to carbon dioxide, as it passes through aerobic regions near the surface of the waste;
- be stored within the waste mass;
- be recovered and disposed of by burning in flares or energy recovery schemes.

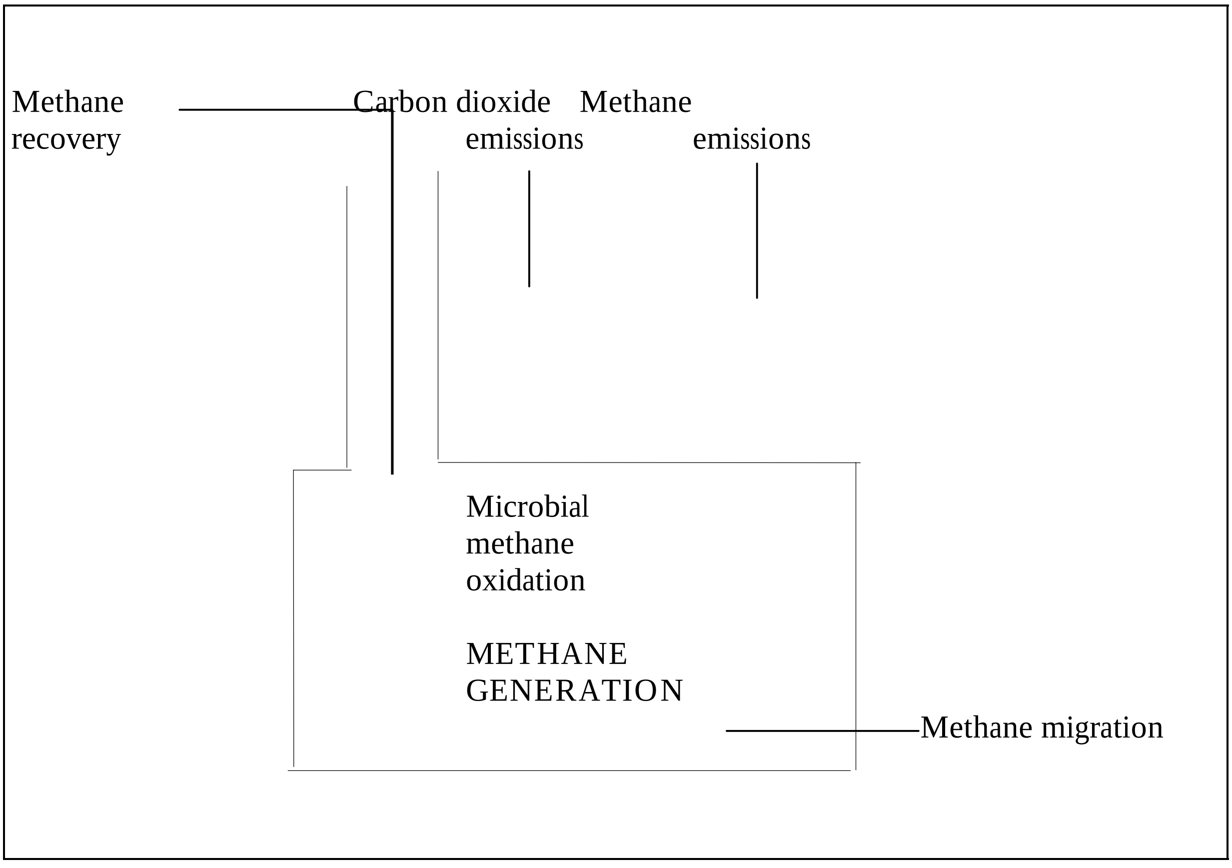
Bogner and Spokas (1993) described the effects of all the different factors using a simple model as follows:

Methane generated = Σ [methane emitted + methane recovered + methane oxidised + methane migrated] + methane stored

It follows that the rate of methane emitted from a landfill depends on the rate at which methane is produced and the overall effect of different factors reducing methane emissions to the atmosphere (Figure 5.2). We have a poor understanding of several components of the methane balance. This is because landfills are, in essence, poorly mixed, water limited anaerobic digesters, containing high solids wastes of mixed composition and biodegradability (Meadows, 1995). The factors affecting methane emissions are difficult to quantify and vary from site to site.

Atmospheric conditions can also affect methane emissions from landfills. Rapid decreases in barometric pressure are often associated with increased emissions of landfill gas and hence methane from landfills. Temperature fluctuations are also known to affect the rate of methane emissions from landfill sites.

Figure 5.2 Factors affecting Landfill Methane Emissions



5.1.3 Gas control

Uncontrolled landfill gas migration from a site poses a threat not only to the global environment but also to human health and can pollute the local environment. There are two main risks: firstly from explosion or fire caused by build up of landfill methane in buildings near landfill sites, and secondly from asphyxiation, as landfill gas, which is heavier than air, may collect in sewers and manholes. In a well engineered landfill site, gas is not allowed to escape in an uncontrolled manner, it will be contained, collected and preferably treated before being destroyed by flaring or used in an energy recovery project.

Gas containment can be achieved through the use of impermeable barrier or liner systems which prevent the gas from migrating laterally away from the landfill site. The use of a capping layer to prevent water ingress also helps to prevent the vertical migration of landfill gas into the atmosphere.

Since the gas cannot escape, measures are then introduced to collect the gas from within the waste mass and transfer it to facilities for destruction or use in energy recovery. Generally the gas is drawn from a number of vertical gas collection wells drilled through the depth of the waste mass or from horizontal perforated pipes laid in gas permeable trenches distributed throughout the wastes. These wells or trenches are then connected to a gas ring main and under an applied vacuum the gas is piped to a central blower/fan or gas compressor unit. Further stages including condensate traps, particulate filters, chillers or gas pressure boosters may also be included in the overall collection system.

Once the gas is collected at a central point, it is then possible to either destroy the gas in a flare or use it for energy recovery either through direct use of the gas or through electricity production.

5.1.4 Landfill site management

Another important factor in determining how much methane is emitted from a landfill site is the operating practices and technologies employed at that site. There are a wide range of options for waste in dumps and landfills and these are listed below in ascending order of organisation and technology:

- uncontrolled dumping at or close to source;
- removal from source followed by uncontrolled dumping;
- uncontrolled dumping of wastes from many sources in a single area;
- low standard, uncontrolled landfilling in small tips close to the source of the waste;
- low standard, uncontrolled landfilling at larger, centralised tips;
- controlled landfilling;
- controlled engineered landfilling;
- best practice engineered landfilling.

In uncontrolled dumping, waste is tipped over an unmanaged area with no restriction of access. The waste is tipped at random or in the nearest convenient place and there is no system of waste placement. The waste is uncompacted except by the weight of additional layers of waste. There are no environmental controls or monitoring and no records of waste input. World-wide, uncontrolled dumping is the most frequent form of waste disposal and is generally found to be in developing countries, although uncontrolled dumping may still be found in some areas of developed countries.

In low standard landfilling, whether uncontrolled or controlled, waste is tipped within defined boundaries and there may be some control of access. The waste might be placed in defined areas within the site in an organised system and it may be mechanically compacted. However, there are few if any environmental controls or barriers to restrict water ingress into the site and leachate and gas movement away from the site and there is no engineered final cap or cover. Little or no environmental monitoring is undertaken and few records of waste input are kept.

In engineered landfills, there are defined site boundaries and disposal areas. There is secure controlled access and good records are kept of waste inputs. These sites have effective environmental controls such as a low permeability barrier or liner system surrounding the wastes that inhibit landfill gas and leachate movement away from the site. Waste is placed according to a specific plan to fill the site and is usually compacted to reduce the void space it occupies and the site has an engineered low permeability cap and cover. Landfill gas and leachate are monitored and controlled.

Best practice sites incorporate all the standards of an engineered site but are designed, constructed, operated, monitored, completed and restored to the highest standards following the latest best practice. It should be noted that best practice will differ from country to country, depending on local waste management policy, local climate and geology.

While all EU Member States should be observing high standards of landfilling, considerable diversity of landfilling practices still exists. Thus many very small sites, especially those serving rural communities, still exist and standards are sometimes very basic. The geometry of many landfills (mainly shallow sites) does not always lend itself to establishing conditions which are optimal for methane-rich gas formation and the wide variability in climatic conditions means that landfilled waste degradation behaviour and rates are quite variable.

5.2 MITIGATION OPTIONS

There are two generic approaches to limiting and controlling emissions of methane from landfills to the atmosphere. The first approach seeks to reduce the mass of methane generated within the landfill itself by either pre-treating wastes to reduce the degradable content or by reducing the overall mass of waste landfilled. The second approach considers ways in which methane emissions from landfill sites can be reduced. This can include improving site engineering standards to cut down methane migration, fugitive emissions and increase methane oxidation activity or installing a collection system to increase the mass of methane which is collected and can be destroyed by flaring or used for energy recovery.

Both approaches can be used independently or in combination to reduce methane emissions and the proposed draft Council Directive on the landfill of waste embraces both approaches (OJ No C 156, 97), and includes requirements to reduce the landfilling of organic waste, and to introduce landfill gas control.

5.2.1 Options to reduce landfilling of organic waste

The extent to which landfills are currently deployed within the overall waste management strategy or plan of a country or region is in part linked to the available waste management resources and in part to national or regional policy decisions, laws and regulations. Many EU Member States still currently rely on landfill as the major route for treatment and disposal of solid wastes. Table 5.2 illustrates the variation in the proportion of wastes landfilled in selected countries within the EU (Coopers & Lybrand, 1996).

Table 5.2 Disposal Routes for Waste in the EU

Country	Landfill	Incineration	Composting	Recycling	Other
Belgium	39	52	2	7	-
Denmark	18	58	1	22	1
France	45	45	6	4	-
Germany	42	25	10	22	1
Greece	94	-	-	6	-
Ireland	99	-	-	1	-
Italy	85	7	-	4	4
Luxembourg	24	47	1	28	-
Netherlands	40	28	15	17	-
Portugal	35	-	10	1	54
Spain	83	6	10	1	-
UK	85	10	-	5	-
Total	58	22	5	10	5

The nature of the waste deposited at landfill sites also varies significantly over the EU as shown in Table 5.3. As already discussed, this has important implications for the subsequent degradation processes and gas production potential, and to a lesser extent waste composition also determines the degree of sophistication required to achieve appropriate environmental protection. Compositional analyses of waste can often be closely related to the sophistication of the site design and the engineering practices adopted.

Table 5.3 Estimated Municipal Solid Waste Composition in the EU

Country	Organics	Paper/ board	Glass	Metals	Plastics	Textiles	Other
Belgium	43	28	9	4	7	9	
Denmark	37	30	6	3	7	18	
Germany	32	24	8	5	9		22
France	21	27	7	4	11	2	28
Greece	49	20	5	4	9	13	
Ireland	42	15	6	4	11	8	14
Italy	32	27	8	4	7	3	19
Luxembourg	41	16	4	3	8	3	25
Netherlands	39	25	8	5	8	15	
Portugal	39	20	4	2	9	5	21
Spain	44	21	7	4	11	5	8
UK	20	33	9	8	6	4	20

Source: Coopers and Lybrand, 1996.

Specific targets for reducing the total amount of biodegradable municipal waste going to landfills in the future are set in the proposal for a directive on the landfill of waste. The Environment Council recently (March 1998) reached a political agreement in view of a common position on the proposal, which set the following targets (assuming that the Directive comes into force in 1999):

- by 2006, biodegradable municipal waste going to landfills must, as far as is possible, be reduced to 75% of the total amount (by weight) of biodegradable municipal waste produced in 1995.
- by 2009, biodegradable municipal waste going to landfills must be reduced to 50% of the total amount (by weight) of biodegradable municipal waste produced in 1995.
- by 2016, biodegradable municipal waste going to landfills must be reduced to 35% of the total amount (by weight) of biodegradable municipal waste produced in 1995.

Countries which have a high reliance on landfill (i.e. more than 80% of waste is currently landfilled) may be given an additional 4 years to comply.

Options for meeting (or even exceeding) these targets include waste reduction, re-use and recovery of organic wastes, and the pre-treatment of wastes to reduce the degradable organic fraction of waste going to landfill, including:

- composting
- anaerobic digestion
- incineration
- recycling (paper, textiles)

All four options are already used to varying degrees throughout the EU. Both composting and anaerobic digestion are used in combination with various waste collection and sorting techniques as a means of reducing unwanted waste components. Incineration is able to deal with a wider range of wastes and as such requires less sorting of wastes prior to the combustion process.

It should be noted that even if waste reduction, re-use and recovery becomes the norm, there will always be waste fractions that will require some final disposal either because a suitable recovery route has not been found, or because the waste is contaminated in some way.

5.2.1.1 Waste reduction reuse and recovery

The European Commission already has a preferred hierarchy for the management of wastes, reflecting a broad indication of their relative environmental benefits and disbenefits:

- Reduce
- Re-use
- Recycle or compost
- Recover
- Dispose

The first priority is to reduce the production of waste to the minimum wherever possible. This includes both the waste that is discarded from production processes or 'process waste' and the waste that is discarded by the consumer as post-consumer or 'product waste'. The second priority is to re-use, that is to put so-called waste back into use so that it does not enter the waste stream. There are two types of re-use. The first is where products are designed to be used a number of times before becoming obsolete, such as the delivery of milk in re-useable bottles or the retreading of tyres. The second form of re-use occurs where new uses are found for items once they have served their original purpose such as discarded tyres used as boat fenders. The third priority is the recovery of materials either through recycling, composting or anaerobic digestion of waste streams. Recycling involves processing waste to produce a useable raw material or product. Recycled material can, in principle, be re-used many times. The fourth priority is for the recovery of energy from waste by stabilising through incineration or anaerobic digestion. The final priority is safe disposal. This hierarchy provides an important policy framework for Member States within which waste management decisions can be taken.

Specific legislation to reduce waste is increasingly being introduced throughout the EU and Table 5.4 summarises current legislation and agreements (Coopers & Lybrand, 1996).

Table 5.4 Legislation and waste reduction targets in the EU

Country	Key legislation and Agreements	Waste Reduction Targets
Belgium	Voluntary agreements at regional level	Flanders - 12% reduction by 2000 Brussels - zero growth
Denmark	Beverage packaging decree (1989) Waste Plan (1993-1997)	no specific targets
France	Household packaging waste decree (1992)	no specific targets but waste reduction seen as priority
Germany	Packaging waste decree (1991)	no specific target but waste reduction seen as priority
Greece	no specific legislation	none
Ireland	no specific legislation	none
Italy	various laws and decrees on particular types of materials	no specific targets
Luxembourg	no specific legislation	none
Netherlands	Voluntary agreement (1991) Framework Law (1994)	standstill at 1990 levels for certain types of waste only
Portugal	no specific legislation	none
Spain	voluntary agreement under negotiation	no specific targets as yet
UK	no specific legislation (although many companies have taken action voluntarily)	none

5.2.1.2 Paper Recycling

The main recycling option which will help to reduce methane emissions from landfill is the recycling of waste paper in the manufacture of paper and board products. The use of waste paper in paper and board manufacture is already an established manufacturing practice throughout the European Union, and the percentage of waste, or recovered paper currently used in the manufacture of new paper products in Member States shown in Table 5.5.

Table 5.5 EU Waste Paper Utilisation Rates (1997)

Member State	Waste Paper Utilisation	Utilisation Rate
	'000 tonnes	%
Austria	1,642.2	43.0
Belgium	448.0	30.1
Denmark	410.0	122.4
Finland	608.8	5.0
France	4,466.7	48.9
Germany	9,414.0	59.1
Greece	275.0	79.3
Ireland	54.0	124.4
Italy	3,657.5	48.6
Netherlands	2,301.0	73.0
Portugal	322.0	29.8
Spain	3,032.0	76.4
Sweden	1,652.0	18.9
United Kingdom	4,618.1	71.5
Total EU	**32,901.5**	**43.7**

Source: CEPI, 1997

The assessment of the cost-effectiveness and reduction potential of this option is more complex than for many other processes, since it involves the manufacture of a high value non-energy commodity. Waste paper is regarded by the industry as a primary raw material and its use is determined on the basis of economic cost, or more specifically the cost of pulp produced from waste paper compared to virgin pulp. A number of factors can be identified as affecting the actual utilisation rate which is achieved by the industry and the scope for increasing the use of waste paper and diverting flows away from landfill. These factors, which are discussed in more detail below, are:

- Market acceptability
- Availability of waste paper supplies (as compared to virgin pulp costs)
- Investment and operating costs
- Waste disposal

The situation is further complicated by the fact that European paper manufacturers operate within a global market place and that the market has shown considerable volatility in recent years.

Market Acceptability

Waste paper cannot be used universally by the paper industry. Constraints of quality limit its use in the manufacture of certain products, most notably the fine papers and graphics grades. Similarly some packaging grades require strength characteristics which can only be provided by virgin kraft fibre. These constraints are imposed by the market place and the limits of customer specification, and must be considered to be beyond the influence of the paper producers. For

other grades, such as newsprint, virgin pulp and waste paper feedstocks can be freely substituted. In such cases observed utilisation rates will be constrained by other factors.
Table 5.6 illustrates differences in utilisation rates between grades for the UK, and provides a qualitative assessment of the scope for increased utilisation. Cultural differences affecting the acceptability of waste paper products in the market place are apparent between Member States.

Table 5.6 Summary of Market Constraints for UK Paper Grades

Grade	Utilisation Rate*	Constrained by Specification	Comments
Newsprint	89	no	Further substitution may possible - limited input of virgin fibre required, which can be met by imports
Graphics	14	yes	Considerable substitution possible - cost constrained
Corrugated Case Materials	>100	no	
Wrappings <150 gsm	72	yes	Further substitution may be possible
Folding Boxboards	79	yes	Further substitution may be possible
Sanitary	93	yes	Approaching saturation
All Others	74	yes	Highly product specific

* Utilisation rates may be higher than 100% due to yield losses.

Availability of Waste Paper Supplies

The cost of waste paper will be a strong determinant in investment decisions over recycling capacity. It is established through a complex interaction of supply and demand:

- ***Demand.*** A wide range of grades of wastepaper are available for recycling, from lower quality contaminated waste, through medium quality container waste to the highest quality unprinted woodfree papers. Certain paper products require specific grades of wastepaper inputs in order to meet specifications and to maintain quality standards. Thus availability of a certain grade of wastepaper will not guarantee that a market exists for it in the short term, although in the longer term it may encourage investment in appropriate production capacity. Similarly a high demand for a specific grade will deter future investment in recycling capacity based on that material.

- ***Supply.*** The constraints described apply to that proportion of waste paper which is recovered and made available for recycling. In practice this will be a limited proportion of the waste paper generated in an economy. The actual supply of the various grades of waste paper is a function of the consumption of new paper products and the operation of the recovery infrastructure. While large volumes of a product may be consumed and hence are theoretically available for consumption, realisation of this potential depends upon the capability of the recovery infrastructure to deliver material at economic cost to the recycling paper mills. In the UK, for example, the theoretical waste paper potential, measured at the stage of the new paper end user, was 8.65M tonnes in 1994. Of this total, the recovery infrastructure succeeded in delivering only 2.95 M tonnes to the paper recycling process, the remainder being disposed of to landfill, incineration or composting.

Since virgin fibre can, in all cases, be substituted for waste paper, the determining factor in investment decisions over recycling plant is the differential between waste paper prices and virgin pulp. For the production of many grades, recent experience has demonstrated this differential to be notoriously variable. As a consequence cashflows can prove difficult to predict and risks associated with marginal investment in recycling plant are high.

This effect is exaggerated by simultaneous movements in both virgin pulp and waste paper prices. Pulp prices tend to follow a cyclical pattern, whereas waste paper prices tend to be more random, being affected by a range of various factors. During the 1990s major swings in waste paper prices in Europe have been attributed to the unilateral introduction of packaging waste schemes and major investment in waste paper-based capacity in South East Asia.

Investment and Operating Costs

The paper industry comprises of a number of discreet manufacturing sectors, each producing paper grades to meet the requirements of a specific market. There is potential to use waste paper in the manufacture of most commodity grades. However, the type of waste paper used as feedstock, and the plant required to recycle it, will vary enormously between sectors.

In general the manufacture of lower grade paper products such as corrugated case will tolerate a low quality feedstock, such as container waste. Similarly the recycling plant required to produce papermaking stock for low grade paper will involve only a limited number of process stages. Manufacture of fine paper grades paper will, by contrast, require a high quality feedstock comprising predominantly of woodfree (office) waste and will require a complex process comprising of multiple stages of deinking, screening and cleaning to produce a stock of comparable quality to virgin pulp.

As a consequence, the investment costs and operating and maintenance costs associated with the recycling of waste paper are highly dependent upon the grade of paper being produced. Typical capital costs for a 200 tonne/day deinking plant, producing pulp of an equivalent quality to virgin pulp, will fall within the range of £25M to £30M (Meeks, 1998). Operating costs are estimated to fall in the range of £60 - £90 per tonne, depending upon quality requirements and exclusive of waste paper and waste disposal costs (see below). Plant of this nature would produce a deinked pulp appropriate for use in newsprint or fine paper production. The manufacture of lower quality grades, such as corrugated case materials, would require an estimated capital investment of £15M to £20M and incur operating costs of between £35 and £65 per tonne, exclusive of waste paper and waste disposal costs.

Waste Disposal

Manufacture of paper products from a wastepaper feedstock results in the generation of considerable quantities of solid waste. Overall for the UK, deinking (recycling) processes generated approximately 480,000 dry tonnes of solid waste in 1996. Total waste paper consumption for 1996 was 4.3M tonnes (Paper Federation, 1997) and hence the average yield loss for paper recycling is approximately 10%.

In practice the solid waste is usually disposed of as a wet sludge, with a solids content of between 30% and 50%. In some cases these sludges may contain a high organic fraction (>50% of the solids) and may therefore have significant methanogenic potential. The actual methane production will depend upon the choice of final disposal route.

The cost of sludge disposal is an important consideration. The relative cost of disposal routes will determine the pattern of disposal on a national or regional basis. Within the UK, landspread is the primary disposal route (42%), although landfill (36%) and incineration (21%) also receive large volumes. The absolute costs of disposal are becoming increasingly significant in many EU countries, and now represent a major consideration in assessing the economic viability of paper manufacture from waste paper.

Summary

Paper manufacture can be considered as an abatement option for methane produced from the waste paper fraction of MSW. In practice, however, its cost and applicability as an abatement option are complicated by the necessity for waste paper-based products to compete in the market place with virgin fibre-based products. Consequently consumer acceptability can represent a major constraint on the use of waste paper for the manufacture of certain grades. Paper manufacturers presently invest in recycling capacity on the basis of economic cost, with predicted price differentials between virgin pulps and the appropriate waste paper grades being a major factor in the decision process. These future differentials are difficult to estimate and are beyond the scope of this study.

Capital and operating costs can be more readily estimated, although differences in the waste paper feedstock and the grade of paper being produced will lead to major variations. Within Europe the greatest scope for further investment in waste paper recycling is likely to be achieved through a substitution of deinked (recycled) pulp for virgin pulp in the higher quality grades such as newsprint, fine papers and tissue. There is considered to be less potential for further expansion of recycled capacity for the manufacture of lower grade products such as corrugated case materials.

No Member State offers direct subsidies for paper manufacturers utilising wastepaper. In practice, however, the operation of regulatory measures or market based instruments may affect waste paper prices. These in turn may impact upon the economics of investment. The implementation of the Packaging and Packaging Waste Directive (94/62/EC) has demonstrated some such effects.

5.2.1.3 Composting

The degradable organic fraction of the waste stream can be composted to stabilise the organic matter. The residue is then disposed of to landfill, or if the feedstock waste is uncontaminated the composted product can be used as a soil improver. In the latter case, where the composted product is to be marketed as a horticultural/agricultural product, it is essential that it contains low levels of contaminants. This in turn requires that the feedstock waste comes from pre-sorted or source-separated waste streams. For wastes derived from, for example, municipal parks or public gardens this is a feasible option, since nearly all the arisings will be suitable for composting. For domestic waste arisings this can prove to be substantially more expensive than collecting mixed waste streams.

Many communities across Europe are now collecting organic wastes from households for organic waste treatment, although the coverage of schemes varies significantly between countries. 90% of the Dutch population have access to a scheme but Spain, Portugal, Ireland and Greece have zero or minimal coverage. In the UK approximately 1.5% of households have access to a kerbside collection for kitchen and/or garden waste, which collects about 0.25% (39 kt/a) of the waste stream. However, substantially more garden waste is collected via recycling centres which householders bring their waste to for disposal or recycling and this collection route provides 166 kt/a (1% of the waste stream) to composting schemes (Composting Association, 1997). Studies have shown that where kerbside collection of both kitchen and garden waste is performed approximately 220 kg/household of organic material can be generated (Smulders et al, 1996), which represents a potential of about 20% of the waste stream (derived from DETR, 1997).

In the UK, it is estimated that approximately 220 kg per household per year of organic waste is collected by source separation schemes (Smulders A, et al, 1996), which represents approximately 20% of the domestic waste stream (derived from DETR, 1997).

The costs of operating source separation schemes can be high and a Dutch study suggested an increase from 45 ECU per tonne for normal mixed waste collection to between 50 and 75 ECU per tonne for source separated organics collection (Anon, 1997). Unpublished estimates in the UK suggest that collection costs rise from £20-30 (27-40 ECU) per tonne to £25-40 (33-54 ECU) per tonne.

Centralised composting

A wide range of methods are available to compost organic waste centrally, ranging from simple, cheap solutions such as open turned windrow to more complex and expensive large engineered systems, and capacities of plants may vary from 500 to 100,000 tonnes per year. The choice of system depends on many factors, but overall simple systems are appropriate for small volumes of waste in sites that are distant from the public whilst the more complex systems are more successful on large waste arisings and on sites closer to the public. The limitations of the less complex systems are that they require large areas and can emit odours and bioaerosols and hence need to be separated from the public but they are significantly cheaper than the more complex alternatives. All forms of composting are energy intensive requiring sorting and multiple handling of the waste material through the use of mechanised equipment and can be technologically difficult on a large scale. Estimates of the energy use within composting facilities vary between 20 and 70 kWh/t.

Small ***windrow systems*** treat up to 25,000 tonnes per year (although there are a few plants much larger than this) and operate by shredding the waste and placing it in piles 2-3m high. These are then agitated or turned on a regular basis to introduce air and control the temperature of the waste. The agitation can be performed by front end loader or by specialised machines. After composting the compost is matured, screened and sorted to remove contaminants prior to sale. This process can take between 3 and 6 months depending on the nature of the feedstock.

A typical ***tunnel composting*** plant will treat 10-50,000 tonnes per year of mixed compostible waste. The compost remains in the tunnel for approximately 20 days and is turned at least once during this period. Parameters such as temperature, humidity and the O_2 to CO_2 ratio are carefully controlled. After processing, the compost is matured, screened and graded. ***Building or hall*** composting is similar to tunnel composting but the container is a building and the compost is agitated by automatic devices as well as air being blown in to control the process.

Approximately 30-50% of the waste input will be turned into a saleable compost, and about 25% will be reject material. This may be disposed of to landfill, but will have a minimal methane generation potential as it is principally plastics and undegraded woody material. The remainder of the original feedstock is ‘lost’ in the form of moisture which evaporates or as CO_2

emissions. As these CO_2 emissions are 'short cycle' carbon (i.e. the carbon in the organic material was recently sequestered from the atmosphere) they are not considered to contribute to global warming.

Capital and operating costs vary considerably between systems and sites and some examples are given in Table 5.7. This highlights the vast discrepancies between the systems and site specific factors.

Table 5.7 Cost data for composting plants

System	date	scale t/year	capital cost	operating cost	capital cost (1995 ECU/t)	operating cost (1995 ECU/t)
Turned windrow	1995	8 000	£600k	£11-15/t	90	16
Turned windrow	1993	9 000	£250k	£11-15/t	38	18
Tunnel	1992	25 000	9,104k NLG	77 NLG/t	173	37
Tunnel	1992	50 000	17,206k NLG	60 NLG/t	163	28
Hall	1992	25 000	10,252k NLG	82 NLG/t	194	39
Hall	1992	50 000	25,655k NLG	100 NLG/t	243	47

Source: (Composting Association, 1997; de Jong et al, 1993)

Other sources of cost data (Joosen and van Zuylen, 1997) suggest that costs for composting are in the range 40-60 ECU/t. The high operational cost is associated with the degree of handling and turning needed for processing wastes into compost. If the compost is of a high enough quality then it can find markets although the value is low and estimates vary, between 0 to £5 (0-7 ECU) per tonne in the UK and 0-15 ECU/t in Europe (Wheeler et al, 1996; Joosen and van Zuylen, 1997).

Home composting

An alternative to centralised composting is home composting, which can reduce the overall environmental impacts of composting by reducing the need to transport the waste and avoiding mechanical processing. With well operated home composting, it is believed that environmental impacts from methane or leachate production should be minimal. Poorly operated composting may however be a source of methane and leachate due to insufficient turning. The amount of methane released from a poorly operated home compost heap is not known, nor are the relative proportions of poorly and well run home composters under the various levels of encouragement and information provided by local authorities. However, given that most poorly run compost heaps will be small and thus have a high surface to volume ratio, most of the degradation will be aerobic or the methane emissions will be degraded in the outer layers of the heap. Thus methane emission can be expected to be minimal.

5.2.1.4 Anaerobic digestion (biogasification)

Anaerobic digestion (AD) is primarily a method of energy recovery which is based on the natural decomposition of organic material in the absence of oxygen. This is the same process which occurs in a landfill, but in the case of anaerobic digestion, it is optimised so that all the gas is generated in about three weeks rather than three (or more) decades.

The process is carried out in an enclosed vessel, and produces:

- ***Biogas***: a mixture of CH_4 and CO_2, which is burnt to allow energy recovery, either through the generation of electricity or production of heat.
- ***Digestate***: the solid residue that provides a soil improver. It may either be stabilised by a very short (1-2 week) aerobic "composting" stage or is applied directly to land. As with composting, the solid residue produced may be sold if it is of high enough quality.
- ***Liquor***: a liquid residue that contains a large proportion of the nutrients in the waste and can be used as a liquid fertiliser or disposed of via the sewer.

Several anaerobic digestion techniques are used, the main variations being in the:

- feedstock (i.e.) the mixture of waste treated
- operating temperature
- percentage of dry matter (solids content) of the feedstock.

Feedstock
Digesters can operate on just one waste stream such as mixed MSW or source separated organics, or in a co-digestion or centralised mode, where a mixture of wastes from different sources such as agricultural, food processing, sewage and municipal wastes are treated. There are technical and administrative advantages to co-digestion in that plants can be more local (as there is a wider and hence greater supply of waste) and the process is generally more robust. However there is a potential for contamination of clean wastes by more contaminated feedstock such as municipal wastes.

Operating Temperature
There are two optimum temperature regimes, mesophilic operating at 35^0C and thermophilic, at approximately 55^0C. Mesophilic operation requires less heating and is considered more stable. Thermophilic operation provides better sanitisation of the digestate product making it more marketable, and achieves faster but less complete degradation.

Solids Content
The solids contents of the waste determines the internal materials handling arrangements in the digester. Systems can be categorised into:

- low solids or slurry systems
- high solids systems
- two phase systems
- leach bed systems.

Low solids or slurry digestion is the simplest and most widely used and studied system; The solid waste is slurried in a continuously stirred tank reactor (CSTR) at a low solids (< 10%) concentration, with mixing provided either mechanically or by gas recirculation. Similar systems which deal with waste with a medium solids content (of up to 16%) rely on the higher solids concentration to avoid scum formation and only provide mixing to aid gas release and material flow through the digester.

The majority of systems on the market today for digestion of solid wastes are of the low (<10%) or medium (10-16%) solids designs, and they can be used to treat municipal solid wastes, agricultural wastes, and many industrial wastes. Many of the systems when applied to more solid waste e.g. biowaste or market wastes, require the use of a carrier medium of sewage sludge or manure.

High solids systems have been developed over the past 15 years principally for the organic fraction of municipal solid waste, but have been extended to other highly organic industrial, market and agricultural wastes. The digestion occurs at solids contents between 16% to 35%. When the solids concentration is between 25-35%, the free water content is low and these systems are referred to as dry phase digestion or anaerobic composting.

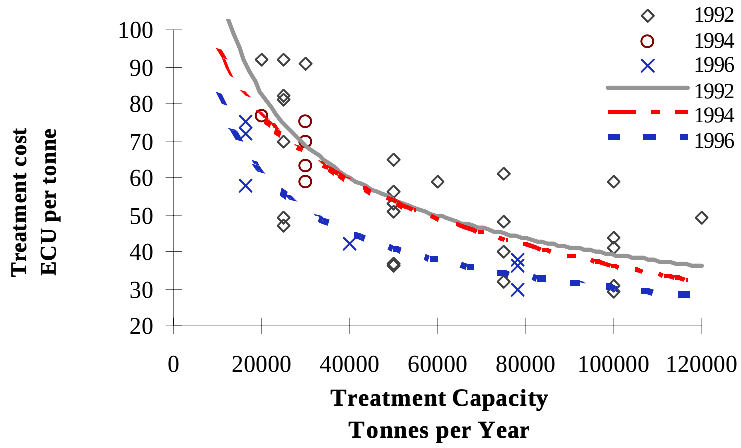
In **two stage digestion** systems the initial stages of the digestion reaction, hydrolysis, acidogenesis and acetogenesis of the waste are carried out separately from the methanogenesis stage. Since each step is optimised separately, enabling each of the digestion reactions to operate closer to its optimum, the rate of stabilisation is significantly increased. However, the system has the disadvantage of requiring two vessels and thus has higher capital costs.

The **batch digestion** design concept is closest to the processes occurring naturally in a landfill. The reactor containing the organic material is inoculated with previously digested waste from another reactor, sealed and allowed to digest naturally. The leachate from the bottom of the reactor is heated and recirculated to promote and accelerate the degradation process. In **leach bed** systems this leachate is treated in a wastewater digester prior to recirculation, and thus the solid phase digester is essentially the hydrolysis/acid forming stage of a two phase system. This type of approach has the distinct advantage of reduced materials handling but overall degradation of the organic matter can be lower than the other systems.

Costs

The costs for AD vary considerably depending on the nature of the feedstock. Municipal waste plants tend to be the most expensive as the waste is the most difficult to deal with and plants designed for agricultural wastes (considered in section 4.2) tend to be significantly cheaper. Detailed cost data is scarce, but some data from specific facilities is given in Table 5.8; this indicates the wide variation currently seen in costs. Overall, costs are reducing dramatically as the technology improves, this is shown is Figure 5.3 which provides the capital costs from a number of facilities (Lusk et al, 1998).

Figure 5.3 Capital costs of anaerobic digestion plants



Source: Lusk et al, 1998.

Table 5.8 Cost data for anaerobic digestion plants

System/ feedstock	Year	Scale t/a	Capital cost	Operational cost/t	Electricity output MWh/a	Capital cost (1995 ECU/t)	Oper'nal cost (1995 ECU/t)
med. solids/ sorted MSW	1996	78 200	£10 500k	£10	7100	165	13
med. solids/ biowaste*	1996	16 200	£6 000k	£27	2620	455	33
high solids/ biowaste*	1992	20 000	19 960k NLG	192 NLG	2580	481	93
low solids/ biowaste*	1992	20 000	28 679k NLG	251 NLG	3840	698	121
high solid/ biowaste*	1992	50 000	24 861k NLG	77 NLG	8700	240	37

*source separated kitchen and garden waste

Electricity produced will form an income for the plant. At present many countries provide a premium price for electricity produced from renewable energy sources, including anaerobic digestion (Table 5.9). In many cases, this premium price for electricity produced can be significant in determining the overall profitability of an anaerobic digestion plant. The solid residue may also be sold as a soil improver if it is of high enough quality (i.e. uncontaminated), although as with compost residue the price received is likely to be low.

The methane generation potential of waste is increased under the controlled conditions within an anaerobic digester. Between 10 and 50% more methane is produced per tonne of waste fed to the digester (depending on the type of waste and the design) than is generated if the waste had been landfilled. As the degradation processes are carried out under controlled conditions in a highly engineered plant, fugitive methane emissions are minimal, typically 1% or less.

Table 5.9 Prices paid to Renewable Energy (Private) Generators ECU/MWh (exc. VAT)

Country	basic	extra for "biomass"	total paid
Austria	35	12	47
Belgium	35	11	46
Denmark	41-80	38	79
Finland	36		36
France	43		43
Germany	53	21	74
Greece	55		55
Italy	23-62	73	135
Netherlands	38-70	14	52
Portugal	34		34
Spain	50-76	4	50-76
Sweden	36	0	36
UK	35	0-77	35-112

Source: Joosen and van Zuylen, 1997

5.2.1.5 Incineration
Within the EU, incineration is one of the most common options for pre-treating biodegradable wastes prior to landfilling. Incineration reduces the waste volume to ~30% of its original volume and produces an inert residue suitable for landfilling. Incineration can be used to treat all the fractions of MSW.

5.2.2 Options to reduce methane generation from landfills
Before any of the options outlined below can be adopted, it is important that waste is disposed of in controlled landfills rather than uncontrolled open dumps. Within the EU, landfilling is generally a controlled operation, but there are areas where uncontrolled dumping is still practised, although this is declining rapidly.

The draft EU Directive addresses this issue and proposes to take appropriate measures to avoid the abandonment, dumping or uncontrolled disposal of waste, by requiring all landfills to have a permit and all landfills to comply with the conditions of that permit. The draft Directive also proposes that appropriate measures should be taken to control the accumulation and migration of landfill gas and suggests that landfill gas be collected from all landfills receiving biodegradable waste and that the collected gas be treated and used. If the collected gas cannot be used to produce energy, it must be flared.

5.2.2.1 Improve site engineering
Where controlled landfills are in operation, improved site engineering can help reduce uncontrolled emissions of methane from the site. Installing barrier or lining systems will prevent the lateral migration of landfill gas from the site and its subsequent release to atmosphere. The liner or barrier system forms an integral part of the containment of the landfill and needs to be designed and constructed to suit the landfill in question. The selection of a liner system will also be influenced by the availability of materials either on-site or locally. Consequently there will be large variations in the designs and costs of a liner or barrier system based on local geology. Clay will generally be used as a mineral component of the liner system

where suitable clay is available on-site or nearby. If clay is not available but there are deposits of silty sand, a good quality bentonite enhanced soil can be formed. If no suitable materials are available, it is more probable that the liner will be formed from synthetic geomembranes.

In addition to being a material of low permeability, liners and barriers should be robust, durable, and resistant to chemical attack, puncture and rupture. Generally multiple layer lining and barrier systems are used to ensure that they remain both leachate and gas tight for the life of the landfill. Barrier and lining systems are increasingly installed at all new landfill sites and are generally specified as part of the overall planning/licensing procedure. They are now considered as an essential and integral part of the overall environmental protection measures of a landfill site.

Improving the capping and restoration layers of a landfill site is a relatively low cost option to reduce methane emissions to atmosphere. Capping landfills with an impermeable clay layer, rather than permeable soily material will reduce the amount of methane being emitted to the atmosphere by providing a physical barrier. However, even the best engineered clay caps will allow small amounts of methane through. Over many years, the clay cap can deteriorate or may dry out and crack under dry weather conditions. But if the restoration layers above the clay cap are also engineered to take advantage of biological methane oxidation activity, then even these small emissions of methane can be controlled. Biological methane oxidation is a naturally occurring process whereby methane is oxidised to carbon dioxide and water by micro-organisms. To facilitate the process, the landfill restoration layer must contain a material with an open structure to allow methane to permeate upwards but more importantly to allow oxygen to diffuse into the restoration layer. Sufficient nutrients must be available for the micro-organisms to function and the depth of the restoration layer must be sufficient to allow complete oxidation of the methane within the restoration layer.

If we can assume for all modern landfills that 80% of the landfill gas is collected and combusted, the remainder of the methane produced by the waste (20%) will pass through the restoration layer and 90% of this methane is oxidised biologically as it passes through the restoration layer. The assumption of 90% oxidation is based on laboratory studies that show landfill cover soils have a large capacity to oxidise methane (Whalen *et al.*, 1990; Kightley *et al.*, 1995). It has been estimated that under optimal conditions, a 500 mm layer of coarse, sandy soil could oxidise 100% of the methane from a 13m depth of waste. Allowing for sub-optimal conditions, a value of 90% oxidation is justified.

The costs for a combined clay cap and soil restoration layer are very dependent on the local availability of materials, otherwise considerable transport costs will be incurred for importing the materials to the landfill site. Generally, the capital costs of installing a clay cap and soil restoration layer are associated with the hire of plant and equipment, the costs of the clay and soil materials and an annual maintenance cost. Costs are dependent on the surface area to be covered but are generally in the range £15 - 30 (20-40 ECU) per square metre for a clay depth of 1 metre and a soil depth of 1 metre. General maintenance costs are about £0.03 (0.04 ECU) per square metre per year.

For a typical landfill of 1 million tonnes, with a surface area of 62,500m^2 (250m x 250 m), approximately 72,000 tonnes of methane will be generated over 50 years (assuming landfill gas has a methane content of 50%). Of this 72,000 tonnes, 80% is collected and burned, leaving some 14,400 tonnes to be emitted to the atmosphere. A 1m impermeable clay capping layer topped with a 1m soil restoration layer is installed which is capable of oxidising 90% of the remaining methane leaving a residual 290 tonnes of methane emitted over the 50 year lifetime of the landfill site.

5.2.2.2 Collection and use in energy recovery

Where a landfill site is lined and has a good quality capping layer, then the methane generated by the decomposition of the wastes will be trapped within the landfill. To prevent uncontrolled leakage of methane from the landfill as the gas pressure increases, gas collection systems are installed, to remove the methane from within the waste mass. Gas collection and combustion can dramatically reduce uncontrolled methane emissions to atmosphere (Bogner *et al*., 1995). The technology for landfill gas collection and combustion, either for energy recovery or in flares is well understood and demonstrated in many EU Member States. Gas collection is increasing throughout the EU as awareness of safety risks and the need to reduce odour nuisance has also increased.

Flaring

Modern flares are designed to work continuously and availabilities of 98% and greater are not uncommon, especially for the less sophisticated smaller units below 1500m^3 per hour, which form the majority of the flares in operation. Combustion efficiency is difficult to assess, but is generally greater then 99% for methane, particularly if the air supply is well adjusted.

Precise costs of landfill gas collection can only be calculated on a site by site basis because of site specific factors such as waste type, depth and area. Flare costs depend on local emissions regulations and best practice requirements, but are generally in the range £90k - 120k (120k-160k ECU) for a unit dealing with 500 m^3 per hour of landfill gas. These costs include the gas compound required to house the flare and the associated pumps and compressors. Operating costs are in the range £3k - 7k (4-9.4k ECU) per year depending on the degree of sophistication of the flare and control systems. Generally the flare will have a lifetime of 10 years and so will have to be replaced periodically.

Energy recovery

Although collecting landfill gas for burning is the priority, use of the methane for energy recovery should be encouraged in all cases. The options for using landfill methane are:

- direct use in boilers or for process heating;
- electricity generation;
- upgrading to substitute natural gas;
- vehicle fuelling;
- fuel cells;
- as a chemical feedstock.

Only where no suitable end use can be found or the project cannot achieve a high enough financial return, should the collected gas be flared.

Direct combustion as a medium calorific value fuel is the cheapest and simplest way to recover energy from landfill gas. It can be burnt in industrial boilers, brick kilns and lime or cement kilns. Landfill gas can also be used for heating greenhouses, district heating and for drying purposes in industrial processes. Direct combustion is an efficient way to recover energy from landfill gas - over 80% of the calorific value of the methane can be recovered as useful energy. Burning landfill gas is similar to burning a dilute natural gas. If the combustion fuel is a mixture of natural gas and landfill gas, only relatively simple modifications are required to adapt the burner. If the landfill gas is the predominant or only fuel, modifications are needed to balance the lower energy content of the gas compared to natural gas.

The main disadvantage to the use of landfill gas as a direct fuel is that the consumer must be located near the landfill site. The costs of transporting the gas over long distances are high and it may be difficult to obtain permission to lay pipework on or under intermediary land between the site and the end user. Ideally the end user should be able to use the landfill gas continuously because the flow from the landfill site cannot be turned on and off to match the end user needs.

Using a low pressure boiler system, about 600m^3 per hour of methane will be combusted, equivalent to an output of 3 MWth.

If direct combustion of landfill gas is not a viable option at a site, the most likely alternative is to **generate electricity**, with or without heat recovery. Electricity generation from landfill gas is a successfully demonstrated technology within the EU, and it is estimated that there are more than 200 schemes operating currently.

For an electricity generation scheme, the following need to be considered:
- prime mover and generator;
- associated civil works to house the generation plant;
- electrical and control equipment;
- access to the distribution grid;
- annual running costs including the operation and maintenance of the plant.

It is estimated that a 1 MW scheme will typically cost ECU 75,000 for capital costs and a further ECU 135 per kW per year in operating costs.

For an example landfill containing 1 million tonnes of mixed waste, 7 m^3 of landfill gas with a methane content of 50% is produced every year by every tonne of waste. Approximately 80% of this landfill gas is collected, so the landfill can support an electricity generation scheme of 1 MW, assuming a load factor of 80% and a conversion efficiency of methane to electricity of 33%.

Other uses for landfill gas are not yet as widely demonstrated as either direct use or electricity generation, but the use of landfill gas as a **substitute natural gas** was in fact one of the first uses of landfill gas.

The major disadvantage to the use of landfill gas as a substitute natural gas is the need to remove all the carbon dioxide, nitrogen and oxygen from the raw gas, as well as most of the trace components. The most common techniques for treating landfill gas prior to pipeline injection are:

- water scrubbing
- non-aqueous solvent based extraction systems
- iron-oxide beds
- activated carbon

- pressure swing absorption
- membranes

Often two or more of these techniques have to be used sequentially to achieve the desired level of purity in the substitute natural gas. This means that considerable costs are incurred in the clean up of the landfill gas which makes it uneconomic when compared to for example natural gas extracted from the North Sea, unless a premium payment can be guaranteed for the substitute natural gas.

Upgraded landfill gas can be used as a **vehicle fuel** to fuel conventional spark ignition or diesel engines that have been modified to run on natural gas. The same technology used to provide substitute natural gas can be used to provide a high calorific value vehicle fuel. However, the costs of providing vehicle fuel from landfill gas are high. Using landfill gas as a vehicle fuel has been demonstrated in a number of plants world-wide. Most of these are prototype plants with a limited number of vehicles running and so the technology is unproven both technically and commercially. However technology using biogas produced from anaerobic digestion, is more advanced and is likely to soon become a commercial activity. For example, in Sweden bus fleets and an increasing number of private vehicles are using biogas, and public biogas filling stations have been set up. This suggests that it is likely that the use of landfill gas in vehicles may also become a proven technology in the future.

Fuel Cells offer a future option for the generation of electricity from landfill gas. Over 200 fuel cell systems running on natural gas have been demonstrated world-wide but to date there has only been one demonstration on landfill gas. At present, fuel cells are still at a relatively early stage of development and, although costs are reducing, they are substantially more expensive than equivalent conventional heat engines. Additionally, clean up of the landfill gas for use as a feedstock to the fuel cells needs to be much more effective than if the gas were to be used as a vehicle fuel, adding to the costs of the technology.

Fuel cells do offer great environmental and electricity conversion benefits and it is likely that further development of the technology will eventually lead to their widespread use at landfill sites. They offer:

- electrical conversion efficiencies of ~40% which are higher than conventional heat engines;
- very low noise in their operation;
- no combustion products such as NOx;
- minimal maintenance and labour requirements;
- modular units of, for example, 200 kW for use at both small and large landfills.

It is technically possible to use landfill gas as a substitute in petrochemical processes using methane as a **chemical feedstock**, but as yet there are no commercial applications of this process. Methanol production, as an alternative fuel, is perhaps the most feasible option, but again this has not been demonstrated commercially for landfill gas.

5.3 COST OF OPTIONS

With respect to landfill, enough cost and performance data is available to calculate the cost-effectiveness of eight options:

- paper recycling
- composting of waste;
- anaerobic digestion of waste;
- the incineration of MSW destined for landfill;
- the capping of landfill;
- the flaring of landfill gas;
- the direct use of landfill gas;
- the use of landfill gas to generate electricity for export.

The methodology used to calculate the annualised cost (in 1995 ECU) of the measures, and details of exchange rates and conversion from different years are described in Appendix 1. A discount rate of 8% has been used in annualising costs.

In the case of composting, anaerobic digestion some energy inputs will be required (e.g. fuel for vehicles, running equipment on site) which will lead to CO_2 emissions. A complete balance of all greenhouse gas missions for these options has not been carried out, but an approximate analysis shows that CO_2 emissions from fuel use will be insignificant compared to the reduction in methane emissions. Furthermore in the case of anaerobic digestion where heat and power are produced, there will be a CO_2 credit for the (mainly) fossil fuel produced energy this will displace. In the case of incineration power use at the plant is typically (apart form start up) drawn from power produced on site, and this is accounted for in the estimation of savings. There will be some liquid fuel use in vehicles etc., but again the resultant CO_2 emissions are likely to be insignificant compared to CH_4 savings and will be more than offset by the CO_2 credit for the (mainly) fossil fuel produced energy, which heat and power from the incinerator will displace.

5.3.1 Cost Assumptions

5.3.1.1 Paper Recycling

Investment costs and operating and maintenance costs associated with the recycling of waste paper are highly dependent upon the grade of paper being produced. Typical costs are given below for a 200 tonne/day deinking plant, producing pulp of an equivalent quality to virgin pulp, for use in newsprint or fine paper production.

- The non-recurring cost of the plant is £25M to £30M (1997 prices) and the plant lifetime is 25 years.
- Recurrent costs are in the range of £60 - £90 per tonne, depending upon quality requirements.
- The cost of waste paper to the plant is £50/tonne; yield losses are 10%. Virgin pulp prices are £250/tonne.
- The waste paper is assumed to have a degradable organic content of 40% (IPPC, 1997). If landfilled, 1 tonne of this waste would have generated 0.205 t of methane, of which 80% is

recovered and combusted so that overall the plant abates 0.04 t of methane per tonne of paper recycled. It is assumed that sludge from the process is not disposed of to landfill.

5.3.1.2 Composting

As discussed in Section 5.2 there is a wide variation in costs of facilities, and also over the additional costs of source separated collection and any income for solid residue. In order to help represent the likely variation in costs, two plant are evaluated, a UK turned windrow scheme and a Dutch tunnel scheme.

For a turned windrow scheme in the UK processing 8000 t/a the following assumptions are made:

- The non-recurring cost of the plant is £600k (1995 prices) and its lifetime is 15 years
- The recurring operational cost is £13/t, i.e. £104k per year (1995 prices)
- The additional cost of source separated collection is £8/t, i.e. £64,000/yr (with a range of £5 to £8/t).
- Of 2,000 t of poor quality material produced, half is sent to landfill. The cost of landfill disposal is 18 ECU/t (1993 prices) for disposal to a rural site with energy recovery (Coopers and Lybrand, 1996) so cost of disposal is £15,827/yr (1995 prices)
- 3,000 t of compost are produced; two cases are evaluated, a worst case, where zero income is received and a best case where £5/t (1995 prices) or £15,000/yr is received.
- The avoided cost of disposing the waste to a rural landfill site with energy recovery is 18 ECU/t (1993 prices) (Coopers and Lybrand, 1996), so avoided costs are £126 640/t
- The waste going to the plant is assumed to be 50% food waste and 50% garden and park waste (by weight), and it thus has a degradable organic content of 16% (derived from IPPC, 1997). If landfilled, 1 tonne of this waste would have generated 0.082 t of methane, of which 80% is recovered and combusted so that overall the plant abates 131 t of methane per year.

For a tunnel composting facility in the Netherlands, processing 25000 t/a the following assumptions are made:

- The non-recurring cost of the plant is 9,104k NLG (1992 prices) and its lifetime is 15 years.
- The recurring operational cost is 77 NLG/t i.e. 1925k NLG per year (1992 prices).
- The additional cost of source separated collection is 10 ECU/t (1997 prices), i.e. 228 546 NLG/yr (1995 prices) (with a range of 5 to 15 ECU/t).
- Of 7,000 t of poor quality material produced, half is sent to landfill. The cost of disposal to a Dutch urban landfill with energy recovery is 36 ECU/t (1993 prices) (Coopers and Lybrand, 1996), so the annual cost of disposal is 269 243 NLG/yr.
- The avoided cost of landfill disposal is 36 ECU/t (1993 prices) (Coopers and Lybrand, 1996), so the total avoided cost of disposal is 1 923 167 NLG/yr.
- 10,000 t of compost are produced; two cases are evaluated, a worst case, where zero income is received and a best case where £15 ECU/t (1997 prices) or 150,000 ECU/yr is received.
- The waste going to the plant is assumed to be 50% food waste and 50% garden and park waste (by weight), and it thus has a degradable organic content of 16% (derived from IPCC, 1997). If landfilled, 1 tonne of this waste would have generated 0.082 t of methane, of which 80% is recovered and combusted so overall the plant abates 411 t of methane per year.

5.3.1.3 Anaerobic Digestion

Due to the inherent variability of the systems, costs are given for a typical facility, assumed to be located in the UK and processing 50,000 t/pa of source separated MSW:

- The non-recurring cost of the plant is £8M (1995 prices) and its lifetime is 15 years
- The recurring operational cost is £24/t, i.e. £1.2M per year (1995 prices)
- The additional cost of source separated collection is £8/t, i.e. £400k/yr (with a range of £5 to £8/t).
- 5,000 t of poor quality material produced which is sent to landfill. The cost of landfill disposal is 25 ECU/t (1993 prices) for disposal to an urban site with energy recovery (Coopers and Lybrand, 1996) so cost of disposal is £109,975/yr (1995 prices).
- 34,500 t of compost and 3,000 t of liquor are produced. Two cases are evaluated: a worst case, where zero income is received for the compost and costs to dispose of the liquor as waste water are £20/t (1997 prices), and a best case where £5/t (1995 prices) or £172,500/yr is received for the composts and a use is found for the liqour, so there are no disposal costs.
- The avoided cost of disposing the waste to a urban landfill site with energy recovery is 25 ECU/t (1993 prices) (Coopers and Lybrand, 1996), so avoided costs are £1,099,750/t (1995 prices)
- The plant produces 7500 t of biogas which is 55% methane, and this is used to generate 8000 MWh for export. A price of 2.1 p/kWh (1996 prices) is received for the electricity produced.
- The waste going to the plant is assumed to be 50% food waste and 50% garden and park waste (by weight), and it thus has a degradable organic content of 16% (derived from IPPC, 1997). If landfilled, 1 tonne of this waste would have generated 0.082 t of methane, of which 80% is recovered and combusted so that overall the plant abates 821 t of methane per year. However 1% of the methane produced in the plant leaks out to the atmosphere (41t), so that the net saving is 780 t/yr.

5.3.1.4 Incineration of Waste Destined for Landfill

In order to estimate the cost–effectiveness of this measure the following assumptions have been made based on Patel, (1996):

- The non-recurring cost is £56 million per plant (in 1996 prices). The unit burns 200,000 tonnes of waste per year with export of heat and electricity.
- The annual recurring costs are £4.2 million per plant per year (in 1996 prices). The plant produces 90 000 MWh of electricity per year and the selling price of electricity is 2.1p/kWh (1996 prices). Therefore, the value of electricity sales is £1.89 million per plant per year. The plant produces and sells an equal amount of heat; the price received for the heat is assumed to be equivalent to cost of heat produced in a natural gas boiler of 90% efficiency with a gas price of £0.0043 per kWh. Therefore the value of heat sales is £0.43M per year.
- As the plant avoids disposal to landfill, there is an income to the plant for disposing of the waste which is taken as equal to the cost of disposal to an urban landfill with energy recovery of 25 ECU/t (1993 ECU) (Coopers and Lybrand, 1996).
- The option has a lifetime of 20 years and requires 2 years to implement.
- Two hundred thousand tonnes of waste per year are incinerated instead of landfilled. If landfilled this waste would produce 14,373 t of CH_4, of which, on a controlled site, 80% would be collected and combusted, and 20% of which would be emitted to atmosphere.

Compared to a controlled landfill, the reduction potential of the plant is thus 2,875 t per year.

5.3.1.5 The Capping of Landfill

To assess the cost-effectiveness of this measure the following assumptions have been made:

- The non-recurring cost is £22 per m^3 (in 1997 prices), with a range of between £16.5 and £29.5 per m^3. One cubic metre of capping material is required to cover one square metre of landfill. Consequently, non-recurring costs range between £16.5 and £29.5 per m^2 (£22 per m^2 best estimate).
- The annual recurring cost is £0.03 per m^2 of landfill per year.
- The option has a lifetime of 50 years and requires one year to install.
- One million tonnes of waste which is landfilled generates 71,867 tonnes of methane in total, 80 per cent of this is collected and burned, while the remaining 20 per cent escapes. The surface area of the landfill is 62,500 m^2. Fugitive methane emissions per m^2 are thus 4.6 kg per year. The capping of landfill reduces fugitive methane emissions by 90 per cent. Consequently, the reduction potential of this measure is 4.1 kg CH_4 per m^2 per year.

5.3.1.6 Flaring Landfill Gas

The following assumptions have been made in estimating the cost–effectiveness of this measure (Robert Eden Organics Ltd, 1998):

- The non-recurring cost of the measure is £90,000 per flare unit (in 1997 prices).
- The annual recurring cost is £5,000 per flare unit per year, with a range of £3,000 to £7,000 per unit per year.
- The option has a lifetime of 10 years and requires one year to implement.
- The flare unit is designed to burn 500 m^3 of landfill gas per hour at 35 per cent methane. The availability of the flare is 98 per cent. The flare therefore burns about 1.5 million m^3 of CH_4 per year. Assuming that there are 16 kg of CH_4 per 22.41 m^3 of CH_4, the reduction potential of the option is 1,073 t CH_4 per unit per year.

5.3.1.7 Direct Use

In order to estimate the cost–effectiveness of this measure the following assumptions have been made:

- The non-recurring cost of this option is £70,000 per boiler (in 1997 prices). The boiler has a capacity of 3 MW and has an availability of 95 per cent.
- The annual recurring costs are £8,000 per boiler per year (in 1997 prices), with a range of between £5,000 and £12,000 per boiler per year. The boiler produces 24,966 MWh of heat per year; the price of natural gas is 0.493 p/kWh (Energy Trends, 1998) and the boiler has an efficiency of 90%. Therefore, the value of foregone natural gas purchases is £136 758 per boiler per year.
- The option has a lifetime of 20 years and requires one year for implementation.
- The boiler is designed to burn 600 m^3 of landfill gas per hour at 60 per cent methane. The boiler therefore burns about 3.01 million m^3 of CH_4 per year. Assuming that there are 16 kg of CH_4 per 22.41 m^3 of CH_4, the reduction potential of the option is 2,139 t CH_4 per boiler per year.

5.3.1.8 Electricity Generation for Export

The following assumptions have been made in estimating the cost–effectiveness of this measure (de Rome, 1998):

- The non-recurring cost is 725 ECU per kWe (in 1997 prices), with a range of between 600 and 900 ECU per kWe. The capacity of the plant is 1 MW. The load factor is 80 per cent, therefore, annual output is 24 * 365 * 0.80 = 7,008 kWhe per kWe. Consequently, non-recurring costs are equal to 725 ECU per kWe divided by 7,008 kWhe per kWe, i.e. 0.1035 ECU per kWhe.
- The annual recurring costs are 135 ECU per kWe per year, with a range of between 65 and 200 ECU per kWe per year. Consequently, recurring costs are equal to 135 ECU per kWe divided by 7,008 kWhe per kWe, or .019 ECU per kWhe per year. The price of electricity is 0.025 ECU/kWh.
- The option has a lifetime of 10 years and requires one year for implementation.
- A plant of this capacity burns 600 m^3 of landfill gas per hour at 50 per cent methane. The boiler therefore burns about 2.1 million m^3 of CH_4 per year. Assuming that there are 16 kg of CH_4 per 22.41 m^3 of CH_4, the reduction potential of the option is 0.0002 t CH_4 per kWhe per year.

5.3.2 Cost-effectiveness

Table 5.10 summarises the cost-effectiveness of measures to reduce emissions of methane from solid waste disposal.

Table 5.10 indicates that the most cost-effective measure is paper recycling, and that measures involving the recovery of landfill gas are the next most cost-effective options. Of these, the direct use of landfill gas to produce heat is the most cost-effective option, and is in fact cost-positive, with the value of the heat produced more than off-setting the operational costs and annualised capital costs. Flaring of landfill gas and electricity generation from landfill gas are both relatively cost-effective options (23 and 44 ECU/t respectively). The cost of abating methane by designing the landfill cover to optimise microbial oxidation of fugitive emissions is an order of magnitude greater than the cost of options involving the recovery of landfill gas.

Apart from paper recycling, the options involving diversion of organic waste from the waste stream all have significantly higher costs of abatement, of above 1000 ECU/t CH_4. Of the options considered, the most cost effective is the turned windrow system in the UK (about 1000 to 1200 ECU/t CH_4), then incineration (about 1400 ECU/t CH_4) and then tunnel composting and anaerobic digestion (1500 to 2000 ECU/t CH_4). For composting and anaerobic digestion, uncertainties in the additional cost of source separated collection and the issue of whether an income is received for the digestate/residue, can affect the abatement cost per tonne by up to 15%. The cost-effectiveness of all these alternative disposal routes is at least doubled if no allowance is made for the avoided cost of disposal to landfill, and shows the sensitivity of the costs to the avoided cost of disposal or the ‘gate fee’ which these types of facilities might expect to receive for disposing of waste.

Table 5.10 Cost-effectiveness of Waste Related Measures

Mitigation Measure		ECU/t CH_4	ECU/t CO_2-equiv
Paper Recycling	Best estimate	-2208	-105
	Range	-2669 to -1747	-127 to -83
Composting (turned windrow, UK)	Best estimate	1 033	49
Allowing for			
- uncertainty in source separation costs	Range	812 to 1 180	39 to 56
- no income for residue		1 171	56
- no avoided cost of disposal		2 197	105
Composting (tunnel composting, Nls)	Best estimate	1 794	88
Allowing for			
- uncertainty in source separation costs	Range	1 495 to 2 093	71 to 100
- no income for residue		2 139	102
- no avoided cost of disposal		4 013	191
Anaerobic digestion	Best estimate	1 858	87
Allowing for			
- uncertainty in source separation costs	Range	1 627 to 2 013	77 to 96
- no income for residue		2 216	105
- no avoided cost of disposal		3 557	169
Incineration	Best estimate	1 423	68
- no allowance for avoided cost of disposal		3 130	149
Capping of Landfill	Best estimate	592	28
	Range	446 to 790	21 to 38
Flaring Landfill Gas	Best estimate	23	1
	Range	15 to 31	0.7 to 1.5
Direct Use of Landfill Gas	Best estimate	-76	-3.6
	Range	-73 to -77	3.5 to 3.7
Generation from Landfill Gas	Best estimate	44	2
	Range	-14 to +104	-1 to 5

5.4 APPLICABILITY OF MEASURES

5.4.1 Base Line Trends

As discussed above methane emissions from waste disposal, depend on total quantities of waste generated, the fraction of this waste disposed to landfill, the degradable organic content of the waste going to landfill, and the amount of landfill gas recovery which takes place. To predict future emissions it is therefore necessary to predict how these factors will develop. As already discussed there are a number of measures which may affect all of these factors, and many countries have already begun to implement some measures.

In order to establish a baseline against which to judge the impact of measures, a business-as-usual scenario is developed in which the fraction of waste disposed to landfill, the degradable organic content of the waste and the amount of landfill recovery which takes place are all assumed to remain constant from 1994 onwards. The quantity of waste generated per capita is assumed to remain constant, and changes in population are based on those assumed in the DGXVII energy and emissions projections (Capros, 1997).

5.4.2 Emissions under a Business as Usual Scenario

Emissions under a business-as-usual scenario were estimated using the time dependant IPCC methodology (IPCC, 1997). Values used for the per capita waste generation rate, fraction of MSW disposed to landfill and percentage of methane recovered are detailed in Table 5.11. For the first two parameters the values provided in the IPCC guidelines were used, unless new country specific data was available from the Second National Communications. The percentage of methane recovered was derived from information in the Second National Communications wherever possible; in all other cases, data was taken from CCPM, 1997. The fraction of waste which is degradable organic content was taken from IPCC guidelines. All countries were assumed to be using deep sites (i.e. no shallow or open dumping). As no data was available on waste arisings in previous years, these were assumed to remain constant.

Overall emissions for 1990 and 1994 estimated using this methodology are extremely close (less than half a percent difference) from emissions reported by countries in their second national communications. As shown in Table 5.12 there are however some significant differences for some countries. As previously, the Second National communications do not provide enough detail for the precise reasons for these differences to be resolved. Some countries (e.g. UK and Netherlands) are known to use a more accurate time dependent model for estimating emissions and this may be the reason for the discrepancy in these cases. In the case of Greece and Italy, where the estimate is one and one and a half times greater than the national estimate, a contributing factor to the difference may be the existence of shallow dumps (with lower emissions factors). For all countries, the methodology used will overestimate emissions where the amount of waste disposed to landfill has been increasing with time.

In order to ensure consistency with national estimates, emissions estimates for future years are scaled by the difference in the 1990 emissions estimates. These projections to 2020 are shown in Table 5.13. Overall in the EU emissions are predicted to rise slightly (from 1990 levels) by 5% by 2010 and 6% by 2020, reflecting the expected increase in population and hence increase in waste generated. As discussed the assumptions implicit in the methodology will mean that this increase has been overestimated. In practice future emissions are unlikely to increase in this way due to the large number of policies and measures which countries have already begun to put in place. These are summarised below.

Table 5.11 Values used in Projections

Country	MSW generation rate (kg/cap/day)		Fraction of MSW disposed to landfill		Percentage of methane recovered	
	1990	1994	1990	1994	1990	1994
Austria	0.92	0.92	0.40	0.40	20%	20%
Belgium	1.10	1.10	0.43	0.43	12%	12%
Denmark	1.26	1.26	0.20	0.18	20%	20%
Finland	1.70	1.70	0.77	0.59	0%	2%
France	1.29	1.29	0.46	0.46	20%	20%
Germany	1.65	1.73	0.66	0.66	26%	29%
Greece	0.85	0.85	0.93	0.93	5%	5%
Ireland	1.33	1.33	1.00	1.00	5%	5%
Italy	0.94	0.94	0.88	0.88	10%	10%
Luxembourg	1.34	1.34	0.35	0.35	20%	20%
Netherlands	1.58	1.58	0.67	0.67	3%	10%
Portugal	1.40	1.40	0.75	0.75	5%	5%
Spain	0.99	0.99	0.85	0.85	0%	6%
Sweden	1.01	1.01	0.44	0.44	28%	38%
UK	1.90	1.90	0.90	0.90	20%	20%

5.4.3 Existing Policies and Measures

Table 5.14 shows the trends and national policies for reducing methane emissions from waste.

Many countries have regulations in place to reducing waste to landfill through recycling and/or combustion of waste. Some of the strongest regulations, such as those in Denmark, France, Germany and the Netherlands, prohibit the landfilling of combustible waste after a certain date. Most countries also have policies in place to increase the collection and use of landfill methane from existing and new landfill sites. In addition to the policies already set out, all countries will have to meet the requirements of the draft Landfill Directive (assuming it comes into force).

5.4.4 Applicability of Measures

Three basic types of measures have been identified, those involving landfill gas recovery, improved capping of landfills to maximise optimisation of fugitive emissions, and measures which divert organic waste from landfill, (paper recycling, anaerobic digestion and composting). The impact of each of these measures is not additive, and 5 scenarios were thus examined, where the measures were deployed independently or in combination. The scenarios examined and assumptions made were:

Table 5.12 Comparison of Estimates of Annual Emissions from Landfills

	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
1990																
Emissions estimate based on IPCC methodology (kt)	67	111	30	170	703	1675	204	119	1067	4	400	264	839	72	1410	**7136**
Emissions as reported in Second National Comm. (kt)	193	173	71	126	758	1777	102	136	302	4	562	493	472	85	1890	**7144**
Difference %	-65%	-36%	-58%	35%	-7%	-6%	100%	-12%	253%	8%	-29%	-47%	78%	-16%	-25%	-0.1%
Difference (kt)	-126	-62	-41	44	-54	-102	102	-17	765	0	-162	-229	368	-13	-480	**-8**
1994																
Emissions estimate based on IPCC methodology (kt)	69	113	27	130	722	1737	212	122	1063	5	386	267	814	64	1434	**7163**
Emissions as reported in Second National Comm. (kt)	187	184	72	122	667	1780	105	136	427	2	505	528	658	61	1790	**7223**
Difference %	-63%	-39%	-62%	6%	8%	-3%	102%	-11%	149%	102%	-24%	-49%	24%	4%	-20%	0.2%
Difference (kt)	-118	-71	-45	8	55	-45	107	-14	635	2	-120	-261	155	3	-358	**-68**

Table 5.13 Emissions from Landfills under a Business-as-Usual Scenario (kt CH_4 per year)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15	Change from 1990
1990	193	173	71	126	758	1777	102	136	302	4	562	493	472	85	1890	**7144**	
1994	187	184	72	122	667	1780	105	136	427	2	505	528	658	61	1790	**7223**	0%
2000	202	178	66	97	787	1867	107	140	302	4	550	504	461	76	1938	**7280**	3%
2005	205	182	67	100	810	1933	111	143	305	4	572	511	467	79	1960	**7449**	4%
2010	208	184	68	102	833	1973	113	147	307	4	587	517	473	80	1981	**7576**	5%
2020	209	185	68	103	857	1971	114	151	306	4	602	518	473	81	1982	**7623**	6%

Table 5.14 Existing Policies in EU Member States to Reduce Landfill Emissions

Country	National Policies and Targets	Estimated Reduction due to measure (kt/yr)		
		in 2000	in 2010	in 2020
Austria	(i) Increased collection and use of landfill gas. (ii) Planned regulation to reduce proportion of organic waste in landfill to 5% by 2004.		-2	
Belgium	(i) Restrictions on new landfill sites. (ii) All existing sites to include gas recovery.	-19		
Denmark	Action Plan for Waste and Recycling introduced. 50% of waste to be recycled by 2000 and landfilling of combustible wastes prohibited after January 1997.	-8	-32	
Finland	Existing and planned legislation to encourage waste reduction, recycling and alternative treatment techniques. Long term aim to eliminate landfilling of organic waste.	-2	-29	-40
France	(i) Only wastes with little or no degradable matter landfilled after 2005. (ii) Methane collected and burned in all landfills established between 1995 and 2002; programme to recover methane from landfills closed after 1995.	-35	-126	-161
Germany	(i) Only mineralised and inert waste to be landfilled after 2005. (ii) Landfill methane from operational and closed sites to be collected and burned.	-37	-83	-92
Greece	(i) Methane collected and burned in all new large landfills by 1999. (ii) Recycling programmes to reduce mass of biodegradable waste landfilled.		not estimated	
Ireland	(i) National recycling strategy in place. (ii) Grant assistance for recycling projects. (iii) Support through renewable energy schemes for generation from landfill gas. (iv) Development of codes of good practice for landfill. (v) Plans for 30 MW waste incinerator		not estimated	
Italy	(i) All landfills to collect and burn landfill gas. (ii) Incentives to generate 100 MWe from landfill gas. (iii) Regional plans to reduce proportion of waste landfilled from 90% in 1994 to 42% in 2000.		2nd National Communication not available	
Luxemb'g	No details available		No details available	
N'lands	(i) Progressive ban on landfilling of combustible wastes from 1995. (ii) Methane collected and burned in all new landfills from 1994.	-40	-71	
Portugal	Methane from landfill expected to reduce due to increased use of incineration.			
Spain	(i) Landfills receiving over 100 kt of waste to have gas collection and faring or utilisation to meet draft directive requirements (ii) Compost 50% of MSW (no target year given).			
Sweden	(i) Considering proposals for a tax on landfilled waste. (ii) Installing systems for recovery of energy from landfill methane. (iii) 1992 Climate Bill targeted 30% reduction of methane from landfill gas by 2000.	-50	-76	
UK	(i) From 1994 all methane collected and utilised (or burned) in all new sites receiving degradable wastes. (ii) From 1994 combustion of landfill gas at all existing sites with significant remaining capacity, where significant gas production likely. (iii) Reduction of proportion of wastes landfilled. Controlled waste to landfills to be reduced from 80% in 1990 to 60% in 2005. (iv) Recovery of value from 40% of municipal waste by 2005.	-46	-77	-95

Source: First and Second National Communications and CCPM, 1997.

- **Improved recovery of landfill gas (A):** almost all Member States have a commitment to recover landfill gas from new sites. It was assumed that from 1995 onwards all new sites would be fitted with a landfill gas recovery system and that 80% of methane generated at the site would be recovered and either flared or used to generate heat and/or power. The average lifetime of a landfill is assumed to be 15 years so every successive year an additional 1/15th of the waste disposed of to landfill goes to a new site with landfill gas recovery. Thus by 2010, all waste is disposed of to sites with landfill gas recovery. The overall recovery rates predicted for Member States in 2010 and 2020 are shown in Table 5.15; the recovery rate does not reach 80% as there is always some methane generated and emitted from old sites where there is no landfill gas recovery.

Three options for using recovered gas were identified, direct use, power generation, and flaring, but it is not possible to estimate what proportion of the gas recovered might be used in each of these three options. The direct use of gas for heat generation was the most cost-effective option, but in practice implementation may be limited by the availability of a suitable heat user within a reasonable distance of the site. There may be more opportunities for the direct use of the gas in countries such as Denmark, where district heating schemes are common. Generation is more cost-effective than flaring and is likely to be suitable for a large number of sites; cases where it may not be possible to implement it are, for example, very small sites.

- **Reduction of biodegradable organic waste to landfill (B):** it is assumed, as a minimum that Member States can achieve the targets set out in the common position (March 1988) on the proposed landfill directive, i.e. biodegradable municipal waste going to landfills is reduced to 75% of the total amount (by weight) of biodegradable municipal waste (produced in 1995) by 2006, 50% of the total amount by 2009 and 35% of the total amount by 2016. Countries (Greece, Ireland, Italy, Spain and the UK) which have a high reliance on landfill (i.e. more than 80% of waste is currently landfilled) are assumed to take an additional 4 years to comply. As most Member States already have policies in place to reduce waste to landfill, the amount of biodegradable waste going to landfill is assumed to fall linearly from 1995 and 2006 (2010 for countries with derogations) and between the other milestone dates in the Directive. From 2006/2020 the amount of biodegradable waste going to landfill is assumed to remain constant.

  The cost-effectiveness of paper recycling, composting, anaerobic digestion and incineration as a way of disposing of the biodegradable waste diverted from landfill were estimated. Paper recycling was found to be a very cost-effective way of reducing CH_4 emissions with a cost of about -2200 ECU/t. the other options were found to be considerably less cost-effective, varying by about a factor of 2 (from about 1000 ECU/t for composting to 1900 ECU/t for anaerobic digestion). Once again it is not possible to determine accurately what proportion of the waste diverted from landfill will be treated by each of these three methods. While some elements of waste policy may be set nationally, decisions on waste disposal strategies are usually taken at the regional or even local level, and are influenced by a large number of factors; for example, different solutions may be appropriate for urban and rural areas. The costs of the options were shown to vary with a number of factors, e.g. the cost of additional source separation, the availability of a market for residues and so the relative cost-effectiveness of options could vary from region to region.

In the case of paper recycling, data on the types of paper currently found in MSW (Atkinson and New, 1994; Environment Agency, 1994a and 1994b) suggest that about half of the paper is of qualities acceptable to the processors. From the data in Table 5.3 on MSW composition, paper forms about 27% of EU MSW; given its high degradable organic content (of 40%)(IPCC, 1997), it accounts for about 60% to 70% of biodegradable carbon in MSW. It is assumed that half of the paper currently going to landfill can be recycled by 2020.

- **Improved landfill gas recovery and improved capping of landfills (A+C);** it is assumed that the benefits from improved capping of landfills will have been demonstrated on a commercial site by 2000, and that from 2000 onwards, all sites which are closed will have an improved cap layer. A landfill site typically receives waste for a period of 15 years, and thus it is assumed that from 2000 onwards in each successive year an additional 1/15th of all the waste which has been deposited since 1985 is in a site which is closed and then capped. The cap layer oxidises 90% of fugitive methane emissions (which are themselves 20% of the) methane generated in the site.

- **Improved landfill gas recovery and reduction of biodegradable organic waste to landfill (A+ B);** the reduction in biodegradable waste assumed in scenario B is accompanied by a programme of fitting landfill gas recovery to all new site as detailed in A.

- **Improved landfill gas recovery and reduction of biodegradable organic waste to landfill and improved capping of landfills:** a combination of all three measures.

The reductions which would be achieved for each of the scenarios (compared to the B-A-U scenario is shown in Table 5.16 for the EU as a whole and in Tables 5.17 for Member States in 2010 and 2020. If all three measures were to be combined then reductions of 5006 kt in 2010 and 6956 kt in 2020 compared to the business as usual scenario would be achieved. These reductions are 66% and 91% respectively of projected emissions in 2010 and 2020 under a business as usual scenario.

As indicated in Table 5.14, a number of countries have plans to introduce greater or more rapid diversion of biodegradable wastes to landfill than those contained in the proposed landfill directive. The additional reduction which would be achieved if these more stringent measures were implemented in addition to those shown in Table 5.16 are shown in Table 5.18. The additional reduction achieved in 2010 (304 kt) is 6% of those already considered; by 2020, the additional reduction is only 36 kt. As it is believed that some of these targets may be modified, given the revisions to the proposed landfill directive, the additional reductions are not included in the cost-effectiveness curves or the with measures implemented scenario (see below).

Table 5.16 Estimate of Achievable Reductions in EU Landfill Emissions (kt of CH_4)

	Measure	**2000**	**2005**	**2010**	**2020**
A	improved landfill gas recovery	384	1237	2230	3683
B	reduction of biodegradable organic waste to landfill	762	1643	3276	5084
A+B	improved landfill gas recovery and reduction of biodegradable organic waste to landfill	1105	2604	4535	6311
A+C	improved landfill gas recovery and improved capping of landfills	384	1604	3064	5620
A+B+C	all three measures	1105	2890	5006	6956

Table 5.15 Predicted Recovery Rates if all New Sites are fitted with Landfill Gas Recovery

Year															
1990	10%	10%	12%	0%	20%	26%	2%	2%	13%	7%	3%	2%	2%	28%	23%	17%
1994	10%	10%	12%	2%	20%	29%	2%	2%	13%	7%	10%	2%	6%	38%	23%	19%
2000	15%	15%	17%	7%	24%	34%	7%	7%	18%	12%	15%	7%	11%	42%	26%	23%
2010	37%	37%	38%	29%	44%	51%	29%	29%	39%	34%	37%	29%	33%	56%	45%	43%
2020	55%	55%	55%	50%	59%	63%	50%	50%	56%	53%	55%	50%	52%	66%	59%	58%

Table 5.17 Estimate of Achievable Reductions in Landfill Emissions by Member States (kt of CH4)

Measure	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
Reduction in 2010 (kt)																
A	63	56	20	28	246	610	31	40	90	1	177	142	137	24	565	2230
B	111	99	36	55	455	1078	33	41	78	2	324	275	125	44	520	3276
A+B	140	124	46	68	567	1355	55	70	145	3	403	341	226	55	937	4535
A+C	83	74	27	37	338	855	41	54	121	2	235	189	181	35	792	3064
A+B+C	150	133	49	72	609	1466	62	80	169	3	429	363	258	60	1104	5006
Reduction in 2020 (kt)																
A	104	92	33	50	414	948	55	73	150	2	300	252	233	37	939	3683
B	139	123	45	70	583	1321	76	102	200	3	411	341	312	54	1305	5084
A+B	174	154	56	86	715	1634	95	126	252	3	506	427	391	67	1626	6311
A+C	151	134	49	72	633	1514	79	105	221	3	436	360	336	64	1464	5620
A+B+C	190	168	62	93	785	1820	102	136	276	4	550	464	426	75	1805	6956

5.4.5 Cost-Effectiveness Curves

The data on the cost-effectiveness and applicability of the measures is combined to produce cost-effectiveness curves for 2010 and 2020 (Figure 5.4 and Figure 5.5). The combined effect of measures is shown. As discussed above, for landfill gas recovery and diversion of waste to landfill, a number of measures of differing costs are available to implement each of these options, and due to factors which limit the implementation of the measures it is very unlikely that the full reduction could be achieved by solely deploying the least cost measure. A mix of the measures identified is therefore likely to be required, but due to the very localised factors which may affect the implementation of each measure, insufficient data is currently available to estimate with any degree of certainty the contribution which each measure might make to the total achievable reduction. To reflect this uncertainty, the cost-effectiveness curves show all of the measures within each option not just the most cost-effective measure. The solid line on the graph shows the cost-effectiveness curve if the lowest cost measure from each group of options (improved landfill gas recovery, improved capping, and reduction of biodegradable waste landfilled) is implemented, the dotted lines show the cost-effectiveness curve if the higher cost measures are implemented. An exception is paper recycling which is shown separately; it is assumed that by 2020, half of the paper which is currently landfilled is recycled and that this accounts for half of the reductions achieved through the diversion of 65% of biodegradable waste from landfill required by the landfill directive. A key for the measures in the Figures is given in Table 5.19.

Table 5.19 Cost and Scope of Measures to Reduce Emissions from Landfill

	Measure	Cost ECU/t	2010	2020
A	**Improved landfill gas recovery by:**		**2230**	**3683**
	1 - Direct use of landfill gas	-76		
	2 - Power generation	23		
	3 - Flaring	44		
C	**Improved capping of landfills**		**833**	**1937**
B	**Reduce biodegradable waste to landfill by:**		**1942**	**1336**
	4. - Paper recycling	-2208		
	5 - Composting (turned windrow)	1033		
	6 - Incineration	1423		
	7 - Composting (tunnel)	1794		
	8 - Anaerobic digestion	1858		

5.4.6 Projection of Emissions under a With Measures Scenario

Emissions under a business-as-usual scenario and with the reductions shown in Table 5.16 for the various measures are shown in Figure 5.6 for the EU and on a country by country basis in Table 5.20. With all three sets of measures implemented emissions are significantly reduced, to 36% of 1990 levels by 2010 and 9% of 1990 levels by 2020.

Table 5.18 Additional Reductions from More Stringent Measures in Some Member States

Country	Measure	Assumed Effect	Additional reduction (kt) 2010	2020
Austria	Proportion of organic waste in landfill reduced to 5% by 2004	Organic waste content currently assumed to be 14%; reduced to 5% in 2004 linearly	16	0
Denmark	Landfilling of combustible waste prohibited after January 1997	Organic content of waste to landfill reduced from 14% in 1994 to 2% in 1997 and thereafter	13	3
France	Only wastes with little or no degradable matter landfilled after 2005	Organic content of waste to landfill reduced from 14% in 1994 to 2% in 2005 and thereafter	85	11
Germany	Only mineralised or inert waste landfilled after 2005	Organic content of waste to landfill reduced from 14% in 1994 to 2% in 2005 and thereafter	191	23
Total			304	36

Table 5.20 Projections of Landfill Emissions under 'With Measures Implemented' Scenarios (kt of CH4/year)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15	% of 1990 levels
1990	193	173	71	126	758	1777	102	136	302	4	562	493	472	85	1890	7144	-
1994	199	176	65	96	778	1842	106	139	301	4	542	500	457	75	1921	7203	1%
2000	167	148	55	82	659	1540	93	122	263	4	456	425	400	63	1697	6175	-14%
2005	123	108	40	61	478	1084	73	96	201	3	334	319	309	44	1286	4559	-36%
2010	58	51	19	30	225	507	51	67	138	1	158	154	214	21	877	2571	-64%
2020	19	17	6	10	72	151	11	15	29	0	53	54	47	6	177	667	-91%

Figure 5.4 Cost- Effectiveness and Scope of Measures to Reduce EU Landfill Emissions in 2010

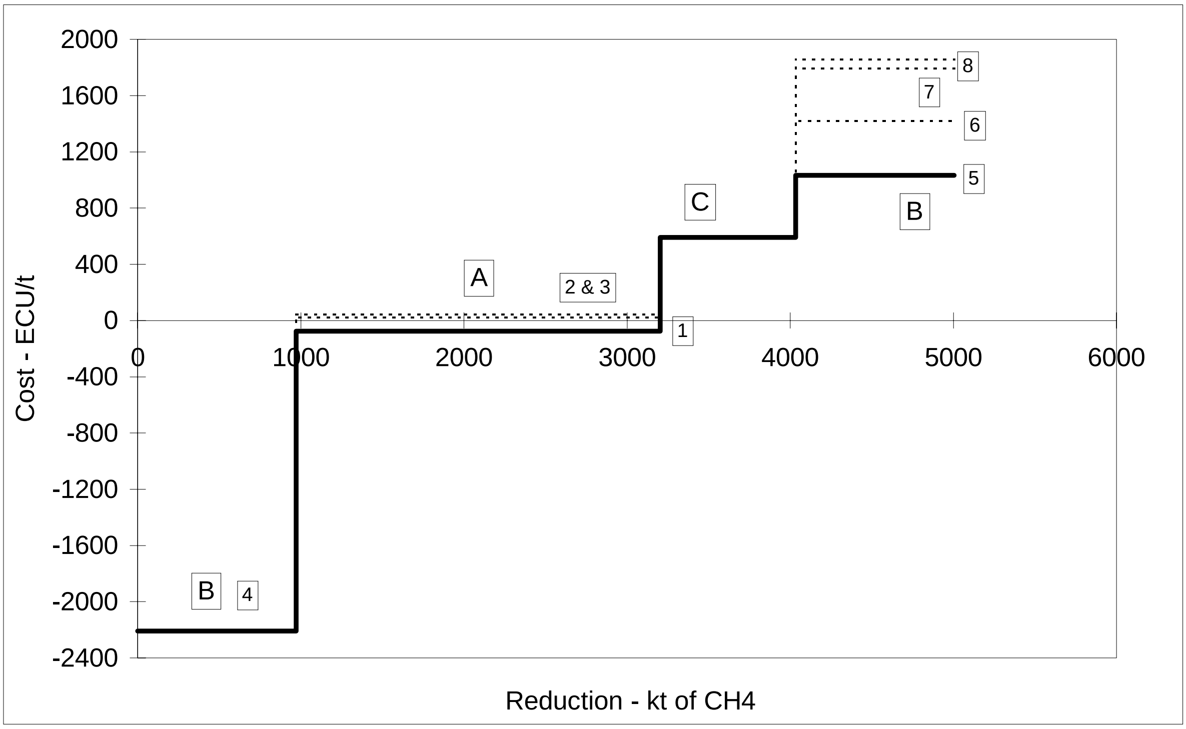


Figure 5.5 Cost- Effectiveness and Scope of Measures to Reduce EU Landfill Emissions in 2020

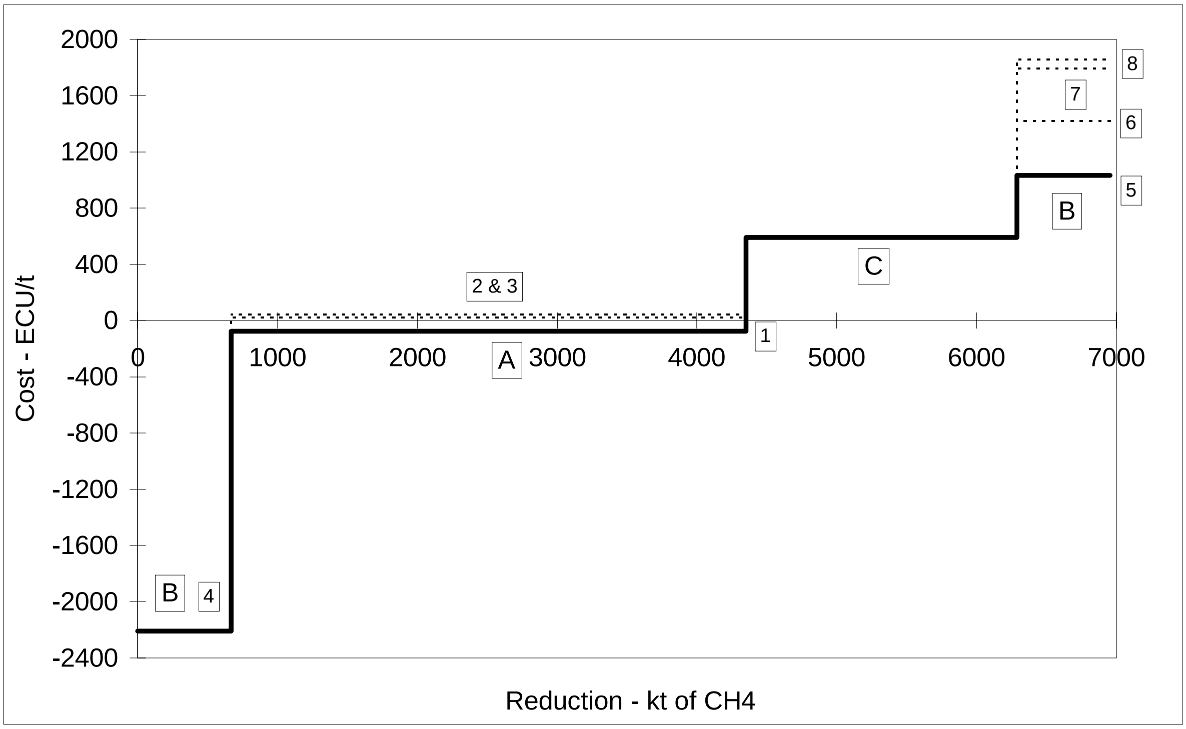
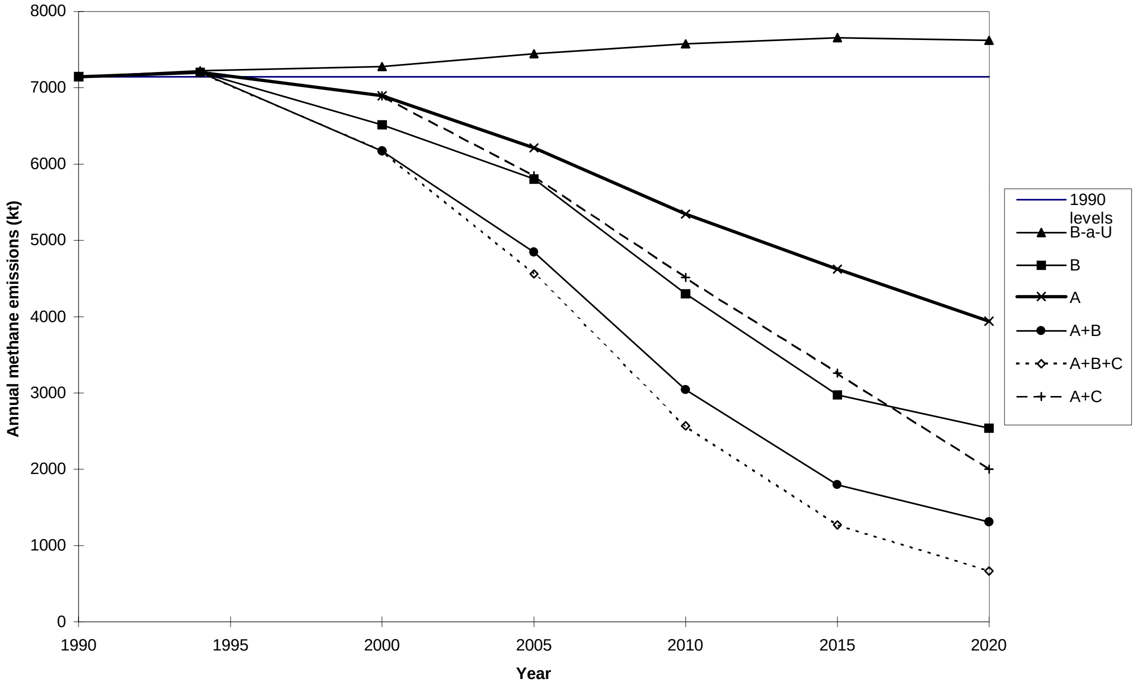


Figure 5.15 Emissions under a B-a-U and with Measures Implemented Scenarios



6. Options to Reduce Methane Emissions from Coal Mining

6.1 METHANE EMISSIONS FROM COAL MINING

6.1.1 Emission mechanisms

The amount of methane generated during coal mining is primarily a function of, coal rank and depth, gas content and mining methods. Coal rank represents the degree of coal formation and depends on the history of the coal seam; high rank coals such as (hard) bituminous coal contain more methane than low rank coals such as lignite. For a given geological setting methane content tends to increase with depth, thus emissions from surface mines are generally lower than those for underground mines.

Methane is emitted when the coal is de-stressed and fractured as part of the mining operation, with further, but reducing, emissions occurring for some time after. The gas which is released, comes not just from the seam being worked, but also from the adjoining strata and seams. The gas generally contains some ethane and higher hydrocarbons (between 2% and 5% is typical) and some coals, for example in Australia, may also contain considerable amounts of carbon dioxide.

In practical mining terms the main release of gas is local to the working place. Measurable emissions also take place during the transportation of the coal from the coal face to the surface; this operation may take an hour or more in large mines. The gas is released into the mine ventilation air, unless specific actions are taken to minimise this. Methane emission has long been recognised as a potential hazard in underground coal mining, since it forms an explosive mixture at concentrations of between 5% and 15% in normal air. Mining engineers therefore developed various methods of predicting and controlling methane emissions, based on safety criteria[2], long before its significance as a greenhouse gas was recognised.

While the major methane emission occurs when coal is first fractured during mining, residual emissions continue for a long time. Thus emissions can continue in abandoned mines, and indeed have resulted in explosions in cellars or other confined spaces in the past.

6.1.2 Mining emissions in the EU

Table 6.1 and Figure 6.1 show the breakdown of methane emissions by country for the EU. Germany produces the most methane from this source (42% of the EU total of 1.98 Mt/year) with Spain (27%), UK (17%) and France (11%) also producing significant emissions. Emissions have already fallen by a third from 1990 levels (2.95 Mt/year) due mainly to the decline of the coal industry (particularly deep mines) in Germany and the UK.

[2] National legislation varies, but most countries require mining to be stopped (or rather electricity supplies to be turned off - which is effectively the same thing in a mechanised mine) at 1% methane concentration, or slightly more. UK legislation, for example, requires electrical isolation at 1.25% and for all men to be withdrawn if the level rises to 2%.

Table 6.1 also shows the annual coal production figures and the emissions of methane per tonne of coal produced for each country. There is a wide variation in the methane emissions per tonne of coal produced. This is due to differences in:

- the ranks of coal mined
- the proportion of open and deep cast mining
- the current extent of collection and utilisation of mine methane
- the release of methane from abandoned mines in countries with historically higher production rates, e.g. France, Italy and the UK.

For example, production figures for Austria, Finland, Greece and Ireland included lignite and, in some cases, peat production.

6.2 MITIGATION OPTIONS

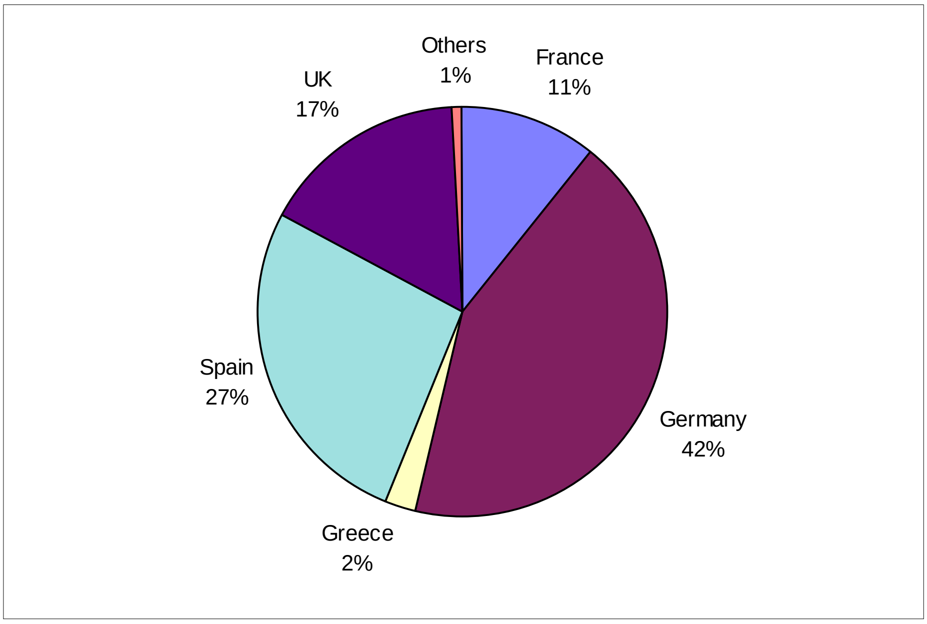
As discussed in Section 6.1, action over mine methane has always been driven primarily by safety concerns. The simplest method of maintaining safe methane levels is to increase the flow of ventilation air, thereby achieving greater dilution. This has limitations however; higher flows mean increased costs, since fan power consumption is proportional to the cube of air quantities. Also high air velocities, particularly at the working face, can cause significant dust problems.

One option, therefore, is to 'drain' at least part of the methane before it enters the mine air stream. If this gas is then brought to the surface, it can potentially be used for steam raising or electricity generation, although in some cases it must be released to atmosphere because the methane content is considered too low for safe utilisation. Methane is explosive at concentrations of between 5% and 15% in atmospheric air. UK legislation therefore prohibits the utilisation of mines-gas containing less than 40% methane, although this figure can be reduced to 30% under certain circumstances. In Germany the limit is lower, at 25%. These figures are considered to provide a reasonable safety margin above the upper explosive limit. It should be noted however that the 5% to 15% range widens as the pressure is increased above atmospheric. This can have implication where it is necessary to compress the gas, such as prior to use in a gas turbine, and can therefore limit the range of end uses[3].
In both Germany and the UK (the two most significant EU coal producers) it was estimated that in 1990 less than one third of the methane emitted was captured before it entered the mine ventilation air. In Germany as much as 70% of this captured gas is currently utilised, but in the UK the figure is nearer 30%. This section considers options, firstly to improve the collection rate for methane and secondly to improve the utilisation rate of the captured methane.

Increasing the utilisation rate cannot be considered independently of improved capture as it is necessary to ensure that the gas collected is of sufficient quality that it can be utilised. A considerable portion of the gas currently collected has to be vented because it does not meet the statutory minimum methane content for utilisation. Above this level, the higher the methane content, the more attractive it is to potential customers. This requires better methane drainage methods and improved pipework systems.

[3] The dilutant in mine methane is mainly air (containing about 20% oxygen). This has safety implications when considering utilisation options. In contrast, the other principal component of landfill gas (see Section 5.1) is carbon dioxide, with the main dilutant being nitrogen.

Figure 6.1 Methane Emissions from Coal Mining in the EU, by Country in 1994



Source: Member States and EU Second National Communications

Table 6.1 Methane Emissions from Coal Mining in the EU (1994)

Country	**Coal Production mtoe/year**[1]	**Methane Emissions kt/year**[2]	**Methane Emissions kt per mtoe coal**
Austria	0.4	0	0
Belgium	0.3	0	0
Denmark	0	6	N/A
Finland	2.1	0	0
France	5.8	213	37
Germany	81.2	850	11
Greece	7.4	48	7
Ireland	1.2	0	0
Italy	0.1	7	110
Luxembourg	0	0	0
Netherlands	0	0	0
Portugal	0.1	2	33
Spain	10.6	525	49
Sweden	0.3	0	0
UK	28.5	327	12
Total	**137.9**	**1,978**	

Sources: [1]IEA, 1997
[2] Second National Communications to the FCCC.

6.2.1 Improved methane capture

6.2.1.1 Current EU methane drainage practice

There are two main methods of underground mining: room-and-pillar and longwall mining. With the **room-and-pillar approach**, coal deposits are mined by cutting a network of 'rooms' or panels into the coal seam and leaving behind 'pillars' of coal to support the roof of the mine. **Longwall mining** involves cutting successive strips of coal from a face, typically 100m to 250m long. The face is accessed by 'roadways' driven at each side of the 'panel' of coal to be extracted. The roof strata is allowed to collapse (or 'cave') behind the face. Usually the face is established as the remote end of the panel and is 'retreated' back towards the main trunk roadways.

Current EU drainage practice is to drill inclined holes into the strata above the seam being mined. These are drilled from one, or occasionally both, of the panel roadways in advance of the face. Gas emission starts as the face approaches and passes beneath the holes. In the case of retreat mining, which is becoming increasingly the norm, the holes then pass into the caved area behind the face, where access is no longer possible. The fact that the drainage is being carried out at a location where the coal is being de-stressed and fractured and the difficulty of regulating, or even isolating, individual holes, means that a considerable amount of air is drawn in and mixed with the methane. The ingress of air that occurs means that the gas, when it reaches the surface, may contain 50% or less of methane. This has obvious cost implications, in that the system must be designed to handle a higher total flow. It may also cause problems at the utilisation stage, as already mentioned above.

6.2.1.2 Other drainage options

It may be possible to drain additional gas by drilling inclined holes into the floor strata, as well as into the roof. One problem with this method, if the seam is wet, is the removal of water from the holes; roof holes, by contrast are naturally self-draining. Floor holes are therefore only used if there is no other option.

Another method, used in the USA, is to drill vertical holes from the surface into the target seam and drain off gas prior to mining. Indeed it is a way of obtaining gas independently of any mining operation. Coal Bed Methane (CBM), as it is called, is extracted in large quantities from the Black Warrior and San Juan Basins in the USA. Because there is minimal opportunity for air ingress, the gas is virtually pure methane and needs only minimal treatment to make it compatible with 'pipeline quality' natural gas.

Unfortunately this system depends on the coal having a high permeability; otherwise the holes have to be very close together, which makes it uneconomic. This has prevented the spread of the technique. While trials are being carried out in both the EU and Eastern Europe, these have had variable success, given the very low permeability of European coals. Its potential for de-gassing EU coals is therefore very limited.

Another approach, which was developed in Germany is to drive a 'superjacent heading'. This is a special roadway some distance above the face and parallel to its run, which then acts as a collection point. If necessary the collection efficiency could be increased by drilling boreholes from this roadway. The problem with this method is the high cost of driving the roadway. Its current relevance to the EU is therefore very limited, given Western European labour costs.

There is, however, a variant of this system which could have considerable potential, which is to replace the heading with a long borehole. This involves drilling an inclined hole from the level of the seam being worked. The hole is then curved until it is running horizontally above the face. Directional drilling of this type is quite feasible using a 'down-hole motor' or other steering method, and holes of over 1000 m have been drilled in this way. The method is not, however, without problems. It requires traversing, at a shallow angle, the strata immediately above the seam being mined. If these strata are strong there is no particular problem; if they are weak, or consist of clays it may be very difficult to drill the hole or to keep it open. This was the experience of a series of trials carried out recently in the UK. Unfortunately such strata are very common in proximity to EU coals. This is in contrast to some other countries, such as Australia and USA, where the overlying strata are often sandstones. More work needs to be done on developing this technique for EU conditions.

6.2.1.3 Underground transmission
Traditionally steel pipework has been used to transmit the gas from the face to the disposal point, with individual lengths being joined by flexible couplings. Flexible couplings are necessary as most countries do not permit welding underground. Also even long-term roadways experience a degree of strata movement, so a degree of flexibility is required. Transport and handling problems mean that individual pipe lengths are short, but the total distance may be several kilometres. The result is that there are a large number of flexible couplings, so even a small leakage at each can have a large cumulative effect. Some improvements can be obtained by using Glassfibre Reinforced Plastic (GRP) or 'plastic' pipes. Their lighter weight means that longer individual lengths can be handled. Also their inherent flexibility means that they are better able to accommodate strata movements. The system is operated under suction because it is preferable to have air leakage into the pipes, rather than risking the potential hazard of methane to leaking out into the mine air.

6.2.1.4 Possible improvements in methane capture
The current methods capture about 30% of the potential methane emissions. It should be possible to increase this to about 50% by improved engineering systems. These would necessarily increase the capture costs. Any increase beyond this is unlikely with current technologies. This view is at variance with figures quoted by some other authors, but is considered to be a realistic target. With improved technologies it could be possible to achieve 70% at some mines. The exact nature of these technologies still has to be defined and there would be obvious additional costs involved. Costs are considered in more detail in Section 6.3.

6.2.1.5 Methane from abandoned mines
In the past many mines were abandoned without any special precautions. Current practice is to seal the tops of the shafts for safety reasons. If the mine has a history of methane emissions it is necessary to provide vents on the shafts. Otherwise the pressure in the mine is likely to build up, leading to an eventual escape of gas in an unsuspected and uncontrolled manner.

6.2.2 Utilisation of mine methane
Mine methane production depends on mine output and mining method. It is therefore likely to vary over time. All projections show that EU coal production will decrease, with further mine closures. The economics of different methane utilisation options are considered below, but the intuitive answer is to go for a system which is simple, flexible and involves the mine in minimum capital costs. There are too many systems, installed in the past, which have worked very successfully; unfortunately the mine closed a few years later. This was the case with several

small spark-ignition engines that were installed at various UK mines. This is where flexibility is important - several of these engines are now enjoying 'second careers' on UK landfill sites.

6.2.2.1 Historical Perspective

The utilisation of mine methane has a surprisingly long history. The earliest recorded example was at Salton Colliery in north west England as long ago as 1730. The gas was brought to the surface through a square wooden pipe and was used to heat specially constructed furnaces. Its more recent application, however, dates from the 1940s. At that time many mines had boilers to generate their own electricity or to drive steam-powered winding engines and ventilation fans. These boilers were typically 'hand-fired fire-tube' type, using low-grade coal for which there was a limited external market. It was fairly easy to adapt them to fire mine gas and if there was a temporary shortage of gas it was quite easy to return to coal firing. This position has now changed; mines generally have electric winders and fans and take their electricity from the public supply. Gas can still, however, have a limited role in providing hot water for showers and space heating for offices and other buildings. In countries with cold winters it can also be used to heat the mine intake air - to prevent freezing conditions in the down-cast shaft.

6.2.2.2 Utilisation at the mine site

These residual uses are not sufficient to use all the gas at mines which practice methane drainage, so other options must be considered. One of these is to generate electricity, and a moderately gassy mine can often become virtually self-sufficient in electrical power. It is normal to retain a connection to the public supply, to cover emergencies or periods of low gas production, and in many cases it is possible to export any surplus power to the public system. There are a number of possible options. These are summarised in Table 6.2 and their relative merits considered below.

Table 6.2 Generation Options

Option	Typical thermal efficiency	Size range MW(per unit)*	Examples of where used
Boiler/steam turbine	20%	10+	not currently used
Spark-ignition reciprocating engine	34%	0.5-2.0	Australia, China, UK
Dual-fuel reciprocating engine	36%	1.0-2.5	UK
Gas turbine - open cycle	22%	1-20	Australia, China, Japan, Germany, UK
Combined cycle (gas turbine/steam)	35%	15+	UK

* Multiple installations are possible, for example 50+ spark-ignition units at Appin colliery, Australia.

Steam turbine

A boiler and steam turbine system is feasible. It would have advantages as boilers are relatively tolerant of varying fuel qualities. However the thermal efficiency would be comparatively low given that the modest steam conditions which would be necessary. This option is not therefore normally used.

Spark-ignition reciprocating engine
Mine gas can be used in a spark ignition engine, either in a naturally-aspirated or turbo-charged form. A current example is two mines in Australia which have a total of ninety four 1 MW engines installed. These have now been operating for over two years. The engines used are modified Caterpillar diesel engines - there being no readily available gasoline engines of this size - and were based on units previously developed for use on landfill gas.

Dual-fuel compression-ignition engine
Mine gas can be used to fuel a compression-ignition engine, but a small amount of diesel oil has to used as a 'pilot fuel' to initiate ignition. The thermal efficiency is marginally higher than for a spark-ignition engine (typically 36% compared with 34%), but against this must be offset the cost of the diesel fuel. The advantage is that it is generally possible to operate as a 'simple' diesel engine in the event of an interruption to the gas supply. This can be useful if a 'secure' electricity supply is required.

Gas turbine
Another option is a gas turbine. One example was the 1.28 MW Kongsberg radial gas turbine which was installed at Point of Ayr colliery in the UK. The machine was a comparatively simple light-weight unit, originally designed for power generation on North Sea oil platforms, and its efficiency was only about 25%. However the overall efficiency was improved by using the waste heat from the exhaust to produce hot water.

A far more sophisticated system was installed at Harworth colliery in the UK. This uses a combined cycle, based on two 4 MW gas turbines. The exhaust heat from these is fed to two waste-heat boilers which supply steam to a single 10 MW steam turbine. The combined output, after deducting parasitic losses, is about 14 MW. In the event of a fall in the gas quality it is possible to fire it directly into the waste-heat boilers. The result is a very flexible system, although one with high capital costs.

6.2.2.3 Flaring
Flaring combusts the gas and is a disposal rather than a utilisation method, reducing methane emissions by 95-99% depending on the flare efficiency. Flaring produces no economic benefit, which is why it has not been done in the past. There may also be legislative barriers to flaring, for example UK legislation is interpreted as prohibiting flaring of mine gas at present.

6.2.2.4 Utilisation off-site
Mine gas has been widely used by other industries; examples include brickworks, potteries and glassworks. These are comparatively energy intensive industries, so they are willing to consider using mine gas if they see an economic benefit. The comparatively low calorific value of mine gas, and the probable requirement to construct a dedicated pipeline, mean that this option is only likely to be economic if the distance involved is less than about 10 km. The inevitable fluctuations in the quality and quantity of mine gas, plus the problems involved in its storage mean that it is advisable to have an alternative fuel source available. In the case of an established user this could be the fuel used before conversion. There are many established mines which are in or close to major conurbations which have a range of other industries. The option is less realistic for a new 'green field site' mine.

In the past mine gas has been used to enrich 'town' gas (made from coal) or to heat coke ovens so that the coke oven gas can be sold as town gas. These applications have, however, been largely overtaken by the displacement of town gas by natural gas.

Electricity generation, using any of the methods described for mine-site generation is another option. An example was a chemical company, part of the Unilever Group, located in north west England. This used mine gas in its boiler plant for a number of years; based on this experience it installed two 1.5 MW spark ignition engines. This enables the site to generate sufficient electricity to meet its base-load requirements.

A more interesting option would be to supply gas, as an auxiliary fuel, to a nearby coal-fired power utility; or even better to an oil- or gas-fired one. The amount of gas produced by a typical EU coal mine, compared to the size of a modern generator 'block', means that it would only serve as an auxiliary fuel. However, for this very reason, it would require only minimal modifications to the boiler. This could be particularly appropriate for a utility which was modifying its coal-fired boilers to overfiring to reduce NO_x emissions, since gas is often used as the overfire fuel.

Other options that have been proposed include upgrading to pipeline quality gas and conversion to methanol. Some authors have shown that these routes can give satisfactory financial returns, but no mining company has actually gone this route to date.

6.2.2.5 Utilisation of mine ventilation air

Even with good methane drainage systems it is likely that more than 50% of the gas will find its way into the mine ventilation air stream. There would be considerable benefits if this gas could be used in some way. One theoretical option would be to use it, instead of normal air, in a power utility or other boilers. If the gas contained 1% methane then it would reduce the fuel requirement by up to 15%. The problem is the difficulties and cost of transporting such large volumes of air, even over very short distances. It would be possible to build a generation plant at the mine. However, to use all the available air, the plant would have to be of several hundred megawatts capacity. This would be far larger than needed to supply the mine, so most of the power would have to be exported.

The two Australian mines referred to previously (Section 6.2.2.2) are using a variant of this system. In addition to using their drained methane as fuel for spark-ignition engines, they take the air required for combustion from the mine exhaust ventilation. Although this is only a small percentage of the total available air.

Another option is based on thermal oxidisation; this involves passing the air through a special heated gravel bed. The 'Vocsidizer' system is an example of this method. The concept was originally devised for disposing of low concentrations of VOCs, where methane was added to achieve the necessary bed temperature. If the system is used with methane alone then there is a net heat release if the methane concentration exceeds about 0.5%. With methane concentrations above this level there is the potential to use the surplus heat for steam generation or other purposes. Indicative capital costs are US$900-2500 per kWe generated. As the system has yet to be proved commercially, in a mine application, it is too soon to estimate likely operating costs.

6.2.2.6 Methane from abandoned mines

The need to drain methane from abandoned mines has already been explained. The prime difference with gas from abandoned mines is that it is likely to have a higher than average methane content and a low oxygen content. This means that it may have an additional utilisation option - mixing with natural gas. One example of this approach is the 'Methamine' project in northern France. When the project started the gas composition was about 51% methane, 47% inerts (mainly nitrogen and carbon dioxide) and less than 1% oxygen. It is expected that the methane content will increase to over 60% after some years' operation.

Gas with this low oxygen content can be safely compressed. While it is not up to normal pipeline gas quality (95% methane) it is possible to mix a small amount into pipeline gas without reducing its calorific value below the contractual level. The economics of this arrangement require there to be a large capacity natural gas pipeline relatively near to the abandoned mine site. The Methamine project meets this requirement in that there are two 400 mm high pressure (60 bar) pipelines located 7 km from the site. These carry about 200,000 m^3/hr under normal conditions, with rather higher flows in the winter. It was decided that injection of 8% to 10% of mine gas would be acceptable. This represents 16,000 to 20,000 m^3/hr. The actual injection rate is controlled by maintaining a consistent Wobbe Index (a measure of the combined effects of specific gravity and calorific value), and therefore a constant quality to the customers.

The capital cost of the project was FF 100,000,000 (at 1987 prices). Initial operating costs were equivalent to 1.6 ECU/GJ, of which 40% was compression costs.

6.3 COST OF OPTIONS

Cost and performance data is available to calculate the cost-effectiveness of five measures aimed to improve the collection and utilisation of methane emissions. These are:

- electricity generation – boiler/steam turbine;
- electricity generation – reciprocating engine (spark-ignition or dual fuel);
- electricity generation – gas turbine (open cycle);
- electricity generation – gas turbine (combined cycle);
- flaring.

The cost-effectiveness of each of these measures can be calculated for three levels of collection efficiency, 30%, 50% and 70%. The cost-effectiveness of the measures, in 1995 ECU per tonne of methane abated, is calculated using the methodology and assumptions described in Section 1 and Appendix 1.

6.3.1 Cost Assumptions

The assumptions used for thermal efficiency, non-recurring and recurring costs for each of the above five measures are shown in Table 6.3. These figures have been derived from a combination of sources, including Bennett *et al* (1995) and in-house data. Note that while the reciprocating engine and the combined cycle gas turbine have similar efficiencies they have different costs. The former has a lower specific capital cost, but a higher specific operating cost.

With respect to collection rates, the following assumptions have been made:

- Collection rates of 30% at no additional cost
- Achieving collection rates of 50% involves additional non-recurring costs equivalent to 1.0 ECU per tonne of coal production, plus annual recurring costs of 5% of the additional non-recurring costs.
- Achieving collection rates of 70% involves a further cost penalty of 1.0 ECU per tonne of coal production, plus additional annual recurring costs of 5% of the non-recurring costs.

Ranges of non-recurring and recurring costs corresponding to collection rates of 30, 50 and 70 per cent are presented in Tables 6.4, 6.5 and 6.6, respectively, assuming gross methane emissions of 20,000 tonnes per year for a typical mine and an income from electricity of 35.78 mECU per kWh. This level of emissions is typical of mines currently operating in Germany and the United Kingdom. Note that the cost-effectiveness of the various options considered here is highly dependent on the assumed unit value of electricity. The range in costs reflects the range in capital and operating costs shown in Table 6.3.

To determine the cost-effectiveness of each measure, it is also assumed that all of the methane emissions collected and utilised are fully combusted. Therefore, annual savings corresponding to collection rates of 30, 50 and 70 per cent are 6,000, 10,000 and 14,000 tonnes of methane.

Table 6.3 Summary of Utilisation Option Efficiencies and Costs

Type	Assumed thermal efficiency	Capital cost (ECU/kW installed)		Operating cost (ECU/kWh)	
		Range	Best estimate	Range	Best estimate
Boiler/steam turbine	20%	500-900	600	0.003-0.005	0.004
Reciprocating engine[a]	35%	600-900	700	0.005-0.008	0.006
Gas turbine (open cycle)	22%	400-700	500	0.002-0.004	0.003
Gas turbine (comb cycle)	35%	600-1600	900	0.003-0.005	0.004

[a]An average value of 35% efficiency has been used for both dual fuel and spark ignition engines

6.3.2 Cost-effectiveness

Table 6.7 summarises the cost-effectiveness of measures to reduce emissions of CH_4 from coal mining, and Table 6.8 presents the data in terms of ECU per tonne of CO_2-equivalent

Table 6.7 indicates that overall the reciprocating engine is the most cost-effective measure for reducing methane emissions from coal mining activities. This is true for 30, 50 and 70 per cent collection rates. The gas turbines (combined and open cycles) however, are also relatively cost-effective at all percentages of methane collected, the former despite the comparatively high capital costs. There is more variation with these latter two options between the high and low estimates though, due to the large range in unit capital costs. Steam boilers, while resulting in operating cost savings, are slightly less cost-effective overall than the three options identified above, due to their relatively low thermal efficiency. Flaring is the least cost-effective option overall, as would be expected, since it effectively represents disposal rather than utilisation, and thus a wasted resource with no economic benefit.

It is also possible to calculate the marginal cost of upgrading a system e.g. from 30% recovery and utilisation to 50% recovery and utilisation, and these costs are shown in Table 6.9 for the most cost effective option, the reciprocating engine.

Table 6.4 Non-recurring and Recurring Costs: 30% CH_4 collection and utilisation (ECU thousand)

	Steam Boiler	Recip Engine	Gas Turbine (open cycle)	Gas Turbine (comb cycle)	Flare
Best Estimate:					
Non-recurring	2,088.0	4,256.0	1,910.0	5,472.0	120.0
Recurring	69.6	182.4	57.3	121.6	12.0
Electricity value	622.6	1,087.7	683.4	1087.7	-
High Estimate:					
Non-recurring	3,132.0	5,472.0	2,674.0	9,728.0	120.0
Recurring	87.0	243.2	76.4	152.0	12.0
Electricity value	622.6	1,087.7	683.4	1087.7	-
Low Estimate:					
Non-recurring	1,740.0	3,648.0	1,528.0	3,648.0	120.0
Recurring	52.2	152.0	38.2	91.2	12.0
Electricity value	622.6	1,087.7	683.4	1087.7	--

Table 6.5 Non-recurring and Recurring Costs: 50% CH_4 collection and utilisation (ECU thousand)

	Steam Boiler	Recip Engine	Gas Turbine (open cycle)	Gas Turbine (comb cycle)	Flare
Best Estimate:					
Non-recurring	5,475.4	9,097.7	5,186.7	11,125.6	2,200.0
Recurring	215.9	404.2	195.6	302.8	120.0
Electricity value	1,036.6	1,813.9	1,140.2	1,813.9	-
High Estimate:					
Non-recurring	7,214.6	11,125.6	6,461.4	18,233.2	2,200.0
Recurring	244.9	505.6	227.5	353.5	120.0
Electricity value	1,036.6	1,813.9	1,140.2	1,813.9	-
Low Estimate:					
Non-recurring	4,897.0	8,083.7	4,549.4	8,083.7	2,200.0
Recurring	186.9	353.5	163.7	252.1	120.0
Electricity value	1,036.6	1,813.9	1,140.2	1,813.9	-

Table 6.6 Non-recurring and Recurring Costs: 70% CH_4 collection and utilisation (ECU thousand)

	Steam Boiler	Recip Engine	Gas Turbine (open cycle)	Gas Turbine (comb cycle)	Flare
Best Estimate:					
Non-recurring	8,867.2	13,937.2	8,461.6	16,776.4	4,280.0
Recurring	362.2	625.9	333.9	483.9	228.0
Electricity value	1,451.24	2,539.7	1,596.4	2,055.7	-
High Estimate:					
Non-recurring	11,300.8	16,776.4	10,246.2	26,713.6	4,280.0
Recurring	402.8	767.8	554.9	378.5	228.0
Electricity value	1,451.24	2,539.7	1,596.4	2,055.7	-
Low Estimate:					
Non-recurring	8,056.0	12,517.6	7,569.3	12,517.6	4,280.0
Recurring	321.7	554.9	289.2	412.9	228.0
Electricity value	1,451.24	2,539.7	1,596.4	2,055.7	-

Table 6.7 Cost-effectiveness of Measures to Reduce Emissions from Coal Mining (ECU per tonne CH_4 abated)

	Steam Boiler	Recip Engine	Gas turbine (open cycle)	Gas turbine (comb cycle)	Flare
30 % collection:					
Best estimate	-55	-78	-69	-70	5
High estimate	-37	-51	-55	-3	-
Low estimate	-63	-92	-77	-101	-
50 % collection:					
Best estimate	-28	-52	-42	-43	44
High estimate	-11	-25	-28	24	-
Low estimate	-36	-65	-51	-74	-
70 % collection:					
Best estimate	-17	-40	-31	-32	61
High estimate	1	-13	-17	35	-
Low estimate	-25	-54	-39	-63	-

Table 6.8 Cost-effectiveness of Measures to Reduce Emissions from Coal Mining (ECU per tonne CO_2-equivalent abated)

	Steam Boiler	Recip Engine	Gas turbine (open cycle)	Gas turbine (comb cycle)	Flare
30 % collection:					
Best estimate	-2.6	-3.7	-3.3	-3.3	0.2
High estimate	-1.8	-2.4	-2.6	-0.1	
Low estimate	-3.0	-4.4	-3.7	-4.8	
50 % collection:					
Best estimate	-1.3	-2.5	-2.0	-2.0	2.1
High estimate	-0.5	-1.2	-1.3	1.1	
Low estimate	-1.7	-3.1	-2.4	-3.5	
70 % collection:					
Best estimate	-0.8	-1.9	-1.5	-1.5	2.9
High estimate	0.0	-0.6	-0.8	1.7	
Low estimate	-1.2	-2.6	-1.9	-3.0	

Table 6.9 Cost-Effectiveness of Upgrading compared to New Installations

Measure		ECU/ t CH4
new plant	30% recovery and utilisation	-78
upgrade	from 30% to 50% recovery and utilisation	-65
upgrade	from 30% to 70% recovery and utilisation	-66.5
new plant	50% recovery and utilisation	-52
upgrade	from 50% to 70% recovery and utilisation	-42
new plant	70% recovery and utilisation	-40

6.4 APPLICABILITY OF MEASURES

6.4.1 Baseline trends

In order to ensure consistency with the other EU projections which are forming the basis for examining trends in CO_2 emissions, future trends in coal production (Table 6.10) are taken from the Energy in Europe 2020 study (pre-Kyoto conventional wisdom scenario) prepared for DGXVII. A comparison of these trends with other predictions of coal production is contained in Appendix 3.

It is clear from Table 6.10 that coal production is expected to decline substantially in the future, with EU production in 2010 only a third of 1990 levels.

6.4.2 Emissions Under A Business As Usual Scenario

Emissions arising from coal production in Table 6.10 have been estimated and are presented in Table 6.11. It is assumed that emissions are directly related to coal production, i.e. that recovery and utilisation rates remain constant (in percentage terms) and that for the major producers, (Germany, UK, France and Spain) the current ratio of deep mine to surface mine production is maintained. The 1990 and 1994 estimates of emissions in Germany include 87 kt from abandoned mines and these emissions are assumed to remain constant in the future.

Table 6.10 Coal Production under the Pre-Kyoto Scenario (mtoe)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
1990	0.6	1.1	0.0	1.5	7.6	125.5	7.1	1.4	0.3	0.0	0.0	0.1	11.7	0.3	53.1	210
1994*	0.4	0.2	0.0	1.9	5.5	85.1	7.6	1.1	0.2	0.0	0.0	0.0	10.4	0.2	36.7	149
2000	0.3	0.0	0.0	2.0	1.3	68.8	7.8	1.2	0.1	0.0	0.0	0.0	9.3	0.0	20.5	111
2005	0.3	0.0	0.0	2.0	0.5	58.6	7.3	1.1	0.0	0.0	0.0	0.0	9.2	0.0	17.8	97
2010	0.3	0.0	0.0	2.0	0.3	48.4	7.0	1.1	0.0	0.0	0.0	0.0	7.5	0.0	12.1	79
2020	0.1	0.0	0.0	2.0	0.0	39.6	6.7	0.9	0.0	0.0	0.0	0.0	3.4	0.0	9.0	62

* Interpolated from data for 1992 and 1995

Table 6.11 Emissions from Coal Mining Under a Business as Usual Scenario (kt of CH_4)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
1990	0	15	3	0	206	1230	43	0	15	0	0	3	613	0	818	2946
1994	0	0	6	0	213	850	48	0	7	0	0	2	525	0	327	1978
2000	0	0	6	0	49	701	50	0	4	0	0	0	469	0	242	1521
2005	0	0	5	0	17	610	46	0	1	0	0	0	467	0	159	1305
2010	0	0	6	0	12	519	44	0	1	0	0	0	381	0	108	1069
2020	0	0	6	0	0	440	42	0	0	0	0	0	173	0	80	741

6.4.3 Existing Policies and Measures

Table 6.12 shows the trends and national policies for reducing methane from coal mining, for those countries with significant mining activities. As already discussed, production is expected to decline so that methane emissions are expected to fall significantly without further measures. Germany and the UK are the only countries reporting specific initiatives to increase the utilisation of mine gas and this is likely to be limited to the introduction and promotion of cost effective measures.

Table 6.12 Existing Policies in EU Member States to Reduce Methane Emissions from Coal Mining

Country	Trends and Policies	Expected change in CH_4 emissions by 2000 as % of 1990 emissions
France	Planned decrease in coal production - no measures planned	-76%
Germany	1. Pit gas recovery has been increased from 70% to 78% since 1990. 2. The Federal Environmental Agency is studying further measures.	-32% without measures -44% with measures
Spain	Not reported.	Not reported
UK	1. Declining coal production expected to continue. 2. Taking cost effective actions to increase the utilisation of methane.	-61%

6.4.4 Applicability of Measures

All of the major coal producers already have some recovery and utilisation of mine methane. For Germany emissions data from the Second National Communication indicates that in 1994, the overall utilisation rate was 36%. For the UK the estimated utilisation and recovery rate is estimated to be 11%; recovery and utilisation rates in Spain are estimated to be lower at 5%. For France, the Second National Communication states that most CH_4 released in former and active French mines is already recovered and used for heating, and that all production is planned to finish by 2010.

The following assumptions are made about the potential for recovery and utilisation for each of these four countries. Assumptions about the measures already installed are based on the premise that lowest cost measures have been installed first.

- **UK and Germany**: A recovery and utilisation rate of 30% could be achieved in all mines, a 50% recovery and utilisation rate at mines equivalent to 70% of coal production and a 70% rate achieved at mines equivalent to 50% of the coal production. In order to achieve it’s current utilisation rate, Germany is assumed to have 30% recovery and utilisation at sites equivalent to 70% of production and 50% recovery and utilisation at mines equivalent to 30% of production, i.e. all mines already have some form of recovery and utilisation. For the UK to achieve its current utilisation rate, it is assumed that there is a recovery and utilisation rate of 30% at mines equivalent to 37% of production.

- **Spain**: A recovery and utilisation rate of 30% could be achieved in 50% of mines. For Spain to achieve its estimated current utilisation rate, it is assumed that there is a recovery and utilisation rate of 30% at mines equivalent to 17% of production.
- **France:** As discussed above a high level of recovery is already reported and due to the planned closure of all mines no further additional recovery is assumed to be implemented.

Table 6.13 sets out how the recovery and utilisation rates discussed above will be met. In some cases it will be possible to improve existing recovery rates and increase generating capacity (termed upgrade in the table); in other cases, it will be necessary to install new generating plant (termed new plant in the table). The reductions which are assumed to be achieved in each country by 2010 and 2020 are presented in Table 6.14. These are based on production, so it is only assumed that measures are implemented in mines which will still be open in 2010 and 2020 respectively. In 2010 reductions from the measures are 21% of emissions under the business as usual scenario.

Table 6.13 Deployment of Measures in Major Coal Producing Countries

Measure		percentage of production capacity measure implemented at		
		Germany	Spain	UK
new plant	30% recovery and utilisation	0%	33%	30%
upgrade	from 30% to 70% recovery and utilisation	40%		37%
new plant	50% recovery and utilisation			20%
upgrade	from 50% to 70% recovery and utilisation	10%		
new plant	70% recovery and utilisation			13%

Table 6.14 Reductions Achieved (kt of CH_4)

Year	Germany	Spain	UK	Total
2010	113	40	68	221
2020	93	18	50	161

6.4.5 Cost-Effectiveness Curves

The data on the cost-effectiveness and applicability of the measures is combined to produce cost-effectiveness curves for 2010 and 2020 (Figures 6.2 and 6.3). It is assumed that the least-cost option, a reciprocating engine is implemented in all cases. A key for the measures in the cost-effectiveness curves is given in Table 6.15.

Figure 6.2 Cost-Effectiveness of Measures to Reduce Emissions from Mining in 2010

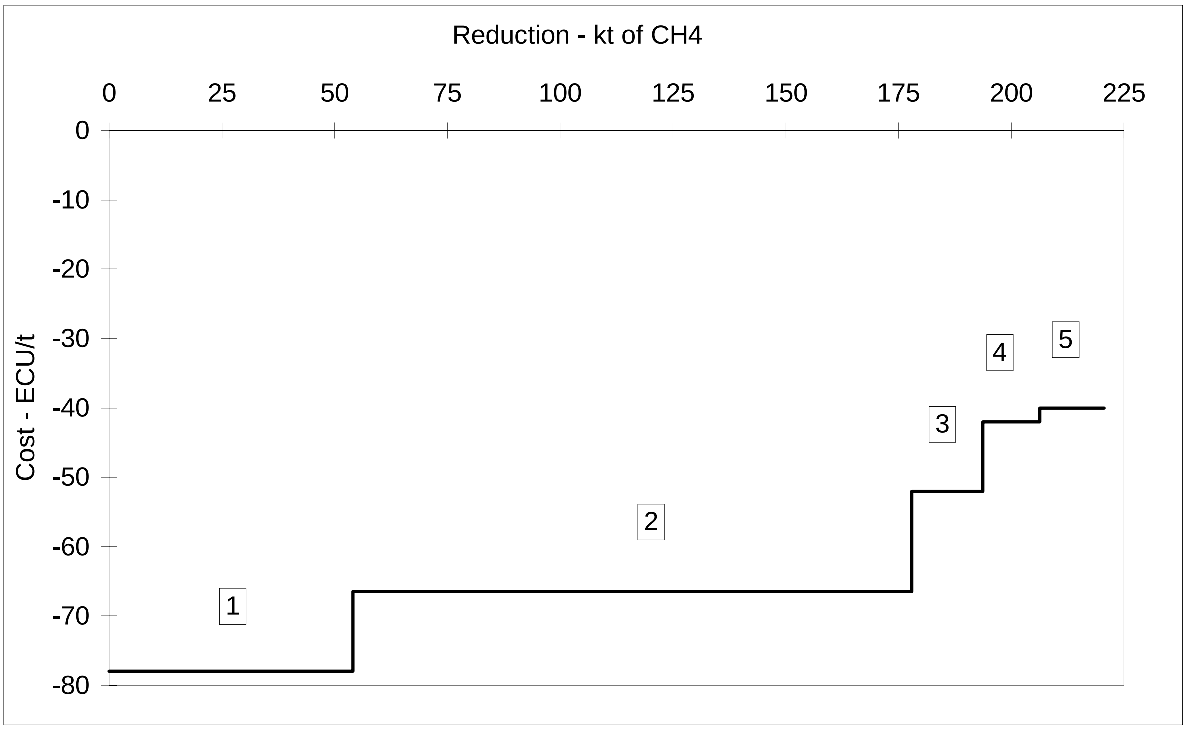


Figure 6.3 Cost-Effectiveness of Measures to Reduce Emissions from Mining in 2020

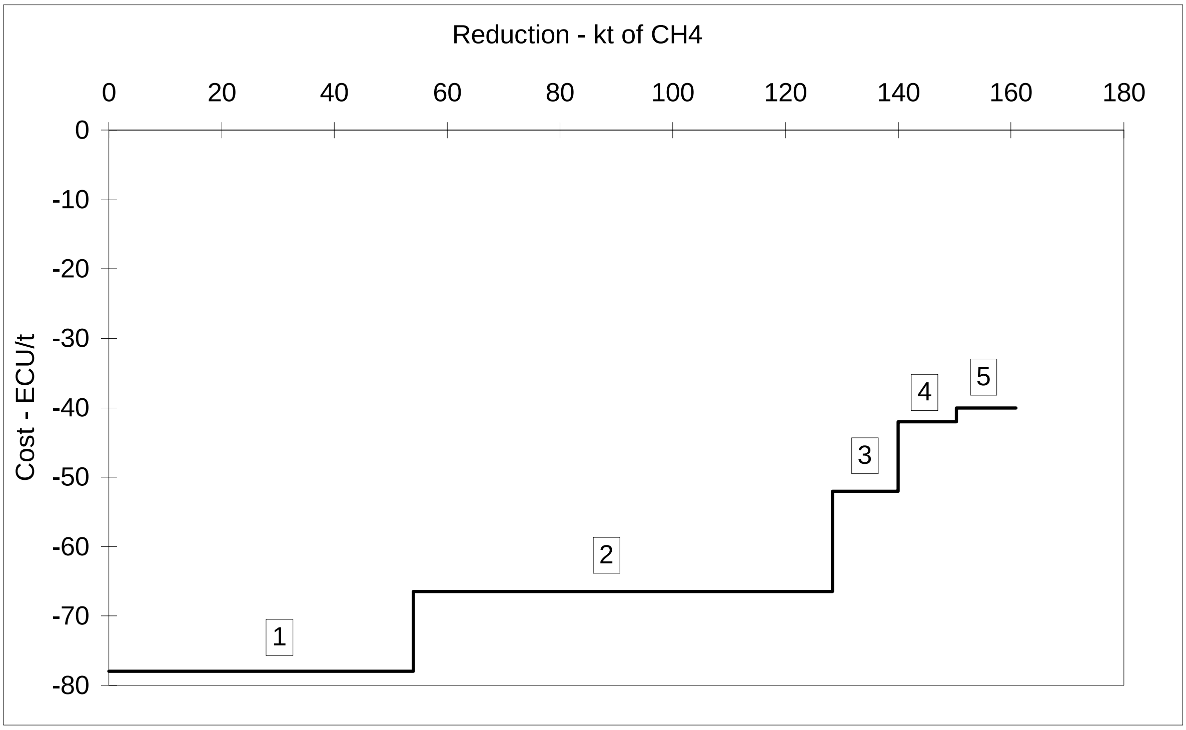


Table 6.15 Cost and Scope of Measures to Reduce Emissions from Mining

Key	Measure	Cost	Scope (kt)	
		ECU/ tonne	2010	2020
1 new plant	30% recovery and utilisation	-78	54	28
2 upgrade	from 30% to 70% recovery and utilisation	-67	124	100
3 new plant	50% recovery and utilisation	-52	16	12
4 upgrade	from 50% to 70% recovery and utilisation	-42	13	10
5 new plant	70% recovery and utilisation	-40	14	11

6.4.6 Projections under a With Measures Scenario

Emissions under a business as usual scenario and with the reductions shown in Table 6.14 ('with measures' scenario) are shown in Figure 6.4 for the EU and on a country by country basis in Table 6.16. Under the with measures scenario mining emissions are predicted to be 29% of 1990 levels by 2010 and 20% of 1990 levels by 2020.

Figure 6.4 Projection of Emissions from Mining under Business-as-Usual and 'With Measures' Scenarios.

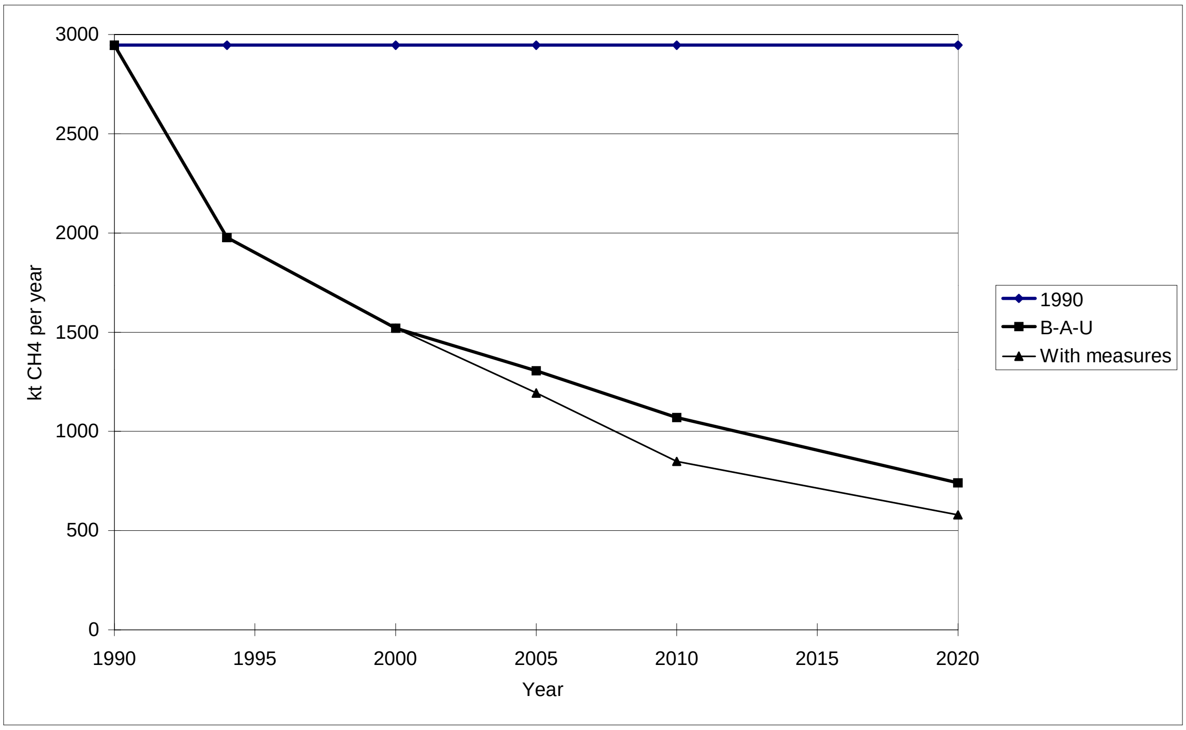


Table 6.16 Emissions from Coal Mining Under a With Measures Scenario (kt of CH_4)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15	Change from 1990 levels
1990	0	15	3	0	206	1230	43	0	15	0	0	3	613	0	818	2946	0%
1994	0	0	6	0	213	850	48	0	7	0	0	2	525	0	327	1978	-33%
2000	0	0	6	0	49	701	50	0	4	0	0	0	469	0	242	1521	-48%
2005	0	0	5	0	17	553	46	0	1	0	0	0	447	0	125	1195	-59%
2010	0	0	6	0	12	405	44	0	1	0	0	0	341	0	40	848	-71%
2020	0	0	6	0	0	347	42	0	0	0	0	0	155	0	30	580	-80%

7. Options to Reduce Methane Emissions from the Oil and Gas Sector

7.1 METHANE EMISSIONS FROM THE OIL AND GAS SECTOR

7.1.1 Oil and Gas Production in the EU

The oil and gas sector includes the exploration and production of on-shore and off-shore oil and gas, and their subsequent processing, transport and distribution. Annual production figures for oil and gas for EU countries are shown in Table 7.1. These show that the UK accounts for about 83% of EU oil production while the major gas producers are the UK (38%), the Netherlands (36%), Italy (10%) and Germany (9%).

Table 7.1 Oil Production, Natural Gas Production and Natural Gas Consumption in the EU by Country in 1995 (IEA, 1995)

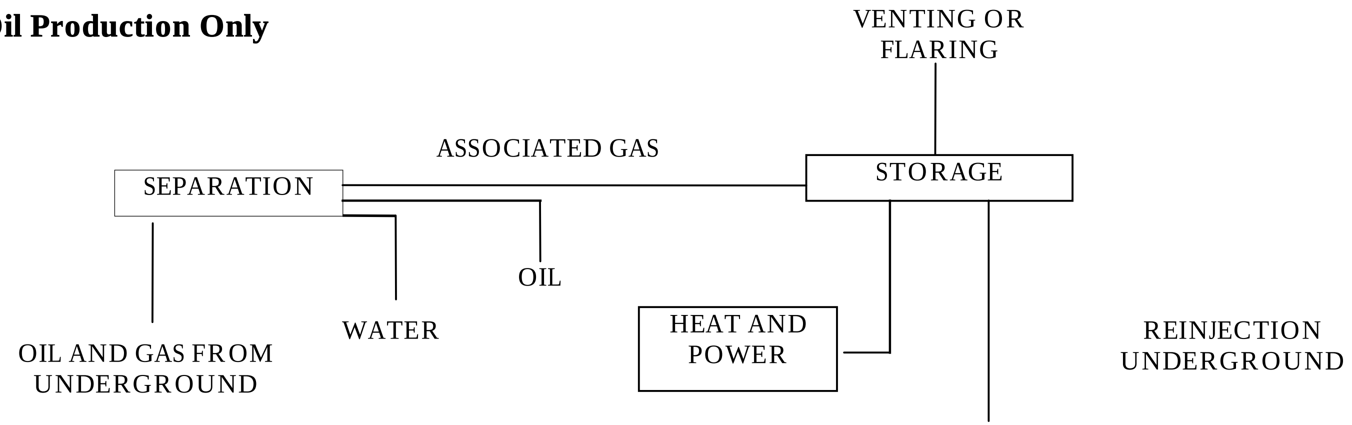
	Oil production Mt/yr	Gas production TJ/yr	Gas consumption TJ/yr
Austria	1.1	59	296
Belgium	0	10	497
Denmark	9.2	216	144
Finland	0	0	137
France	2.9	123	1377
Germany	3.9	680	3038
Greece	0.5	2	2
Ireland	0	104	104
Italy	5.0	769	2079
Luxembourg	0	0	25
Netherlands	3.5	2812	1605
Portugal	0	0	0
Spain	0.8	18	359
Sweden	0	0	31
UK	130.5	2957	3011
Total	**157.4**	**7750**	**12705**

Source: IEA, 1995

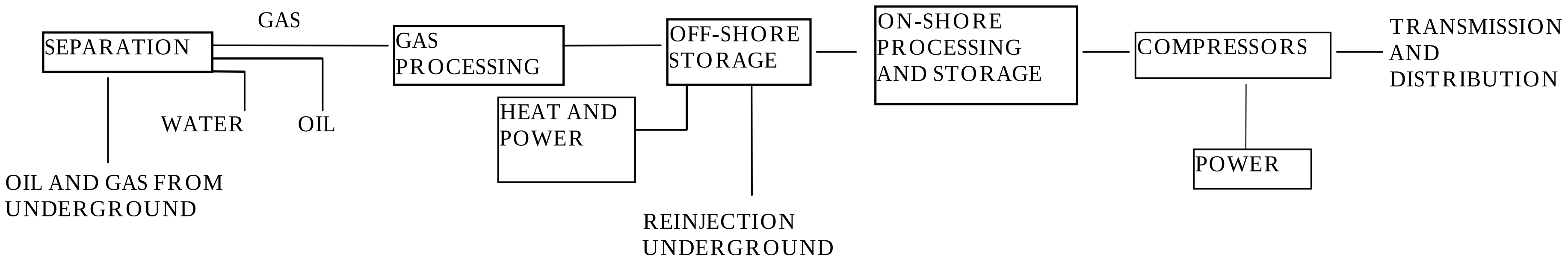
The processes involved in oil and gas production are shown schematically in Figure 7.1. Most oil production in the EU is from off-shore oil fields in the North Sea. All oil fields have associated gas which comes out of the ground with the oil. If there is sufficient gas to sell commercially then the gas will be processed and sent by pipeline to an on-shore terminal for further processing. If there is not sufficient gas for this to be economic, or if the field is too remote, a proportion of the associated gas will be used on the platform and the remainder will be vented or flared. There are two potential uses for the gas on the platform, as a fuel for power generation and for reinjection into the oil field as an alternative to water reinjection.

Figure 7.1 Main Processes in Oil and Gas Production

(a) Oil Production Only



(b) Oil and Gas Production



Off-shore or on-shore gas production involves most of the same processes as oil production, although the initial separation process is a little different. In gas production, the gas is normally associated with higher hydrocarbons which are liquid at normal temperature and pressure. This liquid (known as condensate) is removed from the gas in the initial separation process and is transported, processed and sold as a separate product.

7.1.2 Methane Emission Sources

As discussed in Section 2.2, total methane emissions from the oil and gas sector are about 1.56 Mt/yr, which account for approximately 9% of methane emissions in the EU (from Member States and EC Second Communications under the FCCC). This represents only about 5% of global methane emissions from this sector, with much greater contributions from the former USSR, the Middle East and North America (ECOFYS, 1997).

The major sources of methane emissions in this sector in the EU are:
- the venting and flaring of associated gas;
- venting and flaring of off-gases (emissions from processing, such as leakage past valves) from oil and gas production;
- the loss of gas in the gas transmission and distribution network through compressors and fugitive emissions from pipework.

A number of studies (e.g. ECOFYS (1997), De Jager and Blok (1993) and Woodhill (1994)), have sought to quantify the emissions from the different processes involved in oil and gas production and to identify the methane emission reduction potential from each. These studies have broken down the emission sources in different ways. Some have looked at each stage of the process - exploration, production, transmission and distribution - while others have used functional sources such as power generation, compressors, process vents etc.

For the purposes of this study we have followed the approach used in the ECOFYS (1997) report which breaks emission sources into a number of types of sources to which different mitigation options can be applied, as follows:

- **Exploration** - emissions from well testing and cleaning.
- **Unused associated gas** - if not used for power generation or exported for sale, this gas is vented or flared at the point of oil production.
- **Process vents** - vented gas released from the production process.
- **Process flares** - emissions resulting from incompletely burnt flared gas released from the production process.
- **Maintenance** - releases during routine maintenance in production and distribution processes, caused by the need to isolate and depressurise a part of the system.
- **Power generation** - exhaust gas releases from the use of natural gas in power generation for the production and distribution process.
- **Compressors** - releases from seal losses, passing valve emissions, start-up and shut-down of machinery, and indirectly through compressor energy requirements.
- **Pneumatic devices** - some valves are natural gas operated and can therefore cause methane emissions.

- **System upsets** - system upsets may be caused by pipe breakage or pressure surges leading to losses at pressure release valves.
- **Fugitive emissions** - these occur predominantly in the transmission and distribution system, from ageing pipe networks.

It is important to note that many of these categories, including compressors, maintenance and fugitive emissions cut across the divisions between production, transmission and distribution.

7.1.3 Methane Emissions from the Oil and Gas Sector in the EU

Two sources of data for emissions of methane from the oil and gas sector across the EU have been used for this study: individual countries' Second National Communications to the FCCC and a report by ECOFYS (1997) for the IEA Greenhouse Gas R&D Programme. The National Communications to the FCCC provide more accurate data for total methane emissions in each country (Figure 7.2) but the ECOFYS report gives a useful breakdown of emissions by source as well as country (see Table 7.2). We have therefore used the FCCC country totals and assumed that these are divided between sources in the proportions given by the ECOFYS report. Table 7.3 and Figure 7.3 show the breakdown of EU methane emissions by country and by source given by this method.

Figure 7.2 EU Methane Emissions from the Oil and Gas Sectors, by Country (1990) (Total = 1.56 Mt)

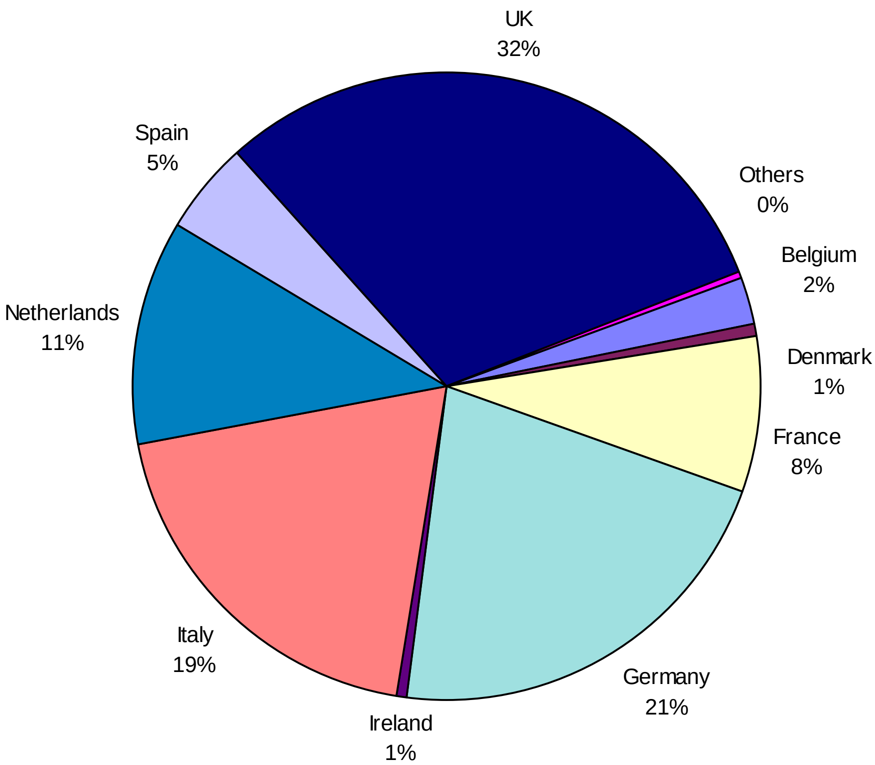


Figure 7.2 shows that there are five key emitting countries in this sector: the UK (32% of EU emissions), Italy (19%), Germany (21%), the Netherlands (11%) and France (8%). As expected, methane emissions from exploration and production sources (e.g. unassociated gas and process vents and flares) are greatest from those countries which produce the most oil and gas (see Table 7.1). Similarly, the major gas consumers produce the most fugitive emissions.

Table 7.2 Disaggregation of Methane Emissions from the Oil and Gas Sectors(kt) as given in ECOFYS (1997)

	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15	% of total
Exploration																0	0
Associated gas vents			4		3	5					8		4		8	32	2
Associated gas flares			2		2						1		2		9	16	1
Process vents						24					54				56	134	8
Process flares											1				11	12	1
Maintenance	1	1			7	6		1	9		4		2			31	2
Power generation	1	2			9			1	13		2		3		1	32	2
Compressors	1	2		1	11	9		1	15		3		3			46	3
Pneumatic devices											1					1	0
System upsets	1	2			9			1	14				3			30	2
Fugitive emissions	51	38	2	1	269	194		7	285	2	75		30		398	1352	79
Other / Not defined						7					1				20	28	2
Total	55	45	8	2	310	245	0	11	336	2	150	0	47	0	503	1714	100

Source: IEA Greenhouse Gas R&D Programme Report: Methane Emissions from the Oil and Gas industry, ECOFYS

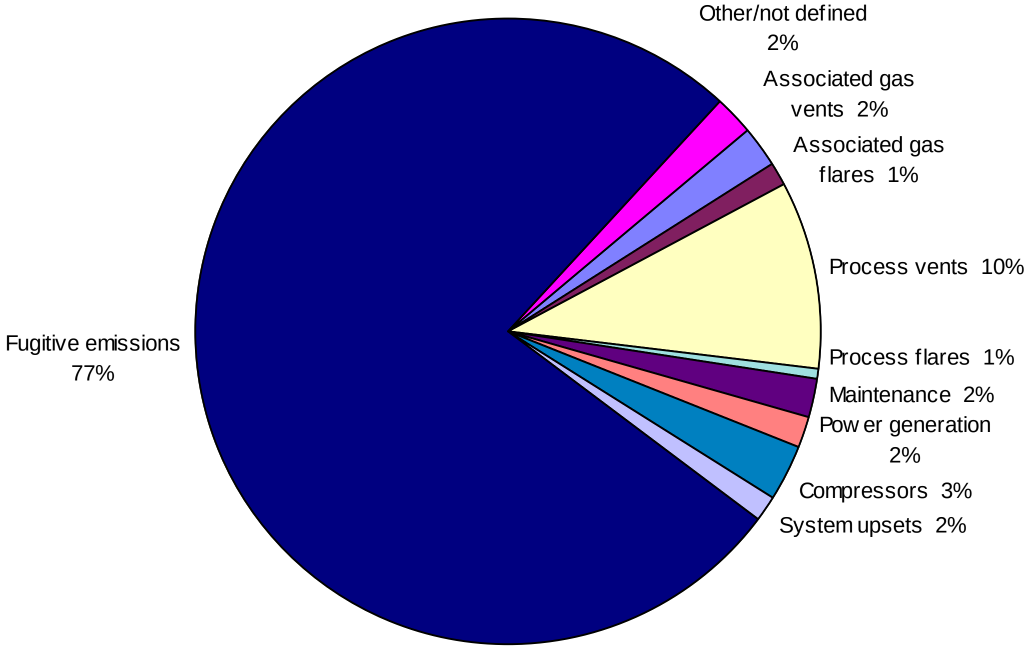
Table 7.3 Disaggregation of Methane Emissions Data for the Oil and Gas Sectors (kt) - Best Estimate for 1990

	Austria	Belgium	Denmark	Finland	France	Germany	Greece	Ireland	Italy	Luxembourg	Netherlands	Portugal	Spain	Sweden	UK	Total	% of total
Exploration																	0.0
Associated gas vents			5		1	7					10		6		8	36	1.8
Associated gas flares			2		1						1		3		9	16	0.9
Process vents						33					64				53	150	7.4
Process flares											1				10	12	0.7
Maintenance		1			3	8		1	8		5		3			29	1.8
Power generation		2			4			1	12		2		5		1	26	1.9
Compressors		2			4	12		1	14		4		5			41	2.7
Pneumatic devices											1					1	0.1
System upsets		2			4			1	13				5			24	1.8
Fugitive emissions	4	33	2		109	264		6	258	2	89		47		380	1194	79.3
Other / Not defined						10					1				19	30	1.5
Total	4	39	9	0	126	333	0	10	304	2	179	0	74	0	480	1560	100

Sources: *IEA Greenhouse Gas R&D Programme Report: Methane Emissions from the Oil and Gas industry, ECOFYS (for breakdown by source type)*
Second National Communications to the FCCC (for country totals)

Figure 7.3 shows that fugitive emissions account for the largest proportion (77%) of all methane emissions in this sector, with process vents being the next biggest source. Emissions from exploration and from pneumatic devices are negligible and so these sources are not considered in the discussion of mitigation options which follows. This would not necessarily be true for other parts of the world where exploration practices are different and where pneumatic devices more commonly use natural gas to provide the hydraulic pressure.

Figure 7.3 EU Methane Emissions from the Oil and Gas Sectors, by source type (1990) (Total 1.56 Mt)



7.2 MITIGATION OPTIONS

As explained above, the emissions sources have been sub-divided into various types, to which different mitigation options apply. This section discusses the possible mitigation options for each source type in terms of their potential effectiveness, their applicability in the EU and the likely costs of implementation, where available. Further discussion of costs and the cost-effectiveness of alternative mitigation options are presented in Section 7.3, and further analysis of the applicability for EU countries is given in Section 7.4.

7.2.1 Unused associated gas

Associated gas is gas which occurs in an oil field, and is a by-product of the extraction of oil. Part of this gas is used for on-site energy requirements or reinjection into the oil field to help maintain the formation pressure, but some goes unused, being either flared or vented. In 1990 the amount of associated gas vented or flared was about 5% of world production (Barns and Edmonds, 1990).

World-wide, it is estimated that a reduction of 50% of the emissions from venting could be achieved in the medium term through the combination of flaring and utilisation of associated gas, with the possibility of an 80% reduction in the longer term (Baudino and Volski, 1991). However, the scope for emissions reduction is likely to be much lower in the EU where producers are already encouraged to utilise associated gas wherever possible, and to flare rather than vent where it is not.

7.2.1.1 Reinjection instead of venting or flaring

To achieve as high a recovery factor as possible in offshore oil fields, reservoir pressures must not be allowed to fall too low as oil and gas are removed. Sea water is therefore pumped into the water-soaked rocks beneath the oil zone in volumes equal to the sub-surface volume of the liquids (oil and gas) produced. Associated gas separated from oil on the platform may also be compressed and injected into the reservoir rocks to maintain pressure. Water and gas injection can improve recovery of oil from less than 15% to more than 50% (Geological Museum/UKOOA). In very deep fields, reinjection of associated gas can also help to improve the yield of condensate, a valuable light oil which exists as gas vapour in the reservoir.

The decision on whether to reinject associated gas, and how much gas to inject, will be taken by the oil producer on a case-by-case basis. This decision will be influenced by costs associated with reinjection and the likely benefits in terms of improved yields. In practice the benefits will usually outweigh the costs in oil fields where the geology is suited to gas reinjection, and so there is little scope for methane emissions reduction by introducing gas reinjection in other oil fields.

7.2.1.2 Flaring instead of venting

Venting involves releasing the gas into the atmosphere, resulting in significant methane emissions. Flaring involves burning this gas, and therefore reducing methane emissions by 95-99% depending on flare efficiency. It should be noted that although flaring reduces emissions of methane, it increases emissions of carbon dioxide. However, the overall global warming potential (GWP) is reduced because of methane's higher GWP.

The applicability of this option will depend on the quantity and composition of the associated gas, the availability of an existing flare system and the location and stability of the platform. This is discussed further in Section 7.2.2.6, below. In practice, most off-shore platforms will already have flare systems, as it has been encouraged for North Sea facilities for some years. Indeed, all Norwegian oil platforms must have flare systems by law.

7.2.1.3 Utilisation of associated gas

An alternative to flaring or venting this associated gas is to use it for domestic consumption or to convert it to liquid natural gas (LNG), or to electricity, for transportation and potentially for export.

In the EU many offshore installations have an existing infrastructure for the transportation and utilisation of associated gas, in the form of a pipeline to an on-shore terminal. Those which do not are generally remote or produce little associated gas and therefore there may be limited additional scope for this measure.

It is possible to use associated gas to meet power and heat requirements on an off-shore platform. However, this is already the established practice on EU facilities and so it cannot be considered as a mitigation option.

The LNG conversion option is most suited to on-shore oil production where there is significant associated gas and no nearby gas pipeline, such as Nigeria. However, there has been recent interest in the production of LNG at some of the more remote platforms in the North Sea, where gas pipelines are not technically or economically feasible (Davies, 1998). The only costs available for LNG conversion come from a Nigerian LNG project which commenced operation in 1996 (ECOFYS, 1997).

The off-shore use of associated gas to generate electricity for export has also been considered (Davies, 1998) but there is no reported application at present. This option may be more attractive than it has been in the past because new smaller gas turbine based generators are now readily available and because electricity cables may be easier to lay on sea beds than gas pipelines. There are no costs available from the literature for this option.

7.2.1.4 Improving flare efficiency

There may be scope to improve the efficiency of combustion in the flare by promoting turbulence. This is discussed further in Section 7.2.2.7, below.

7.2.2 Process Vents and Process Flares

Most off-gases from oil and gas production processes are collected and either vented to the atmosphere or flared (burned). The remaining emissions from production come from incomplete combustion in power generation and from fugitive emissions, i.e. leakages of gas directly to the atmosphere from pipelines, compressor seals etc.

The main emission reduction options for process vents and flares are discussed in the following paragraphs. These options fall into two categories - reducing the volume of gas which comes off at the various stages of production and reducing the percentage of methane from this gas which is emitted to the atmosphere.

There are five main options for reducing the amount of methane gas which must be vented or flared:

- minimising strip gas in glycol dehydration;
- recompressing process emissions;
- recovering and utilising process emissions as a fuel gas;
- reducing passing valve emissions;
- reducing purge gas streams.

There are four options for reducing methane emissions from those gases which must be emitted from process vents and flares:

- flaring instead of venting;
- improving flare efficiencies;
- reducing pilot gas flowrates;
- improving pilot ignition systems.

7.2.2.1 Minimising strip gas in glycol dehydration

Natural gas is often used as a strip gas in the glycol dehydration process during gas drying. This means that it is added to the product gas which has been dissolved in glycol, and it then comes off with the product gas when it is regenerated from the glycol. The use of strip gas improves the efficiency of the glycol regeneration process.

It is possible to minimise or even eliminate methane emissions from strip gas by changing the design and/or operation of this process. Options include reducing the amount of strip gas used (without compromising gas dryness or glycol quality), increasing the temperature at which the glycol is regenerated and using alternative stripping gases such as off-gases from the condensate separation process. ECOFYS (1997) suggests that these options could reduce total emissions from process vents and flares by as much as 20% world-wide, although the figure is likely to be lower for the EU where measures have already been taken to reduce emissions from gas processing.

7.2.2.2 Recompressing process emissions

Process emissions which would otherwise be vented or flared can instead be recompressed and recombined with the dried natural gas stream. This does not necessarily contribute to a reduction in methane emissions since the emissions from compression, processing and transport may be of a similar order to those from a flare. However, it would reduce carbon dioxide emissions and improve overall use of resources.

It is estimated that recompression could reduce emissions from process vents and flares by as much as 75% under favourable local conditions. It is most effective when the process emissions are released at higher pressures.

7.2.2.3 Recovery and utilisation of emissions as a fuel gas

Recovery and utilisation of process emissions as a fuel gas is an option if the quality of the gas is sufficiently high. Variations in quality over time pose problems for gas turbines, therefore gas engines may be used as they are less sensitive to these changes. Both gas turbines and gas engines are demonstrated technologies on- and off-shore.

7.2.2.4 Reduction of passing valve emissions

Pressure safety valves are used to release excess gas, and block-valves are used to shut off a certain part of the process and subsequently depressurise it, such as for maintenance. Both types of valves are connected to the vent or flare system. Leakage, known as passing valve emissions, can occur as a result of these valves becoming worn or fouled. It has been estimated that about 30% of vented emissions are as a result of passing valve emissions (ECOFYS, 1997). Reducing passing valve emissions is possible through strict control and maintenance programmes to reduce leakage, and they can, in theory, be reduced to a negligible amount.

7.2.2.5 Reduction of purge gas streams

Purge gas is normally applied in vent and flare systems to prevent air from entering the system. As the amount used is often unnecessarily high, reductions in methane emissions from this process can be achieved by reducing the amount of purge gas used. Reduced flows can be obtained by installing restriction orifices or flow meters, however there is a safety issue in reducing purge gases. This option is applicable to 5% of emissions from this sub-sector, with an efficiency of 100%. A second option is to use an alternative purge gas, such as nitrogen. This is only applicable to process vents, where the energy content of the off-gas may become too low for them to be flared.

7.2.2.6 Flaring instead of venting

Flaring instead of venting of off-gases can be used to reduce methane emissions by about 95 to 99% depending on the flare efficiency. There are significant safety issues to be considered, especially if this option is to be retrofitted on an off-shore installation. Off-shore the stability of the platform is an issue as the flare needs to be a safe distance from the main part of the platform, and this may be an obstacle to implementation. A further aspect of this option is that there are no positive economic side effects, unlike in the case of reducing leakage, which is associated with recovery and/or reuse of the gas. In spite of these difficulties, nearly all off-shore oil platforms and most off-shore gas facilities in the EU have flare systems installed.

De Jager and Blok (1993) estimate that installing a flare will cost about 0.5-1.0 M Dfl (1.1-2.3 MECU (1995)) with additional operation and maintenance costs of 3% of this investment, based on Dutch costs. Woodhill (1994) gives an estimate for the capital cost of a flare system at an on-shore gas terminal (about £2M or 1.6 MECU(1995)) and a higher estimate for an off-shore flare (about £5M or 4 MECU(1995)). The Dutch data is used for the cost-effectiveness analysis in Section 7.3 of this report.

7.2.2.7 Improvements in flare efficiencies

It may also be possible to improve combustion efficiency in the flare by promoting turbulence. This would reduce methane emissions and emissions of other products of incomplete combustion (soot, carbon monoxide etc.). Flare efficiencies of 95-99% are typical (Woodhill, 1994) with EU flare efficiencies likely to be at the upper end of this range, with limited scope for improvement.

7.2.2.8 Reducing pilot gas flowrates

Further improvements in flaring of the off-gases can reduce methane emissions, such as reduced pilot gas flow rates, which are often at rates higher than is strictly necessary (to prevent the pilot from going out). With respect to total emissions from process vents and flares this could result in an emission reduction of about 1%.

7.2.2.9 Improving pilot ignition systems

Improvements to flare ignition systems through applying flame-out detection could result in emission reduction by another 1%, or 50% of the emission from each pilot.

7.2.3 Maintenance Emissions

Emissions during routine maintenance are caused when parts of pipeline or pieces of equipment are depressurised and flushed with air before maintenance begins. In most cases these off-gases are vented. Emissions can be reduced either through recompression of the emissions using a portable compressor unit, and re-routing them through the system. This can reduce the specific emissions by up to 80%, and the emissions from all maintenance by 20%.

7.2.4 Energy Requirements

Energy is used in oil and gas production for compressors, pumps and other auxiliary equipment, and for compressors used in gas transmission and distribution. This energy is usually supplied by burning some of the process gas in a reciprocating engine or a gas turbine. Emissions can be reduced by replacing reciprocating engines with gas turbines and by improving inspection and maintenance procedures for power generation equipment.

7.2.4.1 Replacing gas engines with gas turbines

The use of gas turbines instead of reciprocating engines will reduce methane emissions because combustion is normally more complete in a turbine. However, turbines generally need a more constant fuel quality and therefore cannot always be applied. It is also possible that increased emissions of carbon dioxide due to the lower efficiency of the gas turbine will negate any methane savings, but this will depend on the relative efficiencies of the turbine and the engine it replaces. US-EPA (1993) estimates that this option is applicable to 10% of emissions from the energy requirements sector, and reduces methane emissions by 90%.

7.2.4.2 Improving maintenance of power generation equipment

Better design, monitoring and maintenance of engines and turbines can lead to emission savings of several percent (US-EPA, 1993) due to increased combustion efficiency and reduced fugitive emissions.

7.2.5 Compressors

Options related to the use of compressors relate to both the compression equipment and also the engines or gas turbines used to power these machines. There is therefore some overlap between the options considered here for compressors and those which apply more generally to energy requirement (Section 7.2.4).

The main options which relate to the compression equipment and its operation are:

- no flushing at start-up;
- electrical start-up;
- improved compressor inspection and maintenance programmes;
- improved valve sealing at compressor stations.

7.2.5.1 No flushing at start-up

Start-up procedures for compressor units and associated power generators can be altered so that flushing of the compressors and engines is not necessary. There are no costs incurred by this measure, but there are profits related to the gas not emitted which is available for onward sale. It has been estimated that 7% of flushing emissions could be saved as a result of this option.

7.2.5.2 Electrical start-up

Electrical start-up instead of the use of a gas expander can be applied as a retrofit option. This will reduce energy requirement emissions by 3% but at a considerable marginal cost. Costs for the use of this option in new installations are zero.

7.2.5.3 Improving inspection and maintenance

Inspection and maintenance programmes to prevent leakage at compressor seals and valves can reduce emissions by up to 70%.

7.2.6 System upsets

System upsets may be caused by pipe breakage or pressure surges which cause pressure relief valves to open. This is a rare occurrence, with a frequency much lower than the frequency of maintenance depressurisations and hence the emissions from this source are just 1.8% of methane emissions from the oil and gas sector.

System upsets can be reduced by using automatic shut-off valves, which detect pressure surges. They can also be reduced by implementing an adequate centralised administration system of the location of pipes, and a system of checks in order to prevent accidental breakage during digging or construction work. Such a system could reduce emissions from this source by 80%.

7.2.7 Fugitive Emissions

Fugitive emissions account for the majority (79%) of emissions from the oil and gas sector. These emissions can be reduced through various leak detection and repair programmes. Two specific options are to replace the old grey cast iron distribution networks and to increase the frequency of leak control.

Grey cast iron networks date from when town gas was used, which was a wetter gas. The network includes hemp and lead joints which leak if they dry out. The change to natural gas therefore resulted in more leakage. Newer distribution networks are constructed of polyethylene (PE) or polyvinylchloride (PVC) which far fewer problems with leaky joints. Replacement of the cast iron network is an expensive option, for example in the Netherlands costs are about 2500 $/yr per Mg CH_4/yr (ECOFYS, 1997), and is generally carried out because of safety implications/requirements.

The alternative option is to improve inspection and maintenance programmes to increase the frequency of leak detection and repair. This can be done by directly measuring gas concentrations along the network or by monitoring system pressures in possible risk areas. An estimate of costs for this is given by ECOFYS(1997). In the Netherlands the gas distribution pipelines in general are controlled every four years, unless a leakage rate higher than 3 leaks per km was found during the last inspection. A doubling of the control frequency could result in an emission reduction of 50%.

7.2.8 Other emission sources

Woodhill (1994) identifies a number of additional methane emission sources:

- tanker loading, both on-shore and offshore;
- on-shore storage tanks;
- gas utilisation for other combustion, such as gas-fired heaters.

None of these is a large source of methane emissions and there is little information available on mitigation options. These sources are therefore not considered further in this report.

7.3 COST OF OPTIONS

7.3.1 Cost Assumptions

Cost and performance data is available to calculate the cost-effectiveness of a total of 17 of the mitigation measures described above. The non-recurring and recurring costs of these measures, their assumed effectiveness, and the applicability to a particular emission source is shown in Table 7.4. Most of this data is drawn from ECOFYS (1997) which used information from a number of different reports to assess availability and cost of different options on a world-wide basis. There are expected to be considerable differences between the EU and the rest of the world, and these are commented on in Table 7.4.

Table 7.4 Summary of costs and effectiveness of options

Source	Option	Effectiveness	Applicability	Investment Cost (US$/ton CH_4)	Operating Cost (US$/ton CH_4)	Cost Saving (US$/ton CH_4)	Original Data Source	Source of Cost Data and Applicability to EU
Associated gas								
7.2.1.2	Flaring rather than venting	97	40	3000	120	0	[1]	Netherlands data - applicable to EU.
7.2.1.3	Associated gas (vented) use (LNG export)	98	0.5	35000	1400	5000	[3]	Data for on-shore LNG production in Nigeria. May not be applicable to off-shore production in the EU.
7.2.1.3	Associated gas (flared) use (LNG export)	98	45	35000	1400	5000	[3]	Data for on-shore LNG production in Nigeria. May not be applicable to off-shore production in the EU.
7.2.1.3	Associated gas (vented) use (domestic use)	98	50	60	50	200	[4]	Costs based on use of associated gas in India, so probably not fully applicable to EU.
7.2.1.3	Associated gas (flared) use (domestic use)	98	45	60	50	200	[4]	Costs based on use of associated gas in India, so probably not fully applicable to EU.
Process vents and flares								
7.2.2	Increased gas utilisation (offshore)	25	80	625	6	150	[1]	Netherlands data - applicable to EU.
7.2.2	Further Increased gas utilisation (offshore)	50	80	1250	13	150	[1]	Netherlands data - applicable to EU.
7.2.2.6	Offshore flaring instead of venting	15	80	3750	112	0	[1]	Netherlands data - applicable to EU.
Maintenance								
7.2.3	Recompression of gas during pipeline maintenance	80	25	0	44	233	[5]	Netherlands data - applicable to EU.

Table 7.4 Summary of costs and effectiveness of options (continued)

Source	Option	Effectiveness	Applicability	Investment Cost (US$/ton CH_4)	Operating Cost (US$/ton CH_4)	Cost Saving (US$/ton CH_4)	Original Data Source	Source of Cost Data and Applicability to EU
Power generation								
7.2.4.2	Inspection and Maintenance programmes	70	20	0	18	235	[4]	US data. Should be similar for EU.
7.4.2.1	Use of gas turbines instead of reciprocating engines	90	10	1500	150	235	[4]	US data. Should be similar for EU.
Compressors								
7.2.5.1	No flushing at start up	100	7	0	0	141	[5]	Netherlands data - applicable to EU.
7.2.5.2	Electrical start up (in new installations)	100	7	0	0	141	[5]	Netherlands data - applicable to EU.
7.2.5.2	Electrical start up (retrofit)	100	3	18000	360	141	[5]	Netherlands data - applicable to EU.
7.2.5.3	Inspection and Maintenance programmes	70	85	0	18	235	[4]	US data. Should be similar for EU.
System upsets								
7.2.6	Prevention of system upsets through I&M programmes etc.	80	90	0	2000	300	[3]	Probably Netherlands data although original source is unascribed.
Fugitive emissions								
7.2.7	Replacement of grey cast iron pipe network	97	75	40000	0	305	[1]	Netherlands data - applicable to EU.
7.2.7	Doubling the frequency of leak controls (I&M1)	50	25	0	1800	300	[1]	Netherlands data - applicable to EU.
7.2.7	Instigate inspection and maintenance (I&M2)	70	5	0	76	342	[4]	US data.

Source: IEA Greenhouse Gas R&D Programme Report: Methane Emissions from the Oil and Gas Industry, ECOFYS

* Original data sources: [1] De Jager and Blok (1993); [2] Woodhill (1994); [3] ECOFYS - own data or unascribed; [4] US-EPA (1993); [5] Coors et al. (1994)

There are negligible recorded emissions of methane from exploration and from pneumatic devices in the EU, therefore no mitigation measures have been considered for these sources. There is also no cost data available for the options of reducing passing valve emissions or reducing purge gas streams, although these options may offer the potential for significant methane emissions savings.

Mitigation options for emissions from process vents and flares have been grouped into two options representing two stages in increased utilisation of this gas, and these are in turn divided into situations where the gas was previously vented and flared. The first stage involves improved process control and minor system adaptations and the second stage involves greater capital investment to maximise reductions from glycol dehydration, utilise process off-gases in power generation and re-compress and re-use process emissions.

7.3.2 Cost-effectiveness

As the cost data shown in Table 7.4 is already expressed in terms of tonnes of methane abated, it can be used directly to determine the cost-effectiveness of each measure. Table 7.5 summarises the cost-effectiveness of each of the measures listed in Table 7.4 in 1995 ECU per tonne of methane abated.

Table 7.5 Cost-effectiveness of Measures to Reduce CH_4 Emissions

Source	Option	ECU per tonne CH_4 Abated	ECU per tonne CO_2 Abated
7.2.1.2	Flaring rather than venting	377	18
7.2.1.3	Associated gas (vented or flared) use (LNG export)	-258	-12
7.2.1.3	Associated gas (vented or flared) use (for onward sale)	-116	-6
7.2.2	Increased gas utilization (offshore)	-57	-3
7.2.2	Further Increased gas utilization (offshore)	7	>1
7.2.2.6	Offshore flaring instead of venting	441	21
7.2.3	Recompression of gas during pipeline maintenance	-153	-7
7.2.4.2	Inspection and Maintenance programme (power generation)	-174	-8
7.2.4.1	Use of gas turbines instead of reciprocating engines	54	3
7.2.5.1	No flushing at start up	-113	-5
7.2.5.2	Electrical start up (in new installations)	-113	-5
7.2.5.2	Electrical start up (retrofit)	1,646	78
7.2.5.3	Inspection and Maintenance programme (compressors)	-174	-8
7.2.6	Prevention of system upsets through I and M programme	1,364	65
7.2.7	Replacement of grey cast iron pipe network	2,378	113
7.2.7	I and M Programme 1 - double leak control frequency	1,203	57
7.2.7	I and M Programme 2 - instigate programme	-213	-10

Table 7.5 shows that there is some considerable variation in cost-effectiveness between the various different measures to reduce methane emissions from gas production and distribution.

As expected, those measures which result in the utilisation of the methane work are more cost-effective overall than those which simply involve its disposal (i.e. flaring). In practice it is likely that such measures will already be built into most EU oil and gas production facilities where possible, and so the scope for further implementation is likely to be limited.

A number of the options represent maintenance and infrastructure improvements which would, as a secondary consequence, result in the reduction of methane emissions. The overall methane reduction associated with such measures is relatively low in relation to the capital and net operating costs, therefore making these measures appear not so cost-effective. However, as mentioned above, such measures would be more likely to be implemented for other reasons. Examples include the replacement of grey cast iron pipe network for safety reasons, and the prevention of system upsets through an improved I and M programme.

7.4 APPLICABILITY OF MEASURES

7.4.1 Baseline Trends

In order to ensure consistency with the other EU projections which are forming the basis for examining trends in CO_2 emissions, future trends in oil and gas production and consumption (Table 7.6) are taken from the Energy in Europe 2020 study (pre-Kyoto conventional wisdom scenario) prepared for DGXVII.

Table 7.6 shows that gas consumption is predicted to double by 2010, with slower growth thereafter; about two thirds of this increased consumption in 2010 is for power generation. Oil production rises slightly and then declines, falling to just below 1990 levels by 2010; gas production is about 30% greater by 2010, but then begins to decline.

7.4.2 Emissions Under A Business As Usual Scenario

Emissions arising from the oil and gas production and gas consumption projections in Table 7.6 have been estimated and are presented in Table 7.7. Using the breakdown set out in Table 7.3, changes in emissions relating to oil and gas production were estimated based on projections of oil and gas production; and changes in emissions associated with gas consumption (apart from fugitive emissions) were estimated on the basis of gas consumption. For fugitive emissions the increase in gas consumption by the non-power sector was assumed to represent new customers who require new distribution networks. These were assumed to be of polyethylene (PE) or polyvinylchloride (PVC) pipelines, and to have a lower leakage rate than the existing network. It was assumed that no measures are implemented.

Under the business as usual scenario, emissions rise by 10% by 2010 and then fall slightly by 2020.

Table 7.6 Gas and Oil Production under the Pre-Kyoto Scenario (mtoe)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
Gas consumption																
1990	5	8	2	2	25	55	0	2	39	0.43	31	0	5	1	47	222
1992	5	9	2	2	28	57	0	2	41	0.48	33	0	6	1	50	238
1995	6	11	3	3	30	69	0	2	44	0.55	34	0	8	1	65	276
2000	6	14	4	5	35	71	2	3	53	0.66	44	2	13	3	77	333
2005	7	16	4	5	38	87	3	4	63	0.67	48	3	18	5	84	385
2010	7	16	4	6	42	103	4	4	69	0.68	53	4	23	6	98	440
2015	8	18	4	7	47	105	4	4	72	0.68	56	5	25	6	101	461
2020	9	19	4	7	52	112	4	4	76	0.68	58	5	27	7	103	489
Gas production																
1990	1.2		6.1		3.5	4.3	0.8		4.7		4.0		0.8		92.1	117.5
1992	1.2		7.9		3.4	3.6	0.7		4.5		3.4		1.1		95.9	121.6
1995	1.1		9.3		3.0	3.2	0.5		5.3		3.5		0.8		131.4	158.0
2000	1.0		7.7		1.8	3.0	0.5		5.5		3.5		1.0		102.9	127.0
2005	0.9		5.6		1.3	1.7	0.3		5.1		3.2		1.0		103.1	122.1
2010	0.8		3.5		0.1	0.3	0.0		4.4		2.9		1.0		102.4	115.4
2015	0.4		3.5		0.1	0.3	0.0		4.1		2.3		1.0		92.1	103.8
2020	0.0		3.5		0.0	0.4	0.0		3.8		1.8		1.0		79.2	89.6
Oil production																
1990	1.1		2.7		2.4	13.5	0.1	1.9	14.0		54.6		1.3		40.9	132.7
1992	1.2		3.6		2.8	13.7	0.1	1.9	14.7		62.0		1.1		45.6	146.7
1995	1.3		4.7		2.8	14.8	0.0	2.2	16.3		60.4		0.4		58.4	161.3
2000	0.9		5.4		1.1	14.0	0.0	1.0	17.5		70.0		0.3		68.0	178.2
2005	0.9		5.6		0.6	13.6	0.0	1.0	16.3		67.5		0.2		68.0	173.6
2010	0.9		5.1		0.0	13.3	0.0	1.0	15.0		65.0		0.0		73.4	173.6
2015	0.7		4.9		0.0	10.5	0.0	0.5	14.0		58.5		0.0		69.0	158.1
2020	0.5		5.3		0.0	7.1	0.0	0.0	13.0		52.0		0.0		60.0	137.9

Table 7.7 Emissions from the Oil and Gas Sector Under a Business as Usual Scenario (kt of CH_4)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15	Change from 1990
1990	4	39	9	2	126	333	0	10	304	2	179	0	74	2	480	1564	
1994	5	45	11	2	121	320	0	11	349	2	168	0	93	2	484	1612	3%
2000	5	49	11	3	125	319	1	12	368	2	182	0	116	3	484	1678	7%
2005	5	49	10	3	125	322	1	13	378	2	181	0	128	3	486	1706	9%
2010	5	49	8	3	126	322	1	14	384	2	179	0	141	3	489	1725	10%
2020	5	51	8	3	131	309	1	14	393	2	165	0	152	2	467	1702	9%

7.4.3 Existing Policies and Measures
Table 7.8 summarises the trends and national policies for reducing methane from the oil and gas sector as described in the countries' Second National Communications. Of those countries with significant emissions from gas networks, many are upgrading pipelines and improving maintenance procedures. Some of these measures, particularly pipeline replacement are driven primarily by safety concerns.

The Netherlands and the UK, both major oil and gas producers, are working with their respective trade associations to reduce emissions from off-shore exploration and production activities. The UK Offshore Operators Association (UKOOA) published environmental guidelines in 1995 on reducing all emissions to air, producing atmospheric emissions inventories and initiating management systems, auditing and training. An agreement between the Netherlands Oil and Gas Exploration and Production Association (NOGEPA) and the Dutch Government is expected to lead to a reduction of methane from on- and off-shore oil and gas production of 30% in the Netherlands by the year 2000 compared to 1990. Much of this reduction is expected to come from the increased utilisation of associated gas which would otherwise be vented.

7.4.4 Scope of Measures
The scope of the measures believed to be relevant to the EU has been estimated using the effectiveness (in abating methane) and applicability factors shown in Table 7.9, which also gives the reduction achieved by each measure in 2010 and 2020 (from emissions under the business as usual scenario). It is assumed that all measures are fully implemented by 2010; the applicability factor refers to the percentage of emissions within that source category which the measure is applicable to.

7.4.5 Cost-Effectiveness Curves
The data on the cost-effectiveness and applicability of the measures is combined to produce cost-effectiveness curves for 2010 (Figure 7.4) In order to show the detail of the lower portion of the graph, a more detailed version showing the lower cost measures is given in Figure 7.5. A key for the measures in the cost-effectiveness curves is given in Table 7.10. The reductions obtained in 2020 are very similar, as all measures are assumed to be implemented by 2010. It should be noted that despite the high cost of pipeline replacement as an option for CH_4 abatement in this sector, this measure is being implemented (to some extent) in most Member States where cast iron pipes form part of the distribution system.

7.4.6 Projections under a With Measures Scenario
Emissions under a business as usual scenario and with the reductions shown in Table 7.9 ('with measures' scenario) are shown in Figure 7.6 for the EU and on a country by country basis in Table 7.11. Under the with measures scenario oil and gas sector emissions are predicted to be 46% of 1990 levels by 2010 and 45% of 1990 levels by 2020.

Table 7.8 Existing Policies in EU Member States to Reduce Methane Emissions from Oil and Gas

Country	Trends and Policies	Expected change in CH_4 emissions by 2000 as % of 1990 emissions
Austria	Switching supply of natural gas from old Soviet republics to other sources employing more modern production plants and distribution networks.	No estimate given.
Belgium	No measures reported.	-10%
Denmark	No measures required.	+22%
Finland	No measures required.	Emissions expected to be negligible.
France	1. GDF is replacing the old distribution network and improving operating practices. 2. Rapid growth in consumption of natural gas.	-23% with proposed measures.
Germany	Modernisation of pipeline networks to reduce leakage.	-19% without additional measures.
Greece	No measures required.	No estimate given.
Ireland	Gas network replacement at a rate of about 3% per year.	+17%
Italy	Maintenance and replacement of old network.	Report not available.
Luxembourg	No measures reported.	No estimate given.
Netherlands	1. Leakage reduction by replacement of gas networks and improved maintenance. 2. Environmental agreement with Netherlands Oil and Gas Exploration and Production Association.	-13%
Portugal	Increase likely due to introduction of natural gas for power generation and domestic fuel in 1997	No estimate given..
Spain	Not reported.	No estimate given.
Sweden	No measures required.	No estimate given.
UK	1. Transco has introduced a leakage control strategy, primarily involving pipe replacement. 2. UK Offshore Operators Association (UKOOA) published environmental guidelines in 1995 on cost effective measures.	-23% for gas distribution. .

Source: EU and National Communications to the FCCC.

Table 7.9 Reductions in EU Methane Emissions from Measures in the Oil and Gas Sector

Measure	Effect-iveness	Applic-ability	Reduction (kt) in 2010	Reduction (kt) in 2020
Flaring rather than venting (associated gas)	97%	40%	12	10
Increase gas utilisation	80%	25%	28	21
Further increase utilisation	80%	25%	28	21
Flaring instead of venting (offshore process vents)	80%	15%	17	13
Recompression of gas during pipeline maintenance	80%	25%	11	12
Inspection and Maintenance programme (power generation)	90%	20%	10	11
Use of gas turbines instead of reciprocating engines	70%	10%	4	4
Compressors - no flushing at start up	100%	7%	6	6
Compressors - Electrical start up (in new installations)	100%	7%	6	6
Compressors - Electrical start up (retrofit)	100%	3%	2	3
Compressors - Inspection and Maintenance programme (compressors)	70%	71%	40	45
Prevention of system upsets through I and M programme	80%	90%	38	43
Replacement of grey cast iron pipe network	97%	50%	600	600
Double leak control frequency for pipelines	50%	25%	156	156
Instigate inspection and maintenance programme	70%	5%	44	44
Total			1003	998

Figure 7.4 Cost Effectiveness Curve for All Measures in the Oil and Gas Sector

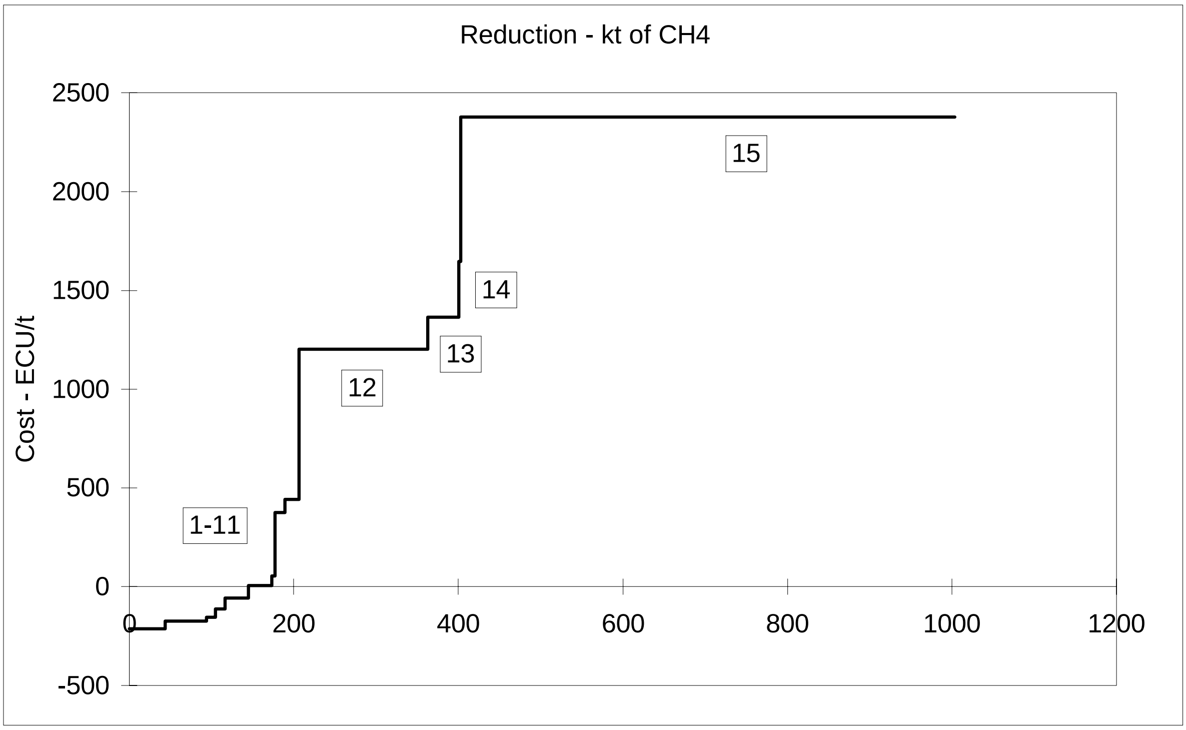


Figure 7.5 Cost-Effectiveness Curve for Measures costing less than 500 ECU/t CH_4 in the Oil and Gas Sector

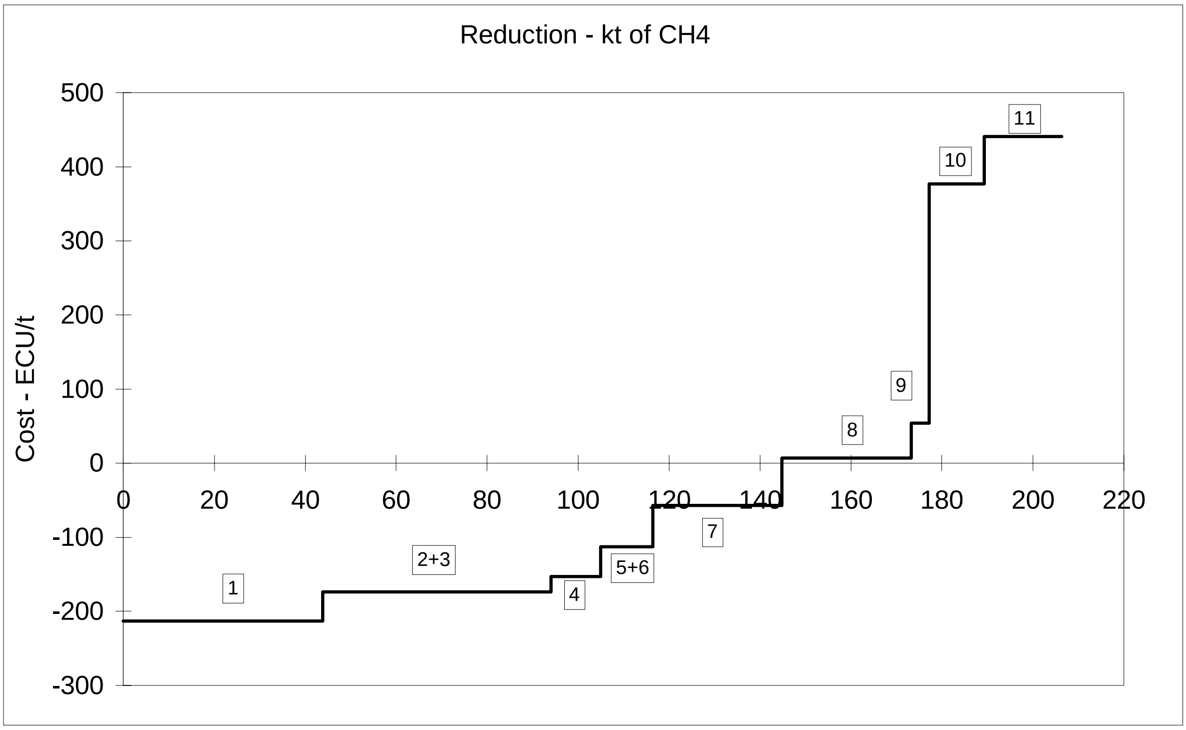


Table 7.10 Cost and Scope of Measures to Reduce Emissions from the Oil and Gas Sector

Key	Measure	Reduction in 2010 (kt)	Cost ECU/t
1	Instigate inspection and maintenance programme	44	-213
2	Inspection and Maintenance programme (power generation)	10	-174
3	Compressors - Inspection and Maintenance programme (compressors)	40	-174
4	Recompression of gas during pipeline maintenance	11	-153
5	Compressors - no flushing at start up	6	-113
6	Compressors - Electrical start up (in new installations)	6	-113
7	Increase gas utilisation	28	-57
8	Further increase utilisation	28	7
9	Use of gas turbines instead of reciprocating engines	4	54
10	Flaring rather than venting (associated gas)	12	377
11	Flaring instead of venting (offshore process vents)	17	441
12	Double leak control frequency for pipelines	156	1203
13	Prevention of system upsets through I and M programme	38	1364
14	Compressors - Electrical start up (retrofit)	2	1646
15	Replacement of grey cast iron pipe network	600	2378

Figure 7.6 Projection of Emissions from the Oil and Gas Sector under Business-as-Usual and 'With Measures' Scenarios.

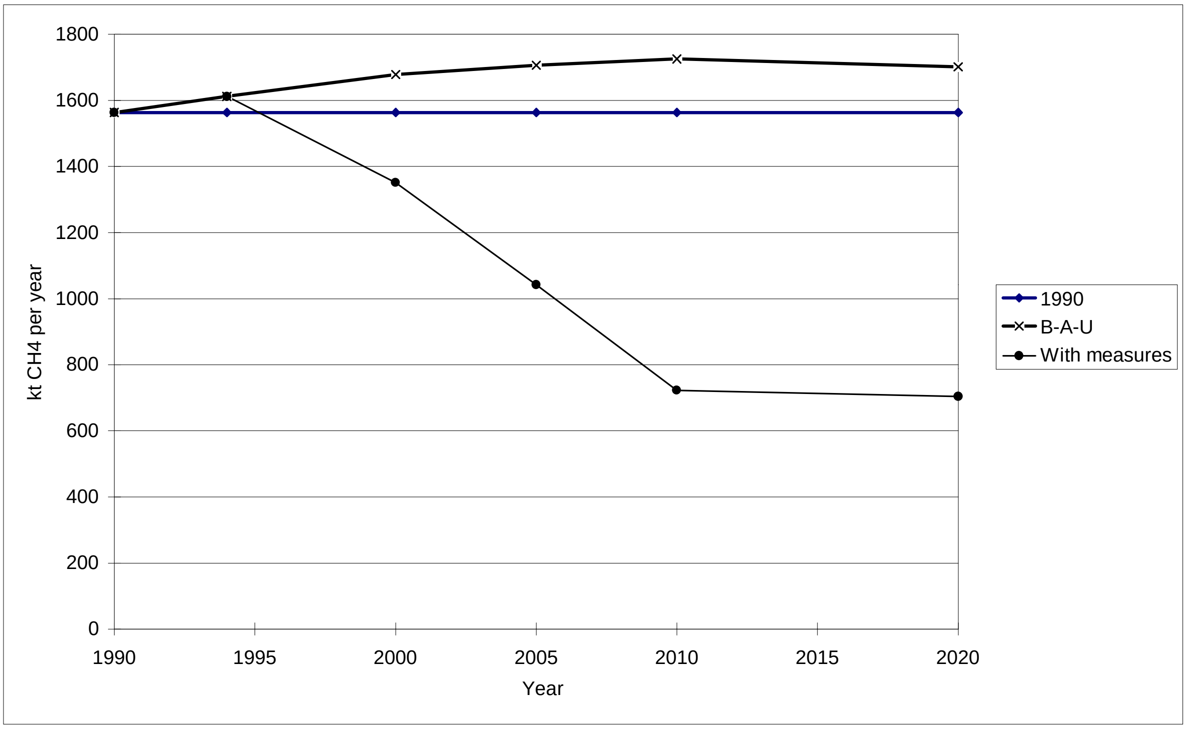


Table 7.11 Emissions from the Oil and Gas Sector Under a With Measures Scenario (kt of CH_4)

Year	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15	Change from 1990 levels
1990	4	39	9	2	126	333	0	10	304	2	179	0	74	2	480	1564	0%
1994	5	45	11	2	121	320	0	11	349	2	168	0	93	2	484	1612	3%
2000	4	39	10	2	99	254	1	10	293	1	149	0	95	2	391	1351	-14%
2005	3	29	7	2	74	193	1	8	224	1	116	0	82	2	301	1043	-33%
2010	2	19	5	1	48	129	0	6	151	1	82	0	66	2	210	722	-54%
2020	2	20	5	1	51	121	0	6	155	1	75	0	72	1	194	705	-55%

8. Emissions from Minor Sources

8.1 OTHER SOURCES OF CH_4 EMISSIONS

In 1990, other sources accounted for 11% of total emissions:

- ***Fuel combustion*** accounted for 3.5 % of EU CH_4 emissions. Of this, over two thirds was from stationary sources and just under one third from transport. CH_4 is produced in small quantities from fuel combustion due to incomplete combustion of hydrocarbons in the fuel. In large efficient combustion facilities such as power stations and large industrial plant, the emission rates are very low. Rates are higher in smaller combustion sources, and highest from small stoves and open fires in the residential sector. In the transport sector, emissions are a function of the methane content of the motor fuel, the amount of hydrocarbons passing unburnt through the engine, the engine type and any post-combustion controls.
- ***Waste water*** treatment accounted for 3.3 % of emissions. These arise from the handling of waste water (from industry and municipal sewerage systems) and treatment of the resulting sludges.
- Emissions from ***'other' agriculture*** (i.e. excluding enteric fermentation and livestock manures) were 1.9% of 1990 emissions. This includes emissions from rice cultivation (in some Southern European Countries) where organic material in flooded rice fields anaerobically decomposes, producing methane which escapes to the atmosphere, and emissions from the burning of agricultural residues.
- ***Land use change activities*** (in Austria, France, Ireland and Italy) led to emissions which were 1.8% of total EU emissions
- Emissions from ***waste incineration*** were 0.7% and from ***industrial processes*** 0.1%.

8.2 EMISSIONS UNDER A BUSINESS AS USUAL SCENARIO

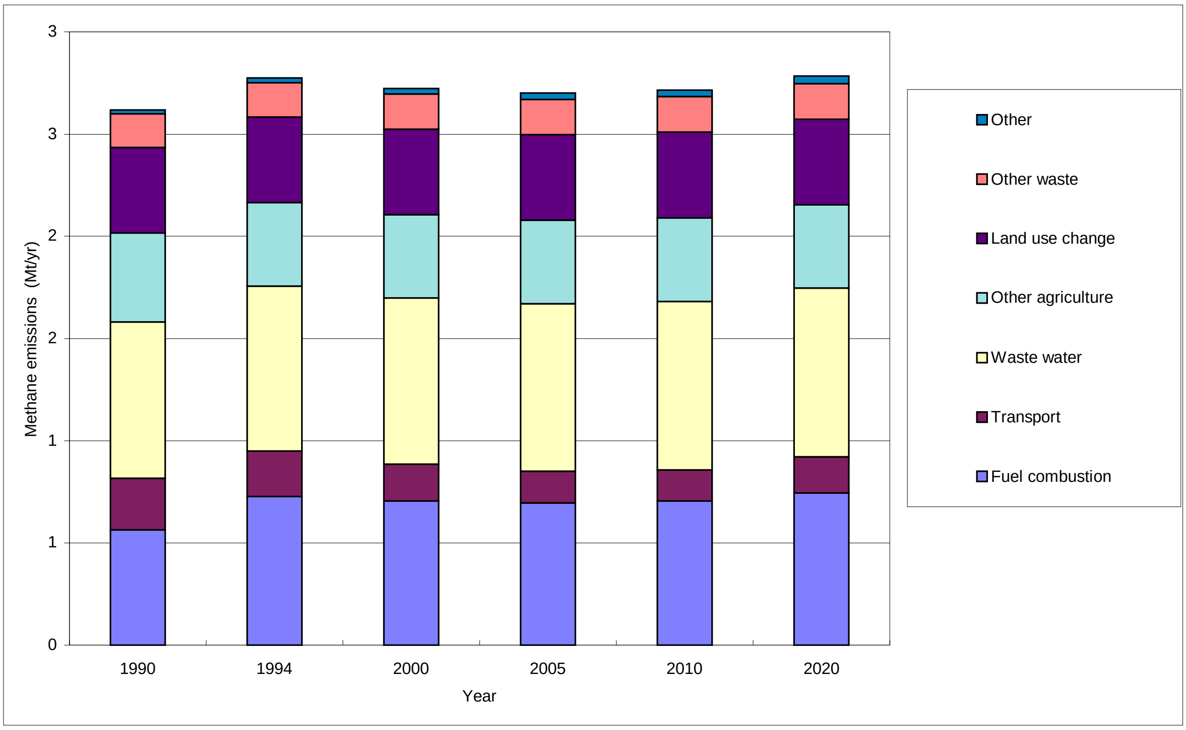
Emissions from each of these minor sectors under a business as usual scenario have been estimated as follows:

- ***Waste Water***: these emissions are assumed to change in line with the changes in population predicted in the Energy in Europe 2020 projections produced for DGXVII (pre-Kyoto scenario)
- ***Fuel Combustion Emissions*:** estimates of these emissions are taken from the Energy in Europe projections (pre-Kyoto scenario), but projections from 2000 onwards are scaled by the difference between the estimate of 1994 emissions and the 1994 emissions reported for this category by Member States in their Second National Communications.
- ***Transport Emissions*** have been estimated using a vehicle stock model produced primarily for work in this study on the abatement of NO_x and NMVOC emissions, and projections of vehicle kilometres drawn mainly from the Energy in Europe Projections. Details of this model are given in Appendix 4. Again projections were based on the 1990 and 1994 emissions reported for transport in Member States' Second National Communications. A spilt between road and non-road transport and transport modes within these categories was taken from CORINAIR90 and CORINAIR94.

- Emissions from ***other agriculture*** and ***land use change*** are assumed to remain constant over time.
- Emissions from ***industrial processes*** are assumed to increase in line with GDP growth. Estimates of future GDP are taken from the Energy in Europe projections (pre-Kyoto scenario).

Projections of emissions from the minor source sectors are shown in Figure 8.1 and are predicted to rise by 4% by 2010 and 6% by 2020.

Figure 8.1 Emissions from Other Sectors under a Business-as-Usual Scenario



8.3 IMPACT OF CO_2 REDUCTION MEASURES ON CH_4 EMISSIONS

As each of these sectors is a minor source of emissions, measures to reduce emissions have not been considered within this study. However the potential impact of measures primarily aimed at reducing CO_2 measures on other direct and indirect greenhouse gases has been examined previously in this study (AEA Technology, 1997), and results for methane are reported below.

8.3.1 Stationary Combustion

Table 8.1 and Figure 8.2 show the reduction in CH_4 emissions which is achieved through saving 1 Mt of CO_2 via a number of measures. The reductions in CH_4 include both changes in emissions from fuel combustion and changes in fugitive emissions associated with coal mining and gas production and transport.

Figure 8.2 Effect of CO_2 Reduction Measures on CH_4 Emissions

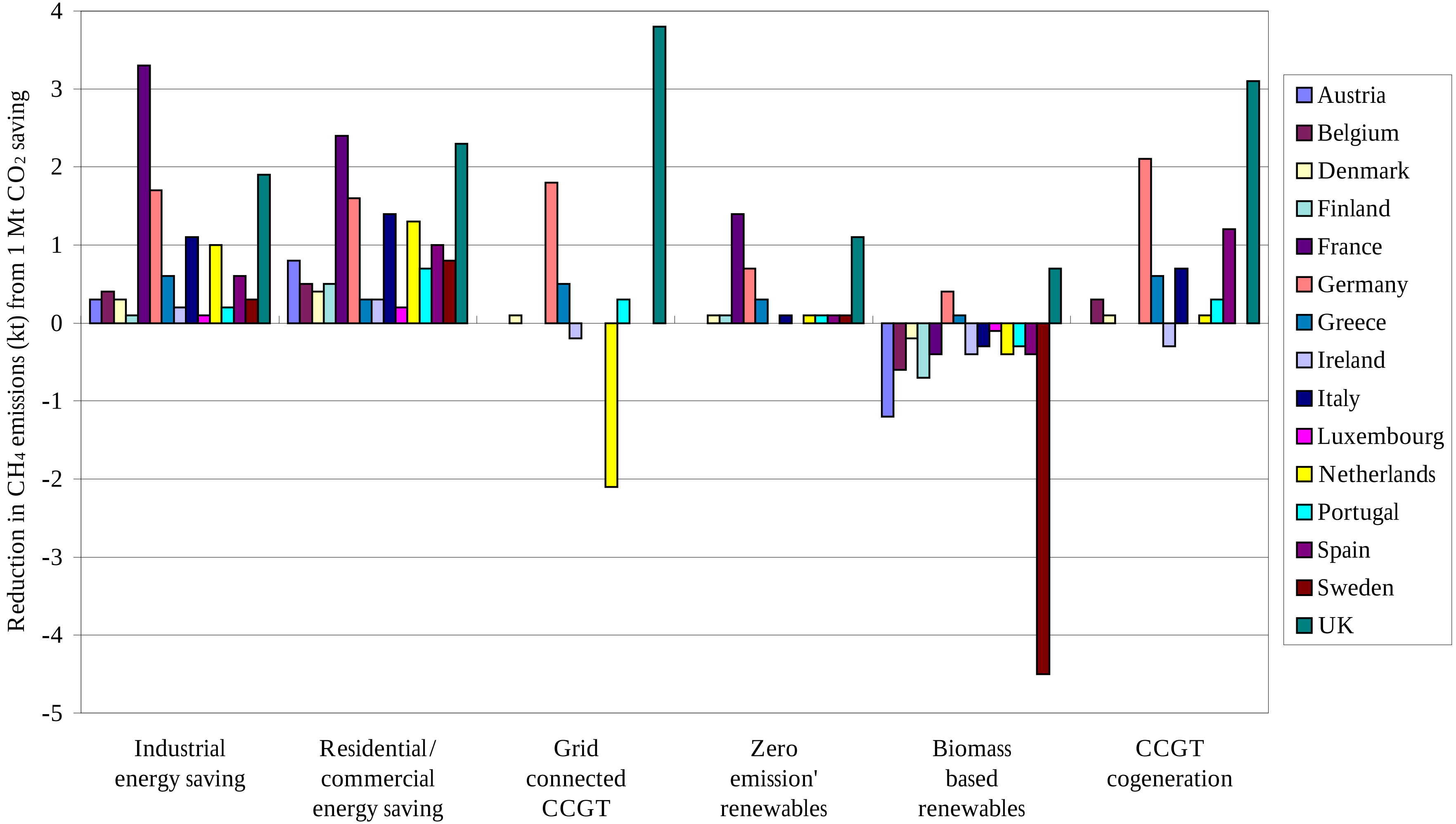


Table 8.1 Effect of CO_2 Reduction Measures on CH_4 Emissions

	Reduction* in CH_4 emissions (kt) from saving 1 Mt of CO_2 through:					
	Industrial energy saving	Res/com energy saving	Grid connected CCGT	Zero emission renewables	Biomass based renewables	CCGT cogeneration
Austria	0.3	0.8	ns	0.0	-1.2	ns
Belgium	0.4	0.5	ns	0.0	-0.6	0.3
Denmark	0.3	0.4	0.1	0.1	-0.2	0.1
Finland	0.1	0.5	ns	0.1	-0.7	ns
France	3.3	2.4	ns	1.4	-0.4	ns
Germany	1.7	1.6	1.8	0.7	0.4	2.1
Greece	0.6	0.3	0.5	0.3	0.1	0.6
Ireland	0.2	0.3	-0.2	0.0	-0.4	-0.3
Italy	1.1	1.4	0.0	0.1	-0.3	0.7
Luxembourg	0.1	0.2	0.0	0.0	-0.1	0.0
Netherlands	1.0	1.3	-2.1	0.1	-0.4	0.1
Portugal	0.2	0.7	0.3	0.1	-0.3	0.3
Spain	0.6	1.0	ns	0.1	-0.4	1.2
Sweden	0.3	0.8	ns	0.1	-4.5	ns
UK	1.9	2.3	3.8	1.1	0.7	3.1

ns - assuming displacement of the average generating mix, this measure would not achieve a net saving in CO_2, emissions, and so the effect on CH_4 has not been considered.
* A minus sign indicates that an increase in CH_4 emissions is associated with a decrease in CO_2 emissions.

For most EU countries the effect of saving 1 Mt of CO_2 via ***industrial energy efficiency*** is to reduce CH_4 emissions by between 0.1 and 0.6 kt. The exceptions are those countries with indigenous coal production (France), gas production (Italy and Netherlands) or both (UK and Germany). For these countries the savings in CH_4 are much greater (1.1 to 3.3 kt CH_4 per Mt of CO_2) as fugitive emissions from the production of these fossil fuels are reduced in addition to the smaller combustion emissions. Greece also has domestic coal production, but the high carbon intensity of its industrial energy use means that the associated CH_4 reduction is reasonably modest. Across the EU, the pattern of ***domestic and commercial energy saving*** is similar to that for industrial energy saving. However, CH_4 saving per Mt of CO_2 is generally higher (0.2 to 1.0 kt) reflecting the different fuel mix in these sectors.

Fuel switching to gas in the electricity supply industry is shown to give no overall CO_2 reduction for those countries (Austria, Belgium, Finland, France, Sweden & Spain) which already have a generating mix based primarily on either nuclear, hydro or both. For the remaining EU countries in which fuel switching to gas is a viable CO_2 control measure, the associated CH_4 saving is greatest (up to 3.8 kt of NO_x per Mt of CO_2) for those countries (Germany, Greece and the UK) in which power stations fired with indigenous coal are being displaced and therefore fugitive emissions from coal production are reduced.

Not surprisingly, ***increased use of zero emission renewables*** can help reduce emissions of both CO_2 and CH_4 in all EU countries. The extent to which a reduction in CO_2 also reduces CH_4 emissions depends on the generating mix and is once again greatest for those countries which have a high proportion of indigenous coal in their electricity fuel mix. In contrast, the use of biomass based renewables while a viable CO_2 reduction measure, is shown to increase CH_4 emissions by between 0.1 and 4.5 kt per Mt of CO_2 saved in all countries except Germany, Greece and the UK. Once again it is the displacement of CH_4 emissions from coal production in these countries that leads to an overall saving.

For ***CCGT cogeneration*** the picture is similar to that for grid connected CCGT. The differences that arise are due to the cogeneration plant displacing heat from boilers and so reducing CO_2 and CH_4 emissions from this source.

In order to estimate the potential magnitude of the impact of CO_2 reduction measures at an EU level, a broad brush methodology was adopted to estimate the emissions reductions which would result from the proposed reductions in CO_2 emissions. The proposed reductions taken were those agreed at the Environment Council in March 97 which overall gave a 10% reduction in EU CO_2 emissions by 2010[4]. As shown in Tables 8.1 and Figure 8.2, the types of CO_2 reduction measures vary considerably in their impact on CH_4 emissions. In reality Member States will achieve reductions through an appropriate mix of measures determined by a number of factors. However, in order to indicate the likely magnitude of the reductions in CH_4 emissions which might occur when CO_2 reduction targets were met, the simplistic assumption was made that each of the measures considered contributes equally to achieving the required CO_2 reductions.

For those Member States where an increase in CO_2 emissions was agreed, it has been assumed that this will be due to an increased use of energy either directly on the demand side or from increased generation by CCGT plants in the ESI. The effect on CO_2, and CH_4 emissions of increasing energy use in the industrial, residential/commercial and transport sectors and of increasing electricity generation from CCGT have been calculated and the results used in each case to find ratios linking an increase in CO_2 emissions to increases in CH_4 emissions. The total effect on CH_4 of a Member State increasing CO_2 emissions has then been estimated in the same way as for those Member States who will reduce CO_2 emissions.

As shown in Table 8.2, overall a 10% reduction in EU CO_2 emissions is estimated to lead to a 2% reduction in CH_4 emissions. As discussed above the estimated changes in CH_4 emissions includes reductions in fugitive emissions from the oil and gas sector and from coal production, as well as changes in emissions from the fuel combustion process. The reduction in CH_4 emissions associated with CO_2 measures would be less, if the measures discussed for these fuel production sectors were implemented.

8.3.2 Transport

In a similar way to stationary combustion, it is possible to reduce transport related CH_4 emissions by reducing the volume of traffic demand i.e. vehicle kilometres, which will also have the advantage of reducing CO_2 emissions and of other transport related emissions such as NMVOC and NO_x. A large number of policy options exist for reducing transport demand and these are being examined in detail in work being carried out on NO_x and NMVOC abatement.

An estimate of the overall potential for CO_2 measures to reduce transport CH_4 emissions has been made based on the recent Commission paper 'Climate Change - The EU Approach for Kyoto' (COM(97)481) which estimates the potential CO_2 emission reductions which might be achieved from various sectors. A reduction of 4 kt in CH_4 emissions in 2010 has been estimated

[4] In 1998 it was necessary to agree a new burden sharing agreement for the 8% reduction agreed by the EU at Kyoto; this was only agreed very recently at the June 1998 Environment Council Meeting.

for the 50 Mt CO_2 savings arising from intermodal shift which is identified in the paper. This reduction is 2.6 % of projected transport CH_4 emissions under the business as usual scenario.

Table 8.2 Estimated Changes in CH_4 Emissions from (Previous) CO_2 Reduction Targets

	Reduction in CO_2 emissions **%**	**Estimated change in CH_4 emissions** **kt**
Austria	-25%	0
Belgium	-10%	-2
Denmark	-25%	-2
Finland	0%	0
France	0%	0
Germany	-25%	-344
Greece	30%	39
Ireland	15%	7
Italy	-7%	-15
Luxembourg	-30%	0
Netherlands	-10%	0
Portugal	40%	41
Spain	17%	79
Sweden	5%	0
UK	-10%	-125
EU Total		**-457**
As % of 1990 CH_4 emissions		**-2%**

Table 8.3 Estimated CH_4 Emission Reductions (kt) from 50 Mt CO_2 Reduction due to Intermodal Shift

	2010	2020
Austria	0.1	0.1
Belgium	0.1	0.1
Denmark	0.1	0.1
Finland	0.1	0.1
France	0.6	0.6
Germany	1.0	0.9
Greece	0.1	0.1
Ireland	0.0	0.0
Italy	0.7	0.7
Luxembourg	0.0	0.0
Netherlands	0.2	0.1
Portugal	0.1	0.1
Spain	0.2	0.2
Sweden	0.1	0.1
UK	0.7	0.7
Total	**4.04**	**3.96**

9. Variations in Costs Across the EU

To gain an insight into how the cost-effectiveness of the various measures might vary across the EU15, each component of the base data has been adjusted to take into account known differences in relative factor prices between the base country, for which detailed cost data exists, and other members of the EU15, for which cost data needs to be estimated. To do this an appropriate relative price index has been constructed for each significant cost component e.g. purchased equipment, labour, energy and materials. A detailed methodology and tables of the cost adjustment factors used are given in Appendix 1.

Table 9.1 shows the highest and lowest cost of each measure designed to reduce emissions of methane from livestock manures (based on the best estimates of costs for each option), and the country in which the cost occurs. (a country key is given at the bottom of the table. Similar information for those measures identified to reduce emissions from landfill, mining and the oil and gas sector are displayed in Tables 9.2, 9.3 and 9.4, respectively. The range in costs for all measures is shown graphically in Figure 9.1. It is important to stress that the cost ranges shown in these tables are due, not to uncertainties in the cost or efficiency data, but to the variation in the cost-effectiveness of each measure that might be expected across the members of the EU15 as a result of differences in relative factor prices.

The figures shown in Tables 9.1 through 9.4 should be interpreted with caution. They should be seen simply as an indication of the degree to which the cost-effectiveness of the various measures considered in this study, might be expected to differ from one Member State to another. The figures also allow those Member States to be identified in which the implementation of the various measures is likely to be relatively more, or less expensive. Large uncertainties exist in relation to the accuracy of the selected indices to reflect actual variations in relative factor prices however, and in the assumption that costs are equally split between the different cost components, so that the use of a weighted average indice is appropriate. Only if one accepts these assumptions can the figures displayed in Tables 9.1 to 9.4 be interpreted as providing a reasonable indication of the cost-effectiveness of the various measures in each Member State. On this basis, it can be seen that in general the cost of measures varies by a factor of between one and a half and three between Member States, with lowest costs most frequently occurring in Ireland, Greece, and Portugal. Highest costs tend to occur in Germany, Denmark and Sweden. For some measures the UK has the lowest mitigation cost, for others the highest. In general the predicted range in costs due to the variations in factor costs is slightly greater than the range in costs estimated in Sections 3 to 7 due to uncertainties in the cost and performance data for measures.

Apart from the variation in factor costs reflected in the Tables, two other factors can be identified as having an influence of the relative cost-effectiveness of measures in Member States:

- In the case of livestock manures, the warmer the climate, the greater the potential release of methane from manures which are stored under anaerobic conditions. Measures such as anaerobic digestion therefore give a greater reduction in countries with warmer climates, and hence the cost-effectiveness of the option is improved.

- For some measures e.g. the anaerobic digestion of wastes and the composting of municipal solid waste, plant design may vary from a fairly ‘low tech’, low cost design to more highly engineered, more expensive designs, with subsequent implications for the cost-effectiveness of options (e.g. the design and cost of centralised anaerobic digestion facilities for livestock manures in the UK and Denmark). National preferences for a particular type of design, perhaps to meet other pollution criteria or standards, may thus influence the cost of abatement in a particular Member State.

Table 9.1 Variation in Costs of Measures to Reduce Emissions from Manures

Mitigation Measure		ECU/t of CH_4 'low'		ECU/t of CH_4 'high'	
Daily spread of manure:					
Cool Climate:	Pigs	2,209	(IRE)	3,989	(D)
	Dairy	4,057	(IRE)	7,327	(D)
	Beef	4,421	(IRE)	9,791	(D)
Temperate Climate:	Pigs	272	(P)	892	(I)
	Dairy	499	(P)	1,639	(I)
	Beef	667	(P)	2,191	(I)
AD - Centralised plant (UK data)					
Cool Climate:	Pigs	450	(UK)	1,930	(S)
	Dairy	828	(UK)	3,552	(S)
	Beef	1,106	(UK)	4,747	(S)
Temperate Climate:	Pigs	-1491	(P)	45	(I)
	Dairy	-810	(P)	83	(I)
	Beef	-606	(P)	111	(I)
AD - Centralised plant (Danish data)					
Cool Climate:	Pigs	1347	(IRE)	3923	(S)
	Dairy	2480	(IRE)	7222	(S)
	Beef	4639	(IRE)	13511	(S)
Temperate Climate:	Pigs	63	(P)	288	(I)
	Dairy	115	(P)	530	(I)
	Beef	216	(P)	992	(I)
AD - Small scale CHP plant (German data)					
Cool Climate:	Pigs	222	(UK)	610	(S)
	Dairy	408	(UK)	1123	(S)
	Beef	545	(UK)	1501	(S)
Temperate Climate:	Pigs	-140	(P)	36	(GR)
	Dairy	-76	(P)	67	(GR)
	Beef	-57	(P)	89	(GR)
AD - Small scale heat only plant (German data)					
Cool Climate:	Pigs	148	(UK)	518	(S)
	Dairy	272	(UK)	953	(S)
	Beef	363	(UK)	1274	(S)
Temperate Climate:	Pigs	-216	(P)	25	(GR)
	Dairy	-118	(P)	46	(GR)
	Beef	-88	(P)	61	(GR)
AD - Small scale CHP plant (Italian data)					
Cool Climate:	Pigs	177	(UK)	508	(S)
	Dairy	326	(UK)	935	(S)
	Beef	435	(UK)	1250	(S)
Temperate Climate:	Pigs	-143	(P)	27	(GR)
	Dairy	-78	(P)	49	(GR)
	Beef	-58	(P)	66	(GR)
Covered lagoons:					
Cool Climate:	Pigs	4,216	(IRE)	6,457	(S)
	Beef	10, 348	(IRE)	15,873	(S)
	Dairy	7,743	(IRE)	11,878	(S)
Temperate Climate:	Pigs	399	(P)	692	(I)
	Beef	978	(P)	1,700	(I)
	Dairy	1,068	(P)	1,272	(I)

Country Key:
Austria:(A); Belgium:(B); Denmark:(DK); Finland:(FIN); France:(F); Germany:(D); Greece:(GR); Ireland:(IRE); Italy:(I); Luxembourg:(L); Netherlands:(NL); Portugal:(P); Spain:(E); Sweden:(S); United Kingdom:(UK)

Table 9.2 Variation in Costs of Measures to Reduce Emissions from Landfills

Mitigation Measure	ECU/t of CH_4 'low'		ECU/t of CH_4 'high'	
Paper Recycling	-4279	(S)	-1145	(P)
Composting (turned windrow)	777	(GR)	1643	(D)
Composting (tunnel composting)	985	(GR)	2445	(D)
Anaerobic Digestion	1287	(GR)	3039	(D)
Incineration	1020	(GR)	2,834	(S)
Capping of Landfill	291	(P)	870	(D)
Flaring Landfill Gas	17	(P)	36	(DK)
Direct Use of Landfill Gas	-145	(P)	-35	(S)
Electricity Generation from Landfill Gas	-63	(P)	139	(S)

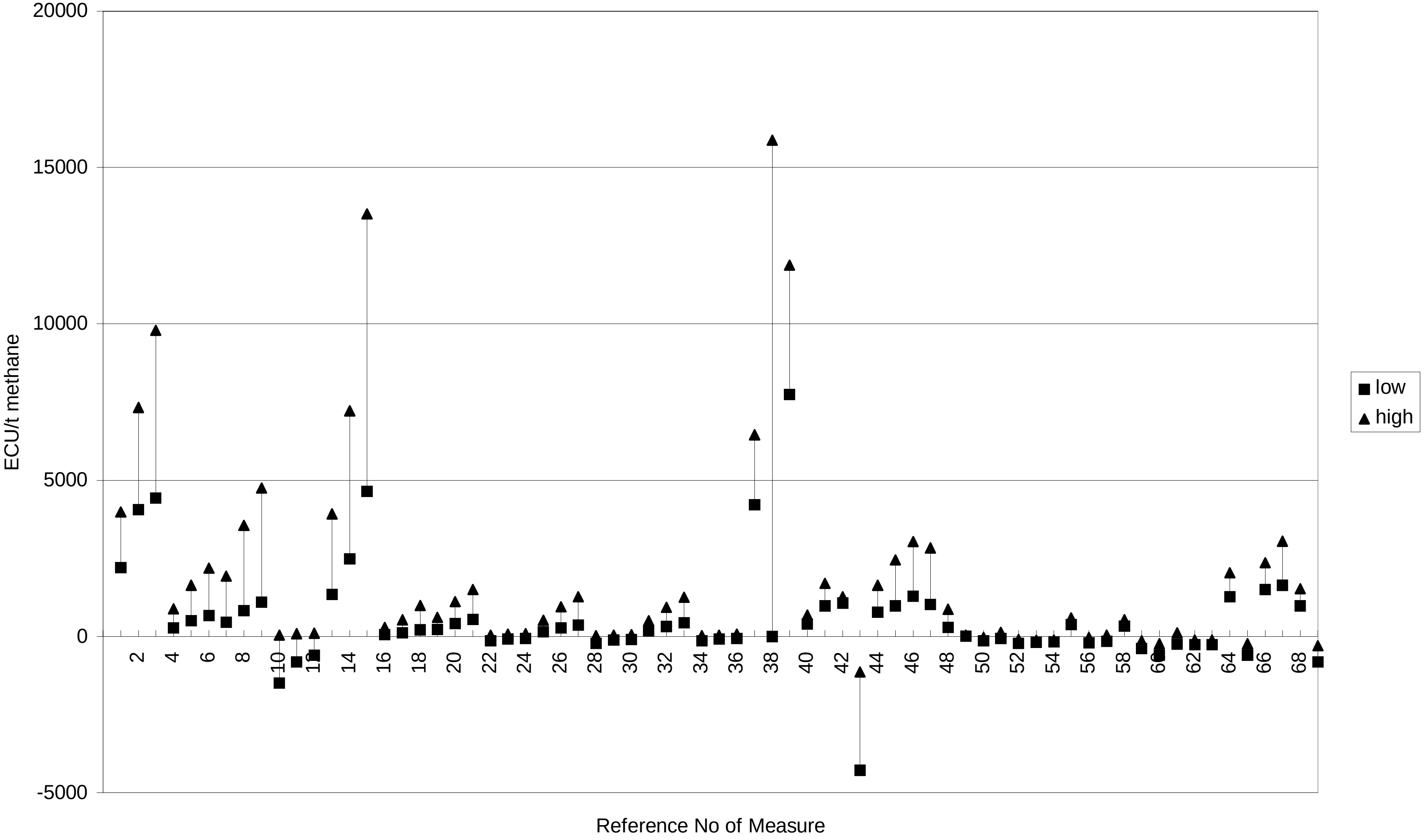
Table 9.3 Variation in Costs of Measures to Reduce Emissions from Coal Mining

Collection and utilisation rate	Steam Boiler	Recip Engine	Gas turbine (open cycle)	Gas turbine (comb cycle)	Flare
30 % collection:					
Low	-145 (D)	-224 (D)	-172 (D)	-214 (D)	8 (D)
High	-66 (F)	-91 (F)	-85 (F)	-79 (F)	5 (E, UK)
50 % collection:					
Low	-104 (D)	-183 (D)	-131 (D)	-174 (D)	44 (UK)
High	-68 (UK)	-120 (UK)	-86 (UK)	-112 (UK)	65 (D)
70 % collection:					
Low	-86 (D)	-166 (D)	-114 (D)	-156 (D)	61 (UK)
High	-55 (UK)	-108 (UK)	-74 (UK)	-100 (UK)	90 (D)

Table 9.3 Variation in Costs of Measures to Reduce Emissions from Oil and Gas Sector

Mitigation Measure	ECU/t of CH_4 'low'	ECU/t of CH_4 'high'
Flaring rather than venting (associated gas)	377 (UK)	591 (DK)
Increased gas utilisation (offshore)	-205 (IRE)	-41 (DK)
Further Increased gas utilisation (offshore)	-151 (IRE)	39 (DK)
Offshore flaring instead of venting (process vents)	335 (UK)	529 (DK)
Recompression of gas during pipeline maintenance	-392 (IRE)	-144 (UK)
Inspection and Maintenance programme (power generation)	-607 (IRE)	-234 (UK)
Use of gas turbines instead of reciprocating engine	-250 (IRE)	114 (DK)
No flushing at start up	-263 (IRE)	-104 (UK)
Electrical start up (in new installations)	-263 (IRE)	-104 (UK)
Electrical start up (retrofit)	1,273 (UK)	2,038 (DK)
Inspection and Maintenance programme (compressors)	-607 (IRE)	-234 (UK)
Prevention of system upsets through I and M programme	1,505 (UK)	2,362 (D)
Replacement of grey cast iron pipe network	1,636 (IRE)	3,054 (DK)
I and M Programme - double leak control frequency	980 (UK)	1,533 (D)
I and M Programme – instigate programme	-810 (IRE)	-295 (UK)

Figure 9.2 Variation in Costs of Measures Across the EU



Key to Figure 9.2

Ref no.	Measure
1	Daily spread of manure: Cool Climate: Pigs
2	Daily spread of manure: Cool Climate: Dairy
3	Daily spread of manure: Cool Climate: Beef
4	Daily spread of manure: Temperate Climate: Pigs
5	Daily spread of manure: Temperate Climate: Dairy
6	Daily spread of manure: Temperate Climate: Beef
7	AD - Centralised plant (UK data) Cool Climate: Pigs
8	AD - Centralised plant (UK data) Cool Climate: Dairy
9	AD - Centralised plant (UK data) Cool Climate: Beef
10	AD - Centralised plant (UK data) Temperate Climate: Pigs
11	AD - Centralised plant (UK data) Temperate Climate: Dairy
12	AD - Centralised plant (UK data) Temperate Climate: Beef
13	AD - Centralised plant (Danish data) Cool Climate: Pigs
14	AD - Centralised plant (Danish data) Cool Climate: Dairy
15	AD - Centralised plant (Danish data) Cool Climate: Beef
16	AD - Centralised plant (Danish data) Temperate Climate: Pigs
17	AD - Centralised plant (Danish data) Temperate Climate: Dairy
18	AD - Centralised plant (Danish data) Temperate Climate: Beef
19	AD - Small scale CHP plant (German data) Cool Climate: Pigs
20	AD - Small scale CHP plant (German data) Cool Climate: Dairy
21	AD - Small scale CHP plant (German data) Cool Climate: Beef
22	AD - Small scale CHP plant (German data) Temperate Climate: Pigs
23	AD - Small scale CHP plant (German data) Temperate Climate: Dairy
24	AD - Small scale CHP plant (German data) Temperate Climate: Beef
25	AD - Small scale heat only plant (German data) Cool Climate: Pigs
26	AD - Small scale heat only plant (German data) Cool Climate: Dairy
27	AD - Small scale heat only plant (German data) Cool Climate: Beef
28	AD - Small scale heat only plant (German data) Temperate Climate: Pigs
29	AD - Small scale heat only plant (German data) Temperate Climate: Dairy
30	AD - Small scale heat only plant (German data) Temperate Climate: Beef
31	AD - Small scale CHP plant (Italian data) Cool Climate: Pigs
32	AD - Small scale CHP plant (Italian data) Cool Climate: Dairy
33	AD - Small scale CHP plant (Italian data) Cool Climate: Beef
34	AD - Small scale CHP plant (Italian data) Temperate Climate: Pigs
35	AD - Small scale CHP plant (Italian data) Temperate Climate: Dairy
36	AD - Small scale CHP plant (Italian data) Temperate Climate: Beef
37	Covered lagoons: Cool Climate: Pigs
38	Covered lagoons: Cool Climate: Beef
39	Covered lagoons: Cool Climate: Dairy
40	Covered lagoons: Temperate Climate: Pigs
41	Covered lagoons: Temperate Climate: Beef
42	Covered lagoons: Temperate Climate: Dairy
43	Paper recycling
44	Composting (turned windrow)
45	Composting (tunnel composting)
46	Anaerobic Digestion
47	Incineration
48	Capping of Landfill
49	Flaring Landfill Gas
50	Direct Use of Landfill Gas
51	Electricity Generation from Landfill Gas
52	Mining 30% collection Recip Engine
53	50% collection
54	70% collection
55	Flaring rather than venting (associated gas)
56	Increased gas utilisation (offshore)
57	Further Increased gas utilisation (offshore)
58	Offshore flaring instead of venting (process vents)
59	Recompression of gas during pipeline maintenance
60	Inspection and Maintenance programme (power generation)
61	Use of gas turbines instead of reciprocating engine
62	No flushing at start up
63	Electrical start up (in new installations)
64	Electrical start up (retrofit)
65	Inspection and Maintenance programme (compressors)
66	Prevention of system upsets through I and M programme
67	Replacement of grey cast iron pipe network
68	I and M Programme- double leak control frequency
69	I and M Programme - instigate programme

10. Summary

10.1 BACKGROUND

Until recently, strategies for addressing climate change have principally been focused on reducing emissions of the main greenhouse gas carbon dioxide, but the importance of other greenhouse gases and opportunities for their abatement have been increasingly recognised in the last couple of years. This culminated in an agreement at the conference of the parties in Kyoto in December 1997 to set legally binding targets for reducing greenhouse gas emissions based on a ‘six gas basket’, that is targets apply to the emissions of the six greenhouse gases (carbon dioxide, methane, nitrous oxide, hydrofluorocarbons, perfluorocarbons and sulphur hexafluoride) all weighted by their (100 year) global warming potential. The EU agreed to reduce emissions of the six gases by 8% of 1990 levels by 2010, and arrangements for sharing this reduction among Member States were agreed recently at the Environment Council in June 1998.

The EU had begun to consider developing a strategy for reducing methane emissions several years ago. In February 1993, the EU in its Fifth Action Programme for the Environment “Towards Sustainability”, defined a series of actions for greenhouse gases: which included the aim of possibly reducing methane emissions. In December 1994, the Environment Council asked the Commission to submit a strategy to reduce emissions of greenhouse gases other than CO_2, in particular methane and nitrous oxide. A strategy paper for methane was produced by the Commission in November 1996, and submitted to the Council and to the European Parliament It set out a number of potential actions in the agricultural, waste and energy sector which could be incorporated into a Community emissions mitigation strategy.

This study for DGXI of the European Commission, considers anthropogenic methane emissions[5] within the EU and for each of the major source sectors (enteric fermentation from livestock, livestock manures, landfills, coal mining and the oil and gas sector) examines the technical feasibility of measures (including those in the methane strategy paper) to reduce emissions. Wherever sufficient cost and performance data is available, the cost-effectiveness of the measures in terms of ECU (1995) per tonne of methane abated is also estimated. The applicability of the measures is assessed to allow the calculation of achievable reductions compared to a business-as- usual scenario to (2020), and to allow the projection of emissions under a ‘with measures scenario’. The results for all sectors are summarised below and indicate that methane emissions could be reduced to 39% below 1990 levels by 2010. A parallel study into reducing nitrous oxide emissions (AEA Technology Environment, 1998) found that emissions of nitrous oxide could be reduced to 20 to 29% below 1990 levels by 2010.

10.2 CURRENT EMISSIONS

In 1994 it is estimated that the EU accounted for about 6% of global anthropogenic emissions of the direct greenhouse gas, methane. Methane is a potent greenhouse gas with a global warming potential (over 100 years) of 21 relative to the main greenhouse gas, carbon dioxide. EU

[5] Natural sources and sinks are excluded from this study.

methane emissions of 22 Mt in 1994, were equivalent to 14% of CO_2 emissions in that year and were thus a significant contributor to total greenhouse gas emissions in the EU.

The main sources of anthropogenic emissions in 1994 are shown in Table 10.1. It should be noted that the uncertainty in emissions estimates for some sources is significant and overall, uncertainty may be around 20%.

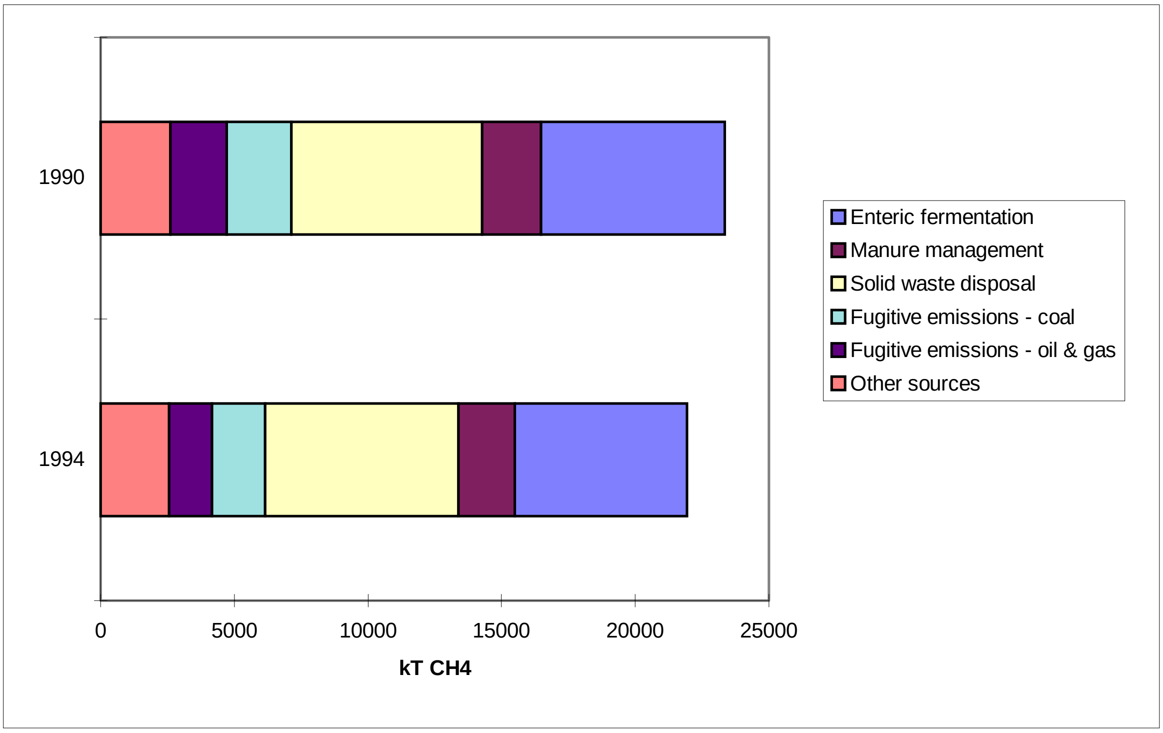
Emissions in 1990 were estimated as 23.3 Mt and emissions had thus fallen by 6% by 1994 (Figure 10.1). This was due mainly to a significant decline in emissions from both coal mining and the oil and gas industry (by 18% and 23% respectively). Reduction in emissions from coal mining were primarily a result of a decline in deep coal mining in the EU over this period, although improved methane capture is also thought to have contributed to this trend. Emissions from the oil and gas sector fell due to the replacement of gas distribution pipework which reduced fugitive emissions. Emissions from the agricultural sector fell by about 6% (0.5 Mt), due mainly to a decline in animal numbers.

Table 10.1 Sources of Methane Emissions in the EU (1994)

Sector	**Emissions**	**% of total**	**Of which:**	
Agriculture	9.0 Mt	40.8%	Enteric fermentation	29.4%
			Livestock manure	9.6%
			Other agriculture	1.9%
Waste	8.2 Mt	37.4%	Solid waste disposal	32.9%
			Wastewater treatment	3.7%
			Waste incineration	0.8%
Energy sector	4.3 Mt	19.6%	Coal mining, transport and storage	9.0%
			Gas production and distribution	7.3%
			Combustion (including transport)	3.3%
Other	0.4 Mt	2.1%	Land use changes	2.0%
			Other sources	0.1%
Total	**21.9 Mt**	**100%**		

Source: Member States and EU Second Communications under the FCCC.

Figure 10.1 Trends in Methane Emissions 1990-1994



Source: Member States and EU Second Communications to the FCCC.

10.3 BUSINESS-AS-USUAL PROJECTIONS

Business-as-usual projections have been made for each of the main sectors on a detailed basis as set out in the relevant sections (3 to 7) of this report. Projections for the minor sectors were made in Section 8. Emissions under the business as usual scenario are shown by sector in Figure 10.1 and Table 10.2. By 2010 emissions have fallen by 2 Mt to 9% below 1990 levels due predominantly to a dramatic decline in emissions from coal mining as coal production falls. There is also a significant reduction in livestock related emissions (particularly enteric fermentation) as cattle numbers are predicted to fall. These reductions are partially offset by increased emissions from landfills due to large volumes of waste produced and hence disposed of to landfill and a small rise in emissions from the oil and gas sector due to increased production levels and increased losses due to extended gas distribution networks.

The business-as-usual projections generally reflect a continuation of existing trends and do not take into account specific policies and measures which the EU and Member States may have already put in place to reduce methane emissions (e.g. the landfill directive). These reductions are included within those estimated for each of the measures identified. It was not possible to separate out the effect of existing policies and measures as many Member State's Second National Communications did not contain enough detail about the scope and impact of identified policies and measures. In some cases where data on expected production was provided, only reductions expected by 2000 are stated.

Figure 10.2 CH_4 Emissions under the Business as Usual Scenario (Mt/year)

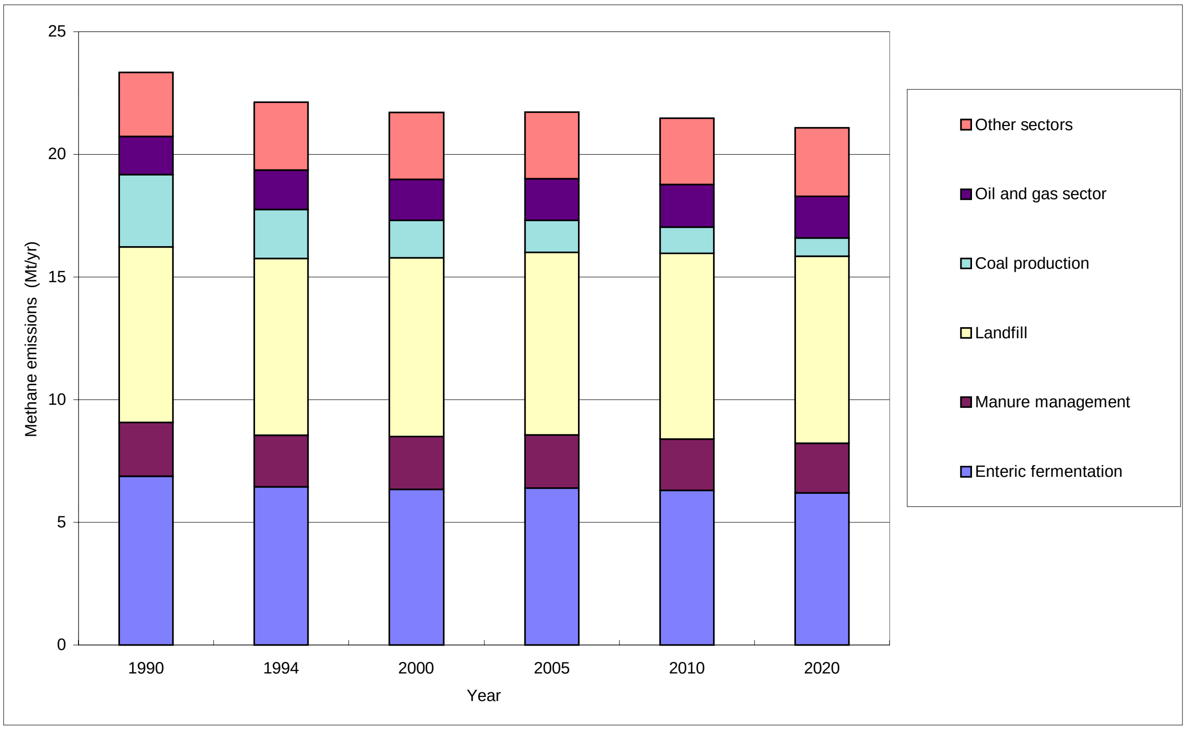


Table 10.2 CH_4 Emissions under the Business as Usual Scenario (kt/year)

	1990	1994	2000	2005	2010	2020
Enteric fermentation	6878	6443	6355	6396	6302	6217
Animal manures	2199	2105	2150	2163	2097	2009
Landfill	7144	7223	7280	7449	7576	7623
Coal production	2946	1978	1521	1305	1069	741
Oil and gas sector	1564	1612	1678	1706	1725	1702
Waste water	764	807	814	821	824	825
Other agriculture and land use change	854	841	841	841	841	841
Fuel combustion	564	507	527	540	553	571
Transport	253	221	179	155	152	175
Industrial processes and solvents	20	23	27	30	34	37
Other	165	169	171	173	174	175
Total	**23349**	**21931**	**21542**	**21579**	**21348**	**20917**
Change from 1990		**-6%**	**-8%**	**-8%**	**-9%**	**-10%**

10.4 MEASURES TO REDUCE EMISSIONS

Measures have been identified to reduce emissions in all of the main sectors and are listed in order of cost-effectiveness in Table 10.3, together with an estimate of the achievable reductions in 2010 and 2020 compared to the business-as-usual scenario. It was noted in Section 5, that it was difficult to apportion the reductions achieved under improved landfill gas recovery and diversion of biodegradable waste from landfill accurately to the individual measures considered.

For the purposes of Table 10.3 and the cost-effectiveness curves the following assumptions concerning reductions have been made:

- **improving landfill gas recovery and utilisation:** 10% of the gas recovered is utilised for heat generation (direct use), 40% is used for electricity generation and 60% is flared.
- **diverting biodegradable waste from landfill:** 50% of the reduction is achieved through paper recycling, 15% is achieved using 'lower' cost techniques such as turned windrow composting, 25% is achieved through 'medium cost' techniques such as incineration, and 15% is achieved through higher cost techniques such as anaerobic digestion (or more highly engineered composting schemes).

10.4.1 Reductions

In total the measures offer reductions of 7.2 Mt and 9.2 Mt respectively in 2010 and 2020 and would bring emissions down to 61% of 1990 levels by 2010 and 50% of 1990 levels by 2020. The largest reductions (Figure 10.3) arise from measures directed at landfilling of waste, although significant reductions are also available from improving manure management and in the oil and gas sector. Implementation of the two groups of measures leading to the largest reductions in landfill gas (utilisation or flaring of landfill gas and diversion of organic waste from landfill) is already being suggested under the proposed Landfill Directive.

Reductions in enteric fermentation emissions are relatively small given that it is the largest single source sector. This reflects the fact that one of the measures identified (propionate precursors) is thought to have a fairly limited applicability; it should also be noted that it had high costs (over 2700 ECU/t of CH_4). A number of other potential measures were identified in this sector but firm data on their effectiveness has not yet been established; furthermore some of the more promising options identified were thought to be unsuitable for implementation due to conflict with animal welfare interests and consumer concerns. Further R&D is thus required in this area.

10.4.2 Cost-effectiveness of Measures

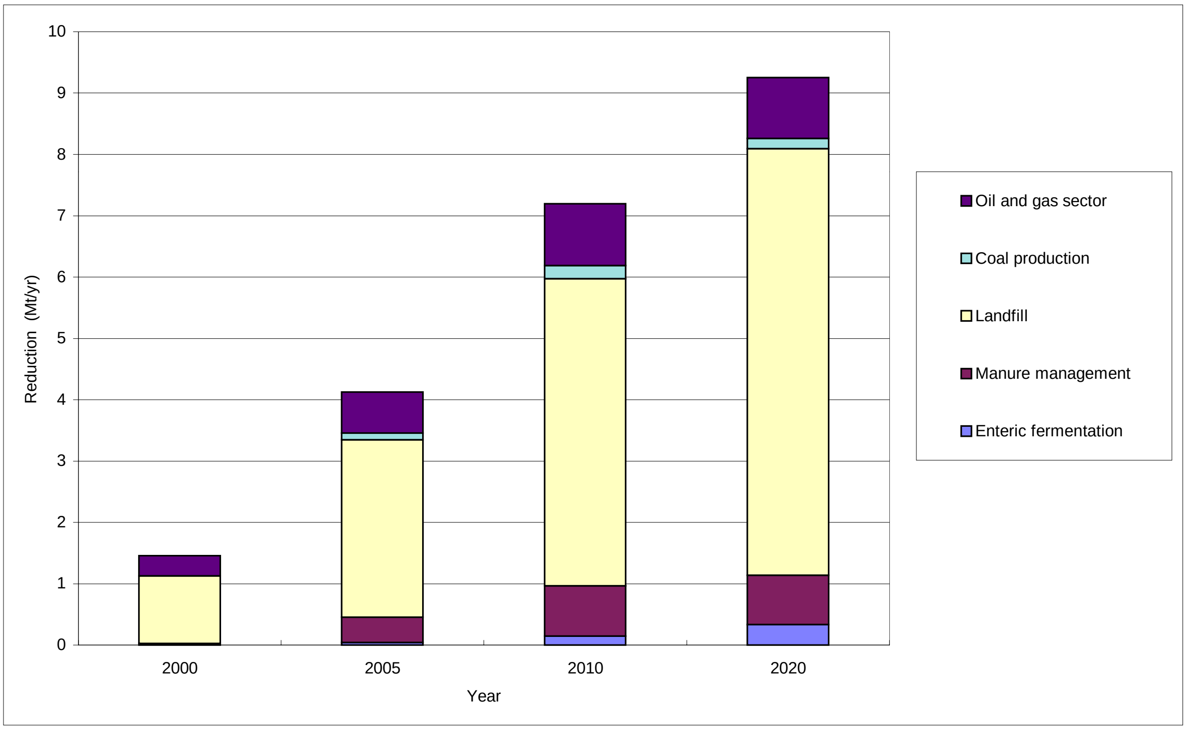
Cost-effectiveness curves illustrating the cost and reductions achieved by each measure are given in Figure 10.4 and Figure 10.5 for 2010 and 2020 respectively. In order to aid readability, the curves are also shown in two parts (Figures 10.4a and 10.4b and 10.5a and 10.5b). Measures are labelled using the numbers in the first column of Table 10.3.

A number of measures (various measures in the oil and gas sector, some recovery of mine methane and recovery of landfill gas for direct use) are identified as being 'cost-positive', as the value of the methane recovered and utilised more than offsets the cost of the measure. Paper recycling is also a 'cost-positive' option. The introduction of high genetic merit cows has no cost associated with it due to concomitant improvements in animal productivity. In total these cost-positive and zero cost measures account for 22% of potential reductions in 2010.

Modelling work carried out previously for the European Commission has indicated that the marginal cost of measures to achieve a 15% CO_2 reductions is 55 to 165 ECU/t CO_2 (1990 ECU) (Klassen, 1997). This is equivalent to a cost of about 1300 to 3890 ECU/t CH_4 (1995 ECU), and it can be seen that almost all measures fall below this limit, indicating that they would be at least as cost-effective as some CO_2 reduction measures. Measures with a cost greater than 3890 ECU/t account for about 2% of the total reduction in 2010 and 3% of the reduction in 2020.

The uncertainty associated with the cost of a number of measures was estimated and generally found to be between 15 to 35%.

Figure 10.3 Reductions Offered by Measures Compared to the Business as Usual Scenario



10.5 EMISSIONS UNDER A WITH MEASURES SCENARIO

Emissions to 2020, if all measures are implemented, are shown by sector in Figure 10.6 and Table 10.4. Figure 10.7 shows emissions compared to the Business as Usual Scenario and also a scenario where only measures with a cost below the limit of 3890 ECU/t CH_4 (identified above) are implemented.

In total, if all measures were implemented, emissions would fall to 39% below 1990 levels by 2010 and 50% by 2020. If only measures with a cost less than 3890 ECU/t CH_4 are implemented then the reductions would be just under 1% less in 2010 and 1.4% less in 2020 (i.e. 38.6% and 48.7% respectively).

10.6 COMPARISON WITH OTHER CH_4 REDUCTION STUDIES

As part of an (ongoing) study for DGXI[6] led by Coherence on the economic evaluation of quantitative objectives for climate change, Ecofys have produced a report on the potential and costs of methane and nitrous oxide emissions reductions in the EU (Hendriks, de Jager and Blok, 1997). There is a significant difference in the business as usual forecasts in the two studies - the Ecofys study predicts a 26% reduction in CH_4 emissions by 2010, as compared to a 9%

[6] This study which is led by Coherence is producing an economic evaluation of quantitative objectives for climate change (contract No B4-3030/95/000449/MAR/B1).

decrease in this study. The main reason for this difference is that this study assumes that landfilling practices continue as at present, i.e. the same fraction of waste is disposed of to landfill, and the same fraction of landfill gas is recovered. The Ecofys study assumes that a number of measures planned by Member States to reduce waste going to landfill and to increase recovery of landfill gas are implemented. Other difference are assumptions about the proportions of open cast and deep mined coal which is extracted, and the way that fugitive emissions from gas networks are modelled.

The Ecofys study forecasts a slightly greater reduction in emissions under a with measures scenario, of 51% by 2010 compared to a 40% reduction in this study. The types of measures considered by both studies are broadly similar, and give similar overall reduction potentials for most sectors (after allowing for the fact that in the Ecofys study, some of the reduction potential for landfills is included in the business as usual scenario). Exceptions are enteric fermentation, where the Ecofys study considers that improved feed conversion techniques will provide significant reductions, and the gas distribution sector, where the Ecofys study considers that replacement of old cast iron pipes will give greater reductions than in this study. A paper written by Coherence identifying differences in the two studies, and suggesting a synthesis of the results is included as Appendix 7.

10.7 CONCLUSIONS

This examination of measures to reduce methane emissions shows that such measures could make a substantial contribution to the EU climate change strategy.

The implementation of all measures identified would lead to a reduction of 7.2 Mt of CH_4 (equivalent to 151 Mt of CO_2) and combined with the downward trend identified in the business as usual scenario would bring CH_4 emissions to 39% (9.2 Mt of CH_4) below 1990 levels by 2010. The EU six gas basket of emissions in 1990 is estimated to be about 4 247 Mt CO_2-equivalent,, and a reduction of 340 Mt of CO_2-equivalent is therefore required to meet the EU's Kyoto target under the Framework Convention on Climate Change. The identified reduction in methane emissions, which is equivalent to 193 Mt of CO_2 could therefore meet just over half of the EU's Kyoto target.

The cost of the measures identified ranges from -2208 ECU (1995) per tonne CH_4 (-105 ECU/t CO_2 equivalent) to 5686 ECU (1995) per tonne CH_4 (262 ECU/t CO_2 equivalent), but the higher cost measures account for only a small proportion of reductions and 98% of the reductions achievable by 2010 have a cost of less than 2400 ECU (1995) per tonne CH_4. This is equivalent to 114 ECU (1995) per tonne CO_2 equivalent and is thus within the range of costs (55 to 165 ECU (1990) per tonne CO_2) calculated in previous European Commission modelling work as the marginal costs of achieving a 10 to 15 % reduction in EU CO_2 emissions.

Table 10.3 Key for Graphs showing Cost and Applicability of All Measures

Sector	Measure	Cost ECU/t CH_4	Cost ECU/t CO_2	Red'n in 2010 (kt)	Red'n in 2020 (kt)
1 LF	Paper recycling	-2208	-105	971	668
2 O&G	Inspection and maintenance programme (pipelines)	-213	-10	44	44
3 O&G	Inspection and maintenance programme (power generation)	-174	-8	10	11
4 O&G	Inspection and maintenance programme (compressors)	-174	-8	40	45
5 O&G	Recompression of gas during pipeline maintenance	-153	-7	11	12
6 O&G	Compressors - no flushing at start up	-113	-5	6	6
7 O&G	Compressors - electrical start up (in new installations)	-113	-5	6	6
8 Coal	Install plant to achieve 30% recovery and utilisation of mine gas	-78	-4	54	28
9 LF	Landfill gas recovery and use for heat generation	-76	-4	223	368
10 Coal	Upgrade plant to achieve 70% instead of 30% recovery and utilisation of mine gas	-66.5	-3	124	100
11 O&G	Increase gas utilisation offshore	-57	-3	28	21
12 Coal	Install plant to achieve 50% recovery and utilisation of mine gas	-52	-2	16	12
13 Coal	Upgrade plant to achieve 50% instead of 30% recovery and utilisation of mine gas	-42	-2	13	10
14 Coal	Install plant to achieve 70% recovery and utilisation of mine gas	-40	-2	14	11
15 EF	High genetic merit cows	0	0	63	95
16 O&G	Further increase gas utilisation offshore	7	0	28	21
17 LF	LFG recovery and use for electricity generation	23	1	1338	2210
18 AM	Anaerobic Digestion - Temperate climate - Pig slurry	30	1	239	243
19 LF	LFG recovery and flaring	44	2	669	1105
20 O&G	Use of gas turbines instead of reciprocating engines	54	3	4	4
21 AM	Anaerobic Digestion - Temperate climate - Dairy cow manure	56	3	26	23
22 AM	Anaerobic Digestion - Temperate climate - Non-dairy cattle manure	74	4	64	71
23 AM	Anaerobic Digestion - Cool climate - Pig slurry	181	9	104	89
24 AM	Anaerobic Digestion - Cool climate - Dairy cow manure	334	16	32	28
25 O&G	Flaring rather than venting (associated gas)	377	18	12	10
26 O&G	Flaring instead of venting (offshore process vents)	441	21	17	13
27 AM	Anaerobic Digestion - Cool climate - Non-dairy cattle manure	446	21	76	77
28 LF	Oxidation of fugitive methane by improved capping of landfill	592	28	833	1937
29 AM	Daily spread of slurry - Temperate climate - Pig slurry	645	31	57	57
30 LF	Diversion of biodegradable waste from landfill through low cost measure (e.g. turned windrow composting)	1033	49	486	334
31 AM	Daily spread - Temperate climate - Dairy cow manure	1183	56	42	40
32 O&G	Double leak control frequency for pipelines	1203	57	156	156
33 O&G	Prevention of system upsets through inspection and maintenance programme	1364	65	38	43
34 LF	Diversion of biodegradable waste from landfill through medium cost measures (e.g. incineration)	1423	68	971	668
35 AM	Daily spread - Temperate climate - Non-dairy manure	1579	75	43	47
36 O&G	Compressors - Electrical start up (retrofit)	1646	78	2	3
37 LF	Diversion of biodegradable waste from landfill through medium cost measures (e.g. anaerobic digestion)	1858	88	486	334
38 AM	Daily spread - Cool climate - Pig slurry	2264	108	36	30
39 O&G	Replacement of grey cast iron pipe network	2378	113	600	600
40 EF	Propionate precursors - Dairy cows	2729	130	124	21
41 AM	Daily spread - Cool climate - Dairy cow manure	4124	196	47	42
42 AM	Daily spread - Cool climate - Non-dairy cattle manure	5505	262	54	56
42 EF	Propionate precursors - Non-dairy cattle	5686	271	70	222
	Total Reduction compared to Business as Usual Scenario			**7196**	**9256**

Key:

AM	Animal manures	LF	Landfill	Coal	Coal mining
O&G	Oil and gas sector	EF	Enteric fermentation		

Figure 10.4 Cost-Effectiveness of Reductions Achieved by All Measures in 2010

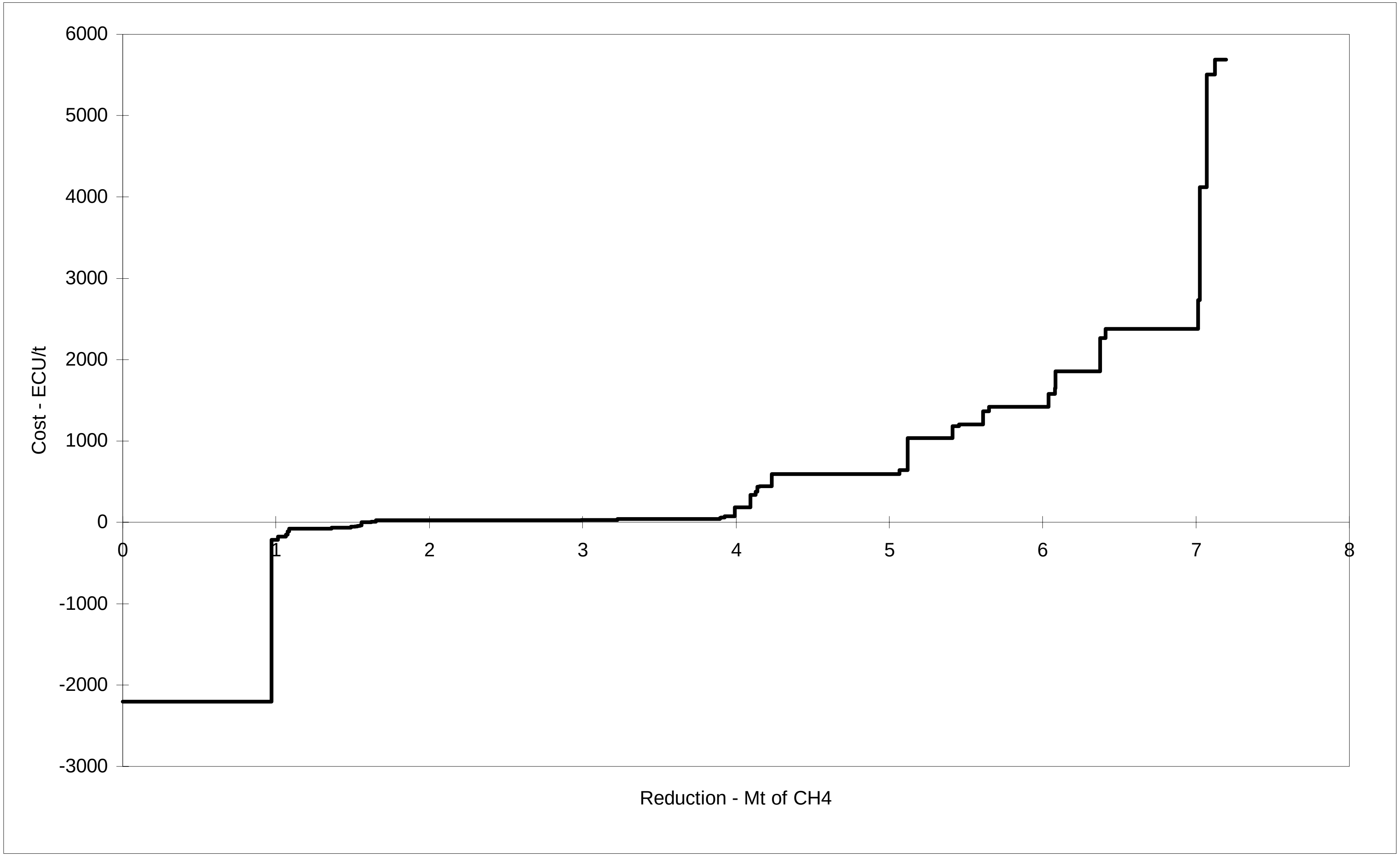
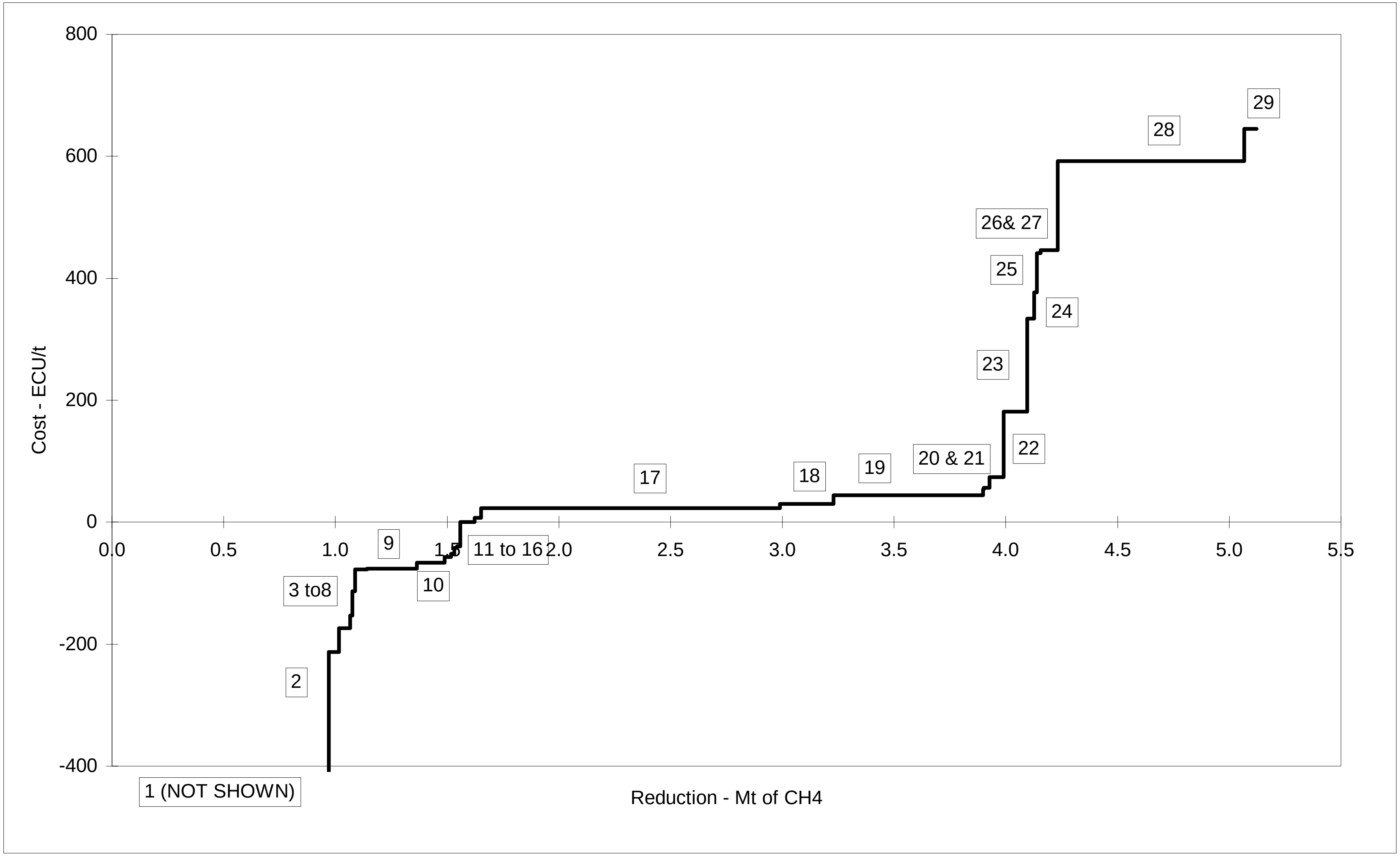


Figure 10.4a Cost-Effectiveness of Reductions Achieved by all Measures costing less than 1000 ECU/t CH_4 in 2010



Note: Measure 1 - Paper recylcing which is not shown in order to aid readability offers a reduction of 971 kt at a cost of -2208 ECU/t methane

Figure 10.4b Cost-Effectiveness of Reductions Achieved by all Measures costing more than 1000 ECU/t CH_4 in 2010

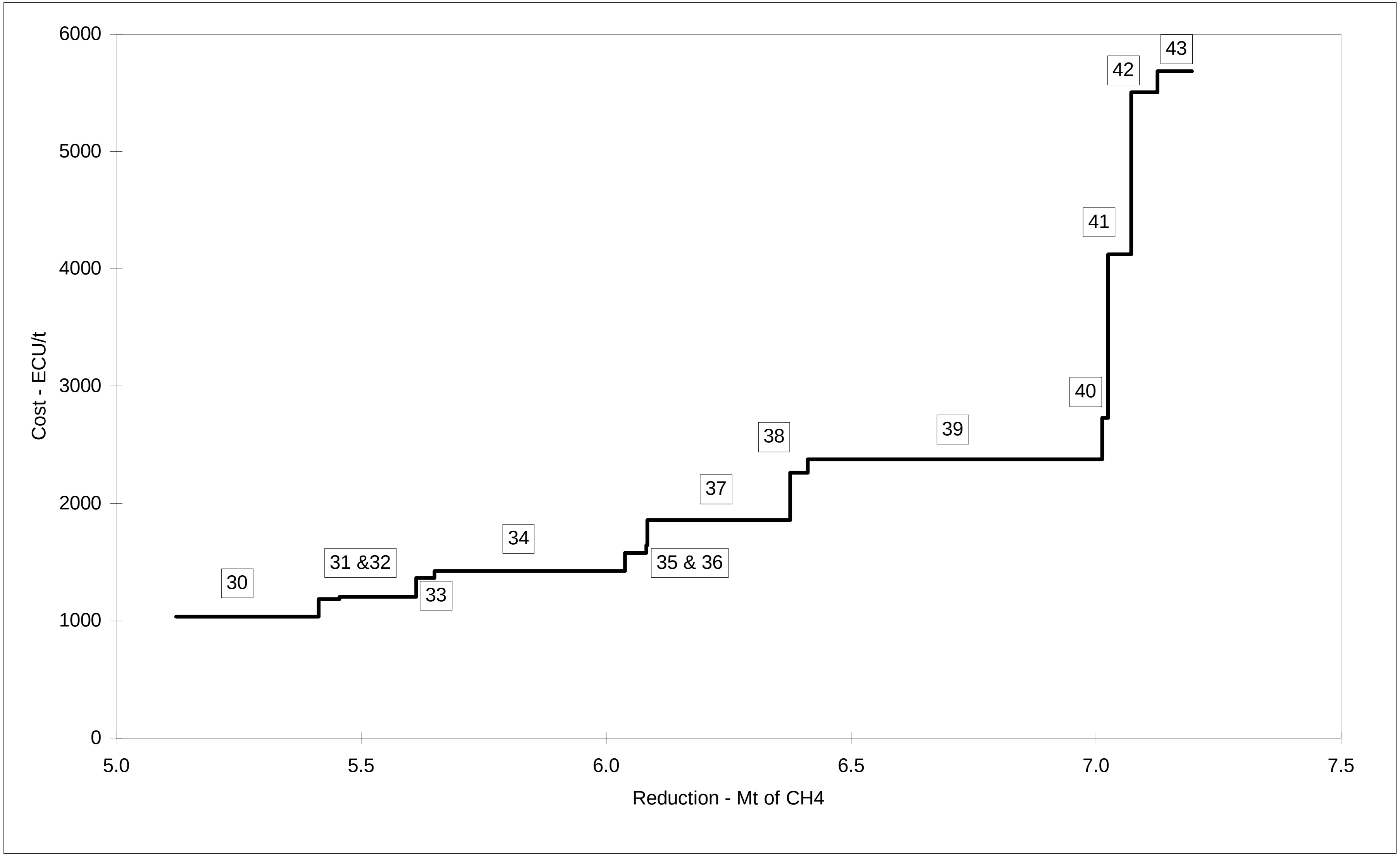


Figure 10.5 Cost-Effectiveness of Reductions Achieved by All Measures in 2020

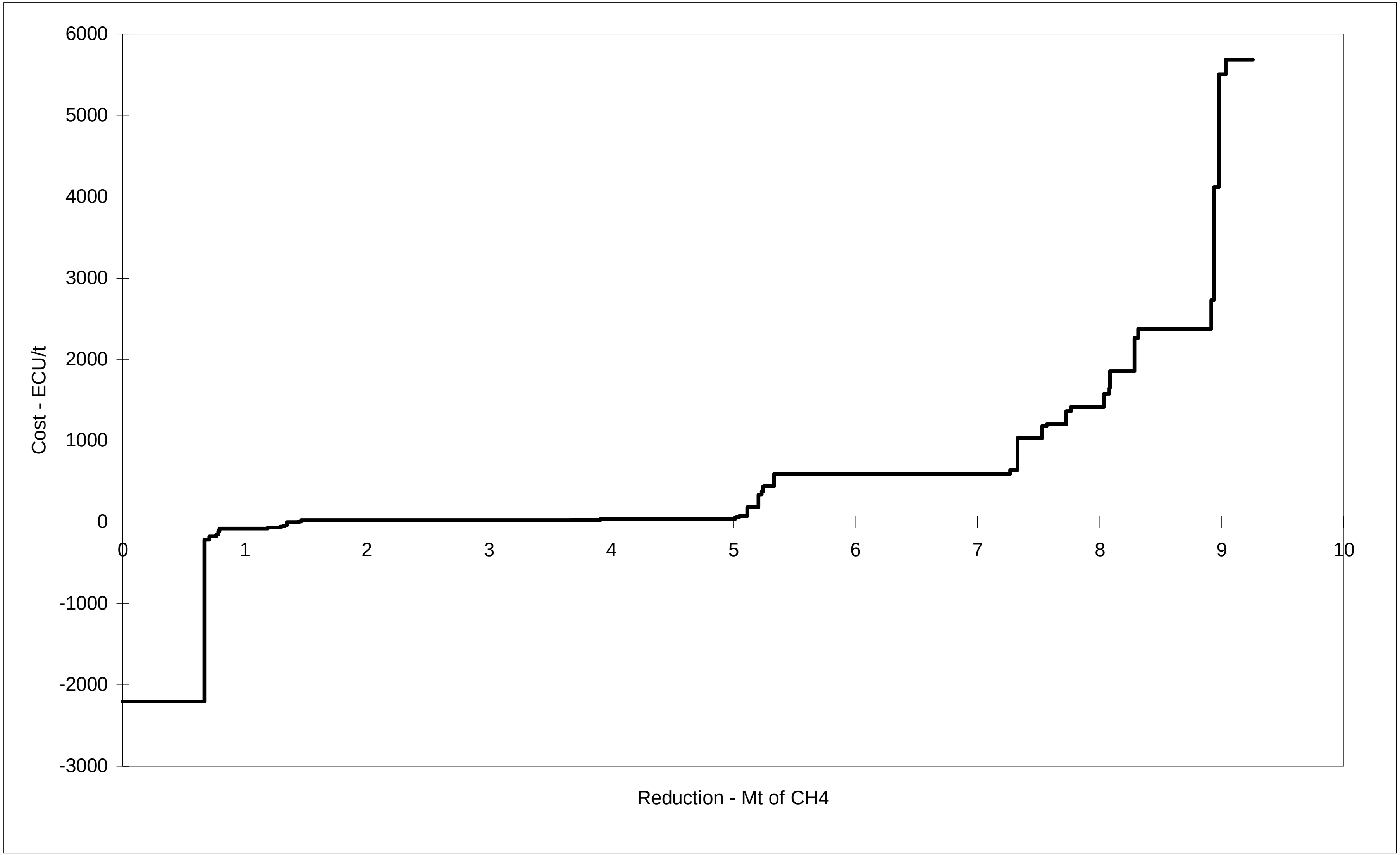
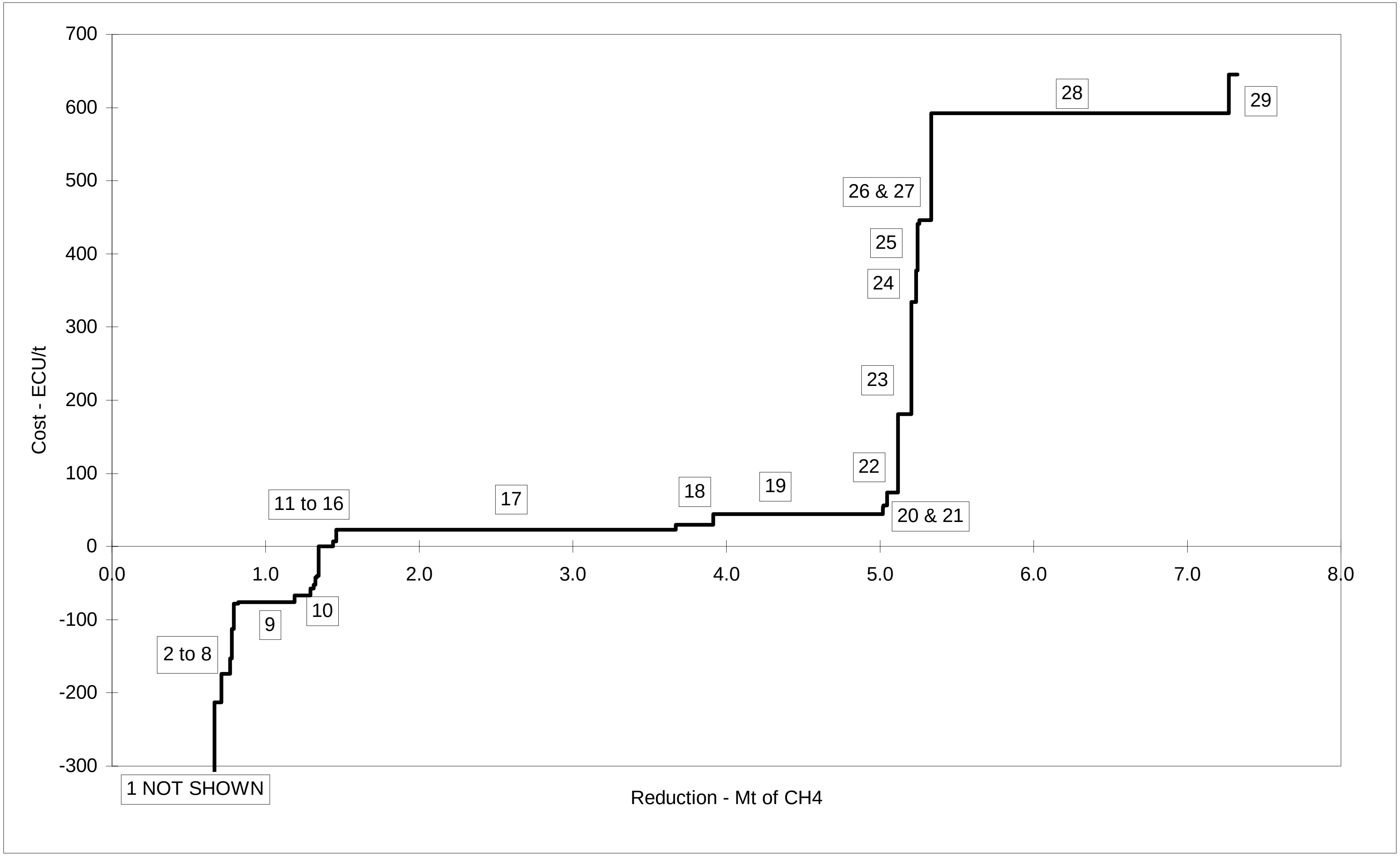


Figure 10.5a Cost-Effectiveness of Reductions Achieved by all Measures costing less than 1000 ECU/t CH_4 in 2020



Note: Measure 1 - Paper recylcing which is not shown in order to aid readability offers a reduction of 668 kt at a cost of -2208 ECU/t methane

Figure 10.5b Cost-Effectiveness of Reductions Achieved by all Measures costing more than 1000 ECU/t CH_4 in 2020

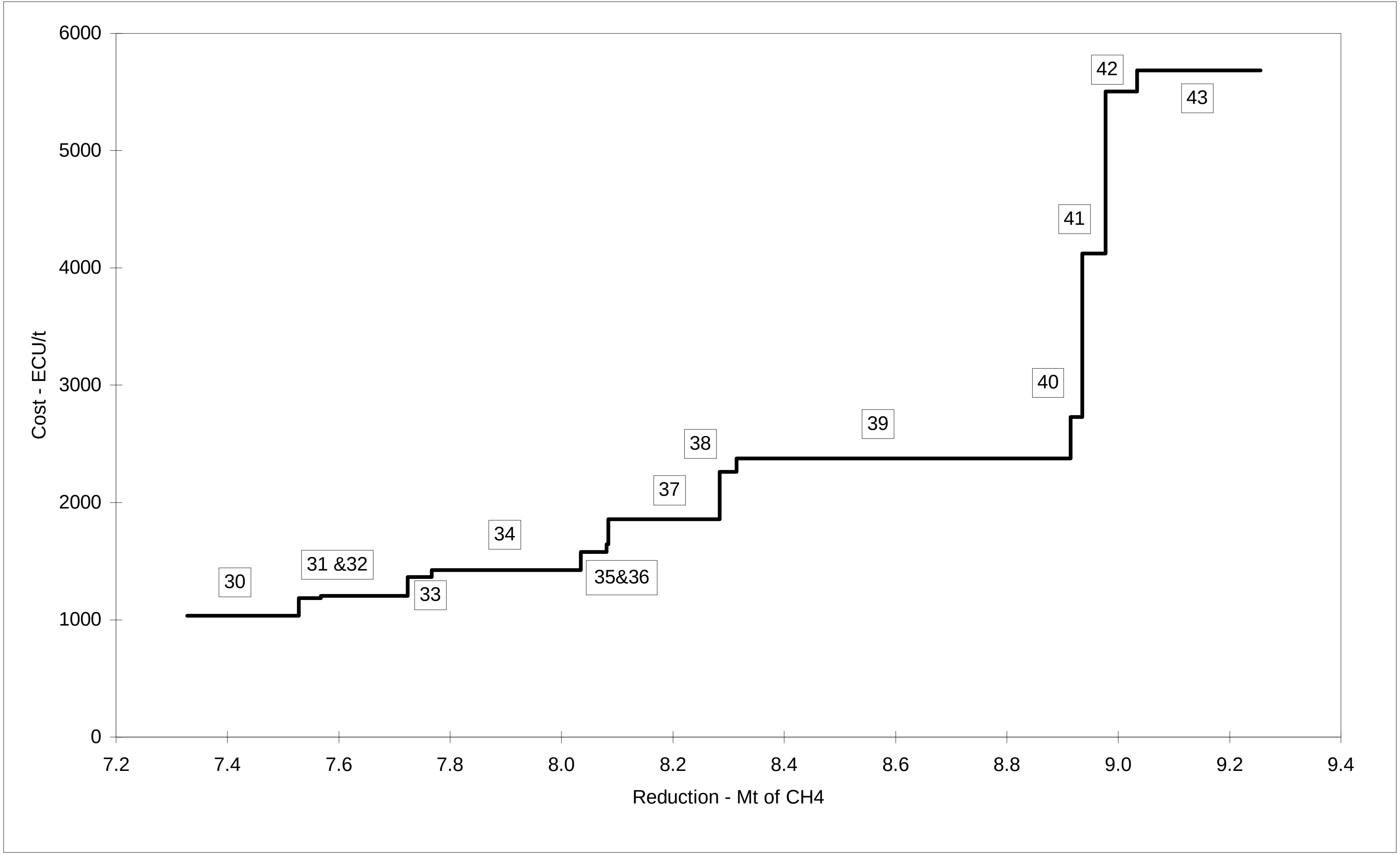


Figure 10.6 EU Methane Emissions by Sector with All Measures Implemented

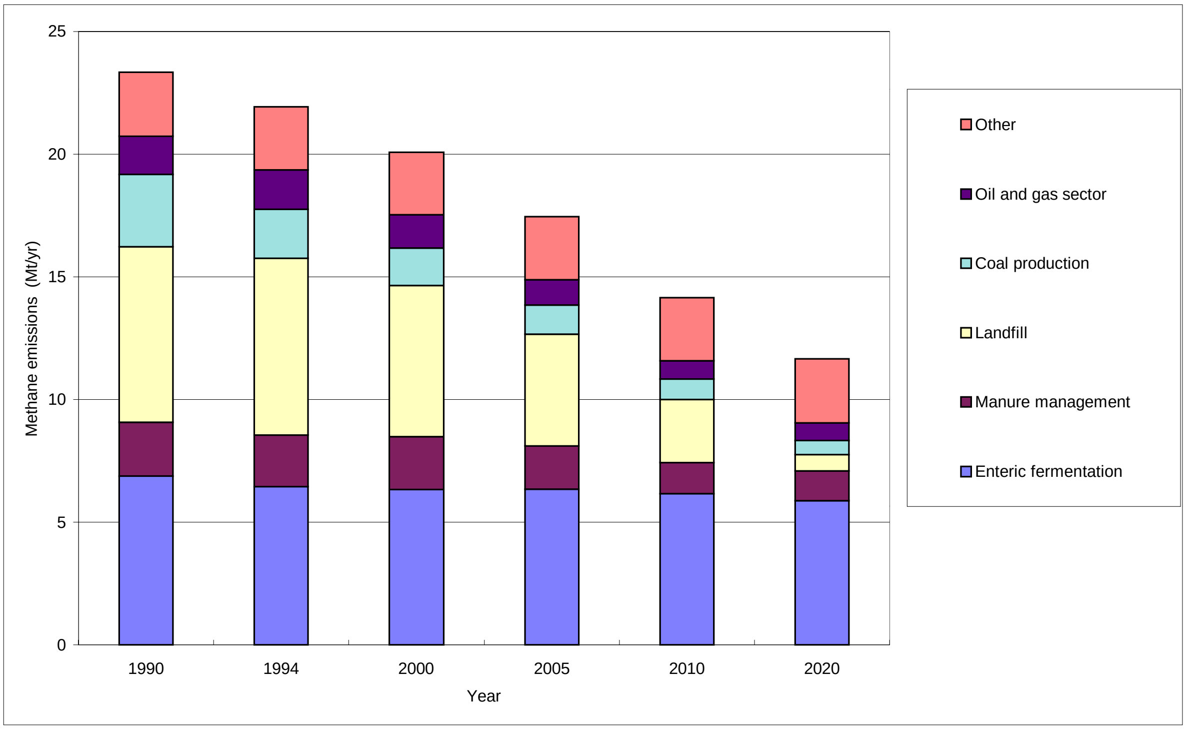
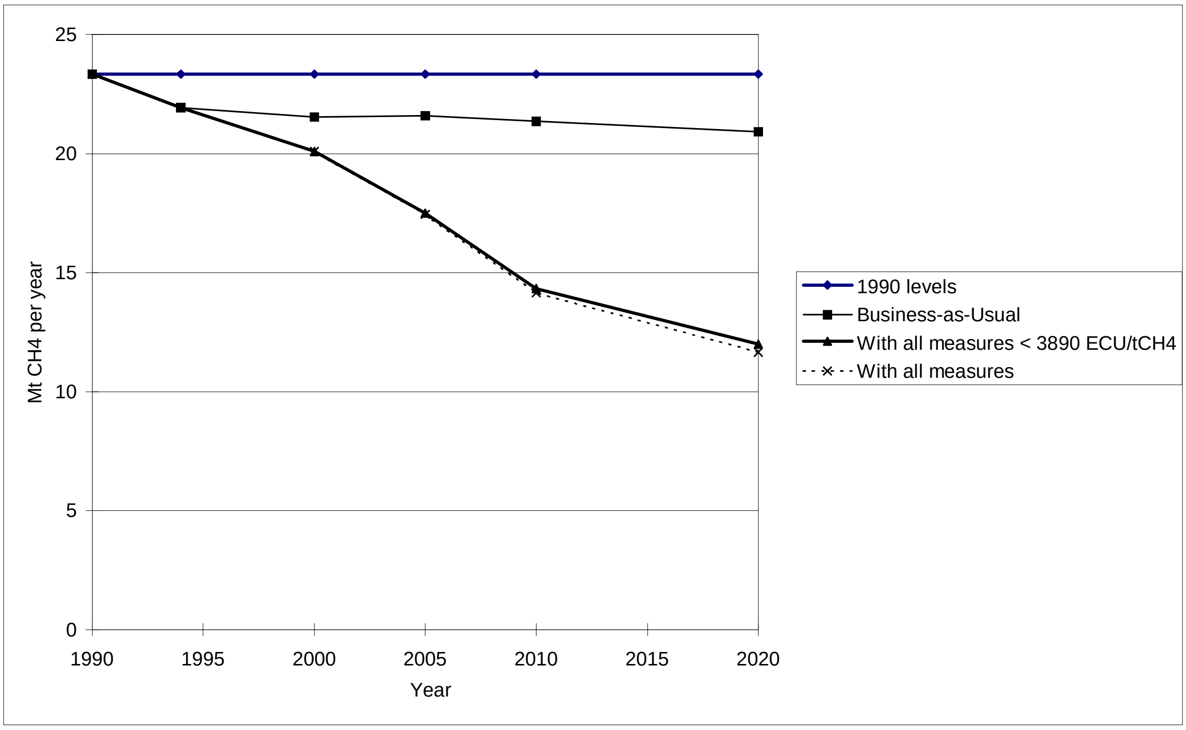


Table 10.4 CH_4 Emissions under a With Measures Scenario (kt/year)

Source Sector	1990	1994	2000	2005	2010	2020
Enteric fermentation	6878	6443	6331	6351	6156	5880
Animal manures	2199	2105	2150	1746	1277	1205
Landfill	7144	7223	6175	4559	2571	667
Coal production	2946	1978	1521	1195	848	580
Oil and gas sector	1564	1612	1351	1043	722	705
Waste water	764	807	814	821	824	825
Other agriculture and land use change	854	841	841	841	841	841
Fuel combustion	564	507	527	540	553	571
Transport	253	221	179	155	152	175
Industrial processes and solvents	20	23	27	30	34	37
Other	165	169	171	173	174	175
Total	**23349**	**21931**	**20086**	**17454**	**14152**	**11661**
Change from 1990	**0%**	**-6%**	**-14%**	**-25%**	**-39%**	**-50%**

Figure 10.7 EU Methane Emissions under a Business as Usual and With Measures Scenario



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Annex A

Effectiveness, Feasibility and Cost of Measures to Reduce Methane Emissions from Livestock in the EU

Sub-contractor's report by ADAS to AEA Technology, 1998

EFFECTIVENESS, FEASIBILITY AND COST OF MEASURES TO REDUCE METHANE EMISSIONS FROM LIVESTOCK IN THE EU

CONTENTS

1.0 INTRODUCTION

The presence of methane in the atmosphere has been known since the 1940's when Migeotte (1948) observed strong absorption bands in the infra-red region of the solar spectrum, which were attributed to the presence of atmospheric methane. Numerous measurements since then have demonstrated the existence of an average temporal increase of atmospheric methane during the period 1980 to 1990 of about 18 ppbv per year (Rodhe, 1990). The current rate of increase in atmospheric methane concentration has subsequently slowed to about 10 ppbv per year (Steele et al, 1992), but the reason for this is uncertain (Watson et al, 1992). The current global average atmospheric concentration of methane is 1720 ppbv, more than double its pre-industrial value of 700 ppbv (Bolle et al, 1986). The concentration of methane in the Northern Hemisphere is about 100 ppbv more than in the Southern Hemisphere, indicating either greater source or lower sink strength in the Northern Hemisphere (Watson et al, 1992).

The rising concentration of methane is correlated with increasing populations and currently about 70% of methane production arises from anthropogenic sources and the remainder from natural sources. Agriculture is considered to be responsible for about two-thirds of the anthropogenic sources (Duxbury et al, 1993). Biological generation in anaerobic environments (natural and man-made wetlands, enteric fermentation and anaerobic waste processing) is the major source of methane, although losses associated with coal and natural gas industries are also significant. The primary sink for methane is reaction with hydroxyl radicals in the troposphere (Crutzen, 1991; Fung et al, 1991), but small soil (Steudler et al, 1989; Whalen and Reeburgh 1990; Mosier et al, 1991) and stratospheric (Crutzen, 1991) sinks have also been identified.

The release of an estimated 205 to 245 million tonnes of methane per year from agricultural sources is derived from enteric fermentation (80 million tonnes), paddy rice production (60 - 100 million tonnes), biomass burning (40 million tonnes) and animal wastes (25 million tonnes) (Watson et al, 1992). The soil sink strength for methane appears to have been reduced by changes in land use, chronic deposition of nitrogen from the atmosphere and alterations in nitrogen dynamics of agricultural soils (Steudler et al, 1989; Keller et al, 1990; Scharffe et al, 1990; Mosier et al, 1991). Ojima et al, (1993) estimated that the consumption of atmospheric methane by soils of temperate forest and grassland eco-systems has been reduced by 30%. Without the temperate soil sink for methane, the atmospheric concentration of methane would be increasing at about 1.5 times the current rate.

Since atmospheric methane is currently increasing at a rate of about 30 to 40 million tonnes per year, stabilising global methane concentrations at current levels would require reductions in methane emissions or increased sinks for methane of approximately the same amount. This reduction represents approximately 10% of current anthropogenic emissions. Three of the main major agricultural sources of methane are flooded rice, enteric fermentation and animal wastes. Decreasing methane emissions from these sources by 10 to 15% would stabilise atmospheric methane at its present level and is a realistic objective (Duxbury and Mosier, 1993).

In 1990, agricultural emissions of methane in the EU-15 were estimated at 10.2 million tonnes and were the greatest source of methane emissions in the EU, amounting to 45% of EU emissions. Of these, approximately two-thirds came from enteric fermentation and one-third from livestock manure. The objective of this review is to identify possible options for reducing methane emissions from both enteric fermentation and animal manures[1] .

2.0 METHANE INVENTORY AND ITS RELIABILITY

This section examines the recommended IPCC methodology for estimating emissions from enteric fermentation and livestock manure and comments on its applicability to EU countries, possible errors and areas where the methodology could be improved.

2.1 Enteric fermentation

2.1.1 IPCC Methodology

The amount of methane that is released from an animal depends on its species, type, age, weight and genetic merit and the type, quality and quantity of the feed and the energy expenditure of the animal. The IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 1996) stipulates that the amount of methane emitted by a population of animals should be calculated by multiplying the emission rate per animal by the number of animals. IPCC suggest that in order to reflect the variation in emission rates among animal types, the population of animals should be divided into sub-groups and an estimate of the emission rate per animal for each sub-group should be made.

The recommended IPCC methodology is a two tier approach:

Tier 1
A simplified approach which relies on default emission factors drawn from previous studies. This approach is likely to be sufficient for most animal types in most countries.

Tier 2
A more complex approach that requires country-specific information on livestock characteristics. For example, cattle characteristics vary significantly by country, therefore countries with large cattle populations should consider Tier 2 approach for estimating methane emissions.

Individual countries are encouraged by the IPCC to go beyond the two tier approach to estimating emissions, when the information is available. The categories recommended for livestock types within Tier 2 approach are as follows:

Cattle	
Mature dairy cattle	Dairy cattle used principally for commercial milk production
Mature non-dairy cattle	Mature females: - Beef cows used principally for producing beef steers and heifers - Multiple-use cows; milk, draft power and other uses

[1] This review forms part of a larger study being carried out by AEA Technology for DGXI, examining options for abating emissions of methane and nitrous oxide in the EU. Its objective is to identify possible options for reducing methane emissions from both enteric fermentation and animal manures. Options relating to the treatment of animal manures such as anaerobic digestion are not considered, as these have already been examined in other parts of the AEA Technology study.

	Mature males:
	- Breeding bulls; used principally for breeding purposes
	- Draft bullocks; used principally for draft power
Young cattle	Pre-weaned calves
	Growing heifers, steers/bullocks and bulls
	Feedlot-fed steers and heifers on high grain diets.

For each of the representative animal types defined the following information is required:

- annual average population (No. of head)
- average daily feed intake (MJ gross energy per day and kg per day of dry matter);
- methane conversion rate (percentage of feed gross energy converted to methane energy).

Generally, average daily feed intakes are not available, so in order to estimate this it is necessary to collect the following data for each representative animal type:

- animal liveweight (kg)
- average daily liveweight gain (kg)
- feeding situation; confined, grazing, free-ranging
- milk production per day (kg)
- average amount of work performed per day (hours)
- percentage of cows that give birth in a year
- feed digestibility (%)

Data on methane conversion rates are not generally available. The following rules are recommended for the methane conversion rates:

- a 6% conversion rate ($\pm$ 0.5% units) is recommended for all cattle except feedlot cattle consuming diets with high inclusion of grain when 4% ($\pm$ 0.5%) is recommended;
- where good quality feed is available (high digestibility and high energy value) the lower bounds of these ranges can be used;
- where poorer feed is available the higher bounds are more appropriate.

The emissions factors for each category of cattle are estimated based on the feed intake and methane conversion rate for the category. Feed intake is estimated based on the feed energy requirements of the representative animals, subject to feed intake limitations. The Net Energy System described in NRC (1984 and 1989) is the recommended starting point for estimates. Emissions estimates are not sensitive to feeding systems (IPCC methodology), so ARC (1984) could equally be used.

Net energy requirements are then converted to gross energy requirements as follows:

$$\text{GE requirement} = \frac{(NE_m + NE_{feed} + NE_l + NE_{draft} + NE_{pregnancy}) * (100/DE\%)}{(NE/DE) + (NE_g/(NE_g/DE))}$$

When DE > 65%

$NE/DE = 1.123 - (4.092x10^{-3} * DE\%) + (1.126x10^{-5} * (DE\%)^2) - 25.4*DE\%$

$NE_g/DE = 1.164 - (5.160x10^{-3} * DE\%) + (1.308x10^{-5} * (DE\%)^2) - 37.4*DE\%$

When DE < 65%

$NE/DE = 0.298 + (3.35x10^{-3} * DE\%)$

$NE_g/DE = -0.036 + (5.35x10^{-3} * DE\%)$

Where DE is digestibility of feed energy.

These data for each animal sub-type are then used to calculate total emissions:

Emissions (kg/year) = ([GE requirement (MJ/d)*Y_m*(365d/year)] / [55.65 MJ/kg of methane]) * animal population

where Y_m is the methane conversion rate expressed in the decimal form.

2.1.2 Potential Improvements to the Method

Weaknesses within this system are the accuracy with which the populations of each sub-group can be defined. For example, in the UK the Ministry of Agriculture carries out an Agricultural Census annually, this provides a relatively accurate measurement of the livestock population, but it is static. Used in the above calculations errors would occur if it is assumed that the number of animals are maintained for 365 days per year.

The main area for discrepancy is the defining of both the animals energy requirements and the methane conversion rate. The conversion of NE requirement to GE is dependent on a regression equation relating NE:DE, this in itself is dependent upon methane energy loss, which is the basis of the methane conversion rate. It may therefore have been more direct to have developed a regression relating NE to methane energy loss and using that to calculate emissions. The error associated with the conversion of NE to GE in the IPCC methodology is not stated.

The recommended methane conversion rates are ill-defined with 6% for dairy cattle, 6.5% for mature females and draft bullocks, up to 7.5% for free-ranging cattle. Calves on forage and replacement/growing cattle have a conversion factor of 6%. In the literature conversion rates for dairy cows (lactating and dry) range from 1.6 to 9.9% (mean 6.3%) (Moe and Tyrell, 1979), or 2.5 to 10.1% (mean 6.4%) (Wilkerson et al, 1995). The mean value is similar to that recommend in the IPCC Methodology, but if it is considered that a dairy cow lactates for 305 d per year and is dry for 60 d per year, it may be more appropriate to use separate conversion rates e.g. 5.5% (min 2.5, max. 7.8) for lactating cows and 7.9% (min 3.5, max. 10.1) for dry cows (Wilkerson et al, 1995) in order to make a more accurate estimate of emissions. Calculating this through using a 550 kg cow producing 5500 l in 305 d plus a 50 kg calf on a diet of 70% DE during lactation and 60% in the dry period, would produce 125kg of methane per year using an emission rate of 6% or 119kg per year with the mean emission rates of Wilkerson et al, (1995) for lactation and dry period. This equates to 5% lower emissions.

Methane emission rates are technically difficult to measure and emissions rates can differ widely from animal to animal and from the use of different types of feed material. The range in methane emission rates reported in the literature emphasise this animal variation and dietary response variation. Wilkerson et al, (1995) reviewed the accuracy of seven published equations of methane prediction using a data file consisting of 16 experiments (602 observations) for Holstein cows. The equation of Moe and Tyrell (1979) using intake of carbohydrate fractions (cellulose, hemicellulose and non-fibre carbohydrate (NFC)) was the most accurate and precise, as follows:

Methane (Mcal/d) = 0.814 + 0.122 NFC (kg/d) + 0.415hemicellulose (kg/d) + 0.633cellulose (kg/d)

$R^2 = 0.67$ $S_{y.x} = 0.62$

The errors of prediction were greater for lactating than non-lactating cows, but if information on carbohydrate composition of feeds is available within member states then it may provide more accurate emission estimates. Methane energy loss as a proportion of gross energy intake is very dependent on feed type, as not all feed energy is the same in its methane producing potential. As fermentation biochemistry suggests that carbohydrates provide most opportunity for methane production and that carbohydrates may differ in the amount of methane produced in the rumen, it seems appropriate that the equation of Moe and Tyrell (1979) could provide more accurate estimates of emissions.

If the accuracy of estimating methane emissions from enteric fermentation is to be improved, further research with each class of livestock will need to be carried out.

2.2 Livestock manure

2.2.1 IPCC Methodology

The same two tier approach to the methodology for estimating methane emissions from livestock is used for livestock manures. For Tier 1 the livestock population should be described by animal type and sub-group within animal type and also by climate - warm, temperate or cool (annual average temperatures >25 ^oC, 15-25 ^oC and <15 ^oC respectively). For each livestock population the fraction in each climate should be estimated. It is then necessary to consider manure management i.e. manure production (based on feed intake and digestibility), methane producing potential (B_o, maximum amount of methane that can be produced from a given quantity of manure), methane conversion factor (MCF) dependent on management and climate and manure management practices.

For Tier 2 calculations animal populations by climatic region are classified, then assigned an average daily volatile solids (VS) excretion (kg OM/d) and a B_o (m^3 of methane per kg VS) under each manure management system. VS may be estimated as follows:
VS (kg OM/d) = Intake (MJ/d) * (1 kg / 18.45MJ) * (1- DE% / 100) * (1-ash% / 100)
VS = VS excretion per day on a dry weight basis.
DE% = the digestibility of feed energy in per cent.
ash% = the ash content of the manure in per cent (DM basis).

The system uses ash contents for cattle and pig manures of 8 and 2% and DE% for pigs of 75%. B_o varies by species and diet but suggested values are: dairy cattle 0.24 m^3/kg VS, non-dairy cattle 0.17 m^3/kg VS and pigs 0.45 m^3/kg VS. Manure management systems categorised in the methodology are pasture, daily spread, solid storage, feedlot, liquid/slurry, anaerobic lagoons, pit storage, anaerobic digester and burned for fuel. MCF range from 0.1% (daily spread) to 90% (anaerobic lagoon).

2.2.2 Potential areas for improvement

Areas which require further research/measurements are feed intakes, digestibility, feed type and the impact of these on the faecal organic matter output from livestock by species. The carbohydrate composition of the faecal organic matter and the storage conditions would enable better estimates of the B_o and MCF.

For example, in a number of studies (Moss, 1997, unpublished) the ash content of faeces from sheep fed grass silage of varying digestibility at the maintenance plane of nutrition ranged from 9 to 44% (DM basis). This range would have a big impact on the estimation of VS (0.168 to 0.243 kg from 1kg DM intake and 70% DE). The GE content of silages in particular also have a large range and are often greater than the standard value of 18.45 MJ/kg DM recommended in the methodology. In the example below, using the methodology calculation for VS with the actual DMI as opposed to using the mean GE concentration, would lower the estimation of VS by 20%. The actual level of VS though was higher than that calculated, particularly when actual DM intake was used.

Example, Grass silage fed to sheep:

DMI=0.7148kg
GE content = 23.3 MJ/kg DM
GE intake = 16.656 MJ/d
Actual faecal output = 0.1896 kg OM/day
DE% = 74.5%
Ash% = 18.47%

Calculation of VS using IPCC methodology:

$$VS(\text{kg OM/d}) = 16.656*(1/18.45)*(1-74.5/100)*(1-18.47/100)$$
$$= 0.1877 \text{ kg OM/d (methodology)}$$

Calculation of VS using actual DM intake:

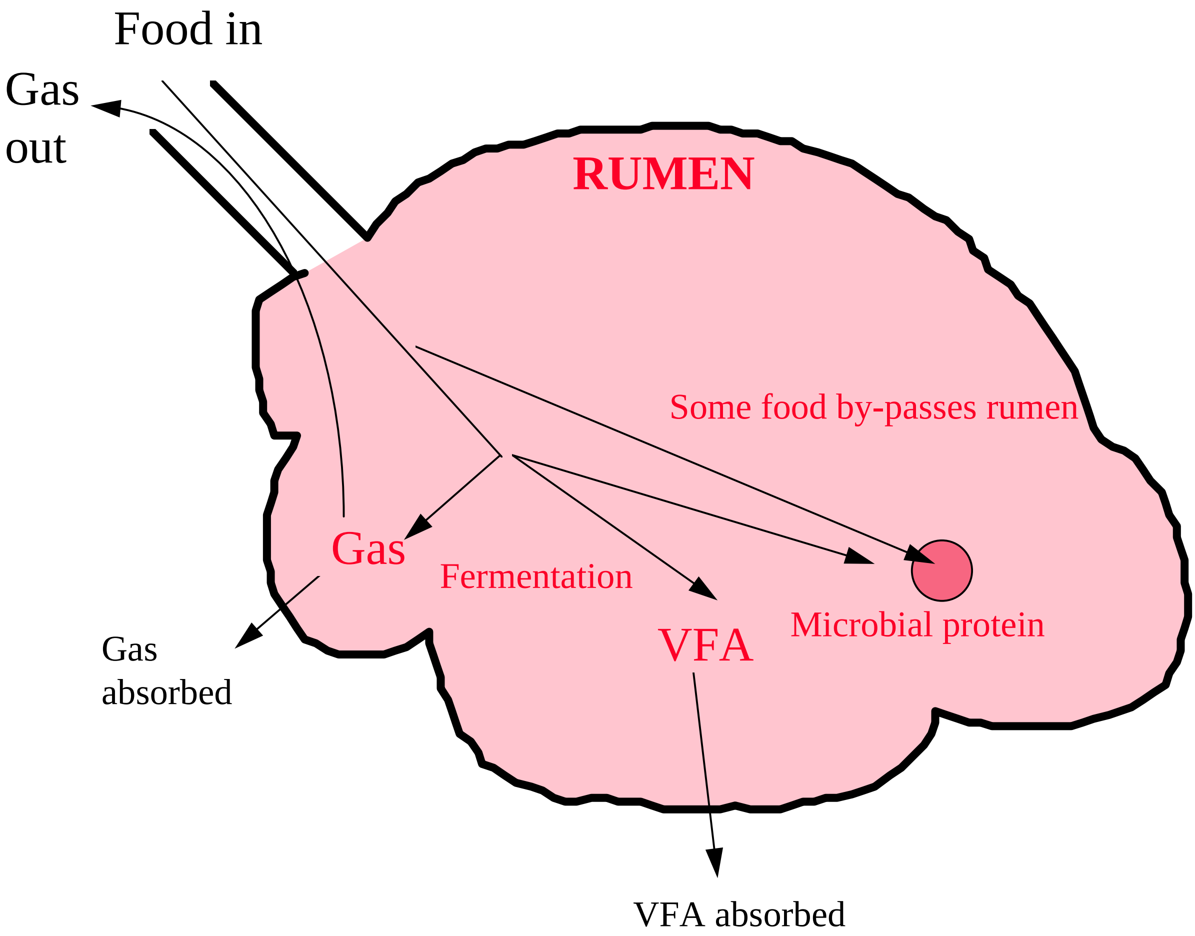
$$VS\ (\text{kg OM/d}) = 0.7148*(1-74.5/100)*(1-18.47/100)$$
$$= 0.1486 \text{ kg OM/d (actual intake).}$$

3.0 METHANE FROM ENTERIC FERMENTATION

3.1 Mechanism

Ruminants have an expanded alimentary tract preceding gastric digestion in the abomasum. In the adult ruminant, the expanded gut (reticulo-rumen, generally termed the rumen) represents about 85% of the total stomach capacity and contains digesta equal to about 10 to 20% of the animal's weight (Campling et al, 1961). Here large amounts of coarse feedstuffs can be retained for a considerable period of time, providing sufficient time for extensive fermentation of the material (Figure 1). Much information on the physiology and structure of the ruminant alimentary canal can be found in Church (1973).

Figure 3.1 Flow diagram of the rumen



The rumen is an ideal habitat for a large and diverse microbial population (Bryant, 1970). The main functions of this group is to degrade plant polymers which cannot be digested by the host enzymes. The material is fermented to volatile fatty acids (VFA), carbon dioxide and methane. These VFAs pass through the rumen wall into the circulatory system and are oxidised in the liver, supplying a major part of the energy needs of the host. Volatile fatty acids may also be directly utilised by the host as building blocks for synthesis of cell material. Fermentation is also coupled to microbial growth and the microbial cell protein synthesised forms the major source of protein for the animal. The gases produced are waste products of the fermentation and are mainly removed from the rumen by eructation (Dougherty et al, 1965). A small proportion of methane is absorbed in the blood and is eliminated through the lungs.

The rumen environment and microbiology has been reviewed and discussed elsewhere (Moss, 1993). Species diversity and the size and activity of the microbial population in the rumen are not constant. In the wild and extensive grazing situation this variation is largely a reflection of seasonal and climatic differences and their effect on the availability, composition and variety of vegetation for ingestion by ruminants. In domesticated ruminants, where conditions are less variable, changes in diet composition, in physical form and amount offered are largely responsible for changes in the microbial population (Thorley et al, 1968; Mackie et al, 1978).

The bacteria are the principal micro-organisms that ferment plant cell wall carbohydrates (Hungate 1966), though the anaerobic phycomycetous fungi may in some conditions be extremely important (Bauchop, 1981). Although absent from plant cell walls, starch is an important component of many ruminant diets, especially those including grain. Some cellulolytic bacteria are also amylolytic (e.g. *Bacteroides succinogenes*), however the principal amylase-producing bacteria have a limited ability to utilise other polysaccharides. It is therefore necessary for populations of both cellulolytic and amylolytic bacteria to be maintained when a mixed forage: concentrate diet is present in order to maximise rumen efficiency.

The majority of the rumen micro-organisms use the Embden-Meyerhof-Parnas and pentose phosphate pathways to ferment the hexose and pentose (simple sugars) products of polysaccharide degradation to pyruvate. Pyruvate can then be metabolised in a number of different ways to various end-products, including formate, acetate, propionate, butyrate, lactate, succinate, methanol, ethanol, carbon dioxide and hydrogen. In the rumen ecosystem, however, some of these compounds are present in only trace amounts as they are utilised as substrates for growth by secondary micro-organisms. For example, lactate and succinate utilising species convert these compounds to acetate or propionate. Rumen methanogenic archaebacteria utilise either hydrogen and carbon dioxide or formate, acetate, methylamine and methanol for production of methane. The involvement of these bacteria in inter-species hydrogen transfer (maintaining low partial pressure of hydrogen within the rumen) is an important interaction which alters the fermentation balance and results in a shift of the overall fermentation from less-reduced to more-reduced end-products (Wolin, 1974). While methanogens can convert acetate to methane and carbon dioxide, this pathway for methane production in the rumen is of minor importance. The major substrates for methanogens are hydrogen and carbon dioxide or formate.

Protozoa are able to degrade all the major plant constituents, but much of their energy and nitrogen requirements are derived from the phagocytosis of other microbes. The role of protozoa in the rumen is not clear, animals without a protozoal population (defaunated) remain healthy, however protozoa can form a significant proportion of the microbial biomass apparently selectively retained within the rumen (Michalowski et al, 1986). The size and the effects of the protozoal population on the overall rumen fermentation will be dependent on diet type, animal and rumen fauna. A specific interaction between protozoa and episymbiotic methanogens has been observed by microscopy (Stumm et al, 1982; Krumholz et al, 1983; Stumm and Zwart, 1986). This association was confirmed by physical separation and washes which showed 70% of total methanogenesis was associated with the protozoa (Table 3.1).

Table 3.1 Methanogenic activity of fractions prepared from rumen fluid (Source: Krumholz et al, 1983)

Fraction	Methanogenic activity (nmol methane/min/mol rumen fluid)
Strained rumen fluid	6.9±0.4
Top fraction	14.9±1.9
Middle fraction	3.5±0.2
Protozoal fraction	41.5±1.4

The methanogens utilise the hydrogen produced during protozoal metabolism to convert it into methane which is the major electron acceptor in the rumen. When the protozoal population is removed there is lower hydrogen production from this source and this may be responsible for the observed decrease in energy losses in the form of methane (Vermorel and Jouany, 1989). Other workers have shown defaunation (removal of the protozoal population from the rumen) to increase methane production when a hay-only ration was offered (Itabashi et al, 1984).

4.0 OPTIONS FOR REDUCING METHANE EMISSIONS FROM ENTERIC FERMENTATION

The European Commission has already produced a strategy paper for reducing methane emissions and has considered three main areas: reducing livestock numbers, improving rumen fermentation efficiency and improving productivity.

4.1 Reduction of livestock numbers

Trends in animal numbers are dependent on both the EU's Common Agricultural Policy (CAP), and national policies. CAP was substantially reformed in 1992, with the intention of moving from subsidies towards market based prices, and the recent Agenda 2000 document (Reference?? 1997) has indicated that further reforms in this direction are likely in the future. Table 4.1 shows projections of livestock numbers for 2010 compiled by IIASA and based on national information and a number of other studies (Amann et al, 1996)[2].

Overall, in the EU-15, numbers of cattle, sheep and pigs are projected to decline, whilst numbers of poultry will increase slightly. The impact of the projected cattle population decline would be to reduce total emissions of methane. This response is complicated however when productivity and output of product are considered. Total emissions of methane from an animal are dependent on the animals size and feed intake, with methane release in litres per day increasing with increasing body size and intake. Therefore if animal numbers are reduced the methane release will also decline. It is possible to consider methane release in terms of litres per animal per day or in productivity terms as litres of methane per unit of useful animal product (milk, meat or wool). If an animal's productivity increases (i.e. more milk per animal) total methane release from the animal will increase compared with an animal of the same size producing less product but the amount of methane per unit product will decline. It is therefore beneficial in terms of reducing methane emissions to produce the same amount of product from less animals. If animal numbers decline as forecast by 2010 and animal productivity remains static then methane emissions will reduce by x amount, but if animal productivity continues to rise as has been noted for milk production over the previous 10 year period and total output is continued to be controlled by quotas then emissions per unit product will decline. The combination of lower animal numbers and increased productivity for the same total useful product output will likely decrease emissions of methane further.

From Table 4.1 it can be seen that countries projected to increase their cattle populations are Belgium, Ireland, Italy, Spain and Sweden. Generally these countries have the lower productivity and may not be self-sufficient in cattle products. In order to sustain the increased population of cattle with limited land resource, productivity will have to increase, which would reduce methane emissions per unit product. It is apparent that Belgium, Luxembourg and

[2] These projections are being used by IIASA in work on a number of environmental problems (acidification, ground level ozone) for DGXI of the Commission and are also being used in the AEA Technology study as a basis for estimating agricultural emissions under a business as usual scenario.

Table 4.1 Projection of livestock numbers up to the year 2010 for the EU-15 (million animals) (Source: Amann et al, 1996)

	COWS[1]			SHEEP			PIGS[2]			POULTRY[3]		
	1990	2010	Change %	1990	2010	Change %	1990	2010	Change %	1990	2010	Change %
Austria	2.6	2.5	- 1	0.323	0.231	-28	3.8	4.5	20	14.0	17.3	23
Belgium	3.0	5.1	68	0.163	0.110	-33	6.4	4.7	- 26	35.3	27.1	- 23
Denmark	2.2	1.7	- 23	0.100	0.080	-20	9.3	11.7	26	16.2	17.1	5
Finland	1.4	0.9	- 34	0.065	0.070	8	1.3	1.2	- 11	6.0	4.5	- 25
France	21.4	20.9	- 3	12.219	11.000	-10	12.4	17.4	41	236.0	279.3	18
Germany	20.3	15.7	- 23	4.213	3.230	-23	34.2	21.2	- 38	125.5	78.6	- 37
Greece	0.6	0.6	- 1	13.994	15.161	8	1.0	1.5	46	27.4	33.0	20
Ireland	5.9	7.7	31	5.791	6.000	4	1.0	1.9	93	8.9	13.6	52
Italy	8.7	9.5	9	12.094	11.716	-3	9.3	10.5	13	161.0	204.1	27
Luxembourg	0.2	0.4	78	0.007	0.005	-29	0.1	0.1	- 33	0.1	0.1	- 28
Netherlands	4.9	4.8	- 2	1.737	1.340	-23	13.4	11.2	- 16	93.8	79.5	- 15
Portugal	1.3	1.2	- 7	6.424	7.355	14	2.5	1.5	- 41	21.9	26.8	22
Spain	5.1	5.3	3	27.700	26.577	-4	16.0	21.4	34	51.0	56.1	10
Sweden	1.7	1.9	10	0.406	0.483	19	2.3	2.1	- 7	12.3	9.0	- 27
UK	11.9	9.9	- 17	29.678	24.160	-19	7.4	4.8	- 34	141.0	120.5	- 15
EU - 15	*91.4*	*88.2*	*- 4%*	*114.91*	*107.52*	*-6%*	*120.3*	*115.6*	*- 4%*	*950.5*	*966.5*	*2%*

1. Cows, include dairy cows & other cattle
2. Pigs, include fattening pigs & sows
3. Poultry, include laying hens, broilers and other poultry

Sweden are switching livestock enterprises from pigs and poultry to cattle, whereas the other countries (Ireland, Italy and Spain) are expanding all livestock enterprises.

A danger of artificially implementing the reduction of livestock numbers, using a blanket reduction over all member states is that more of the product (milk, meat etc.) is produced less efficiently with a resultant increase in methane emissions per unit product. Currently for milk production there is a range of productivity across the EU member states with Greece producing less than 4000 l per cow per year compared with Denmark producing in excess of 6000 l per cow per year (Eurostat 1995). Dairy productivity is increasing though for all member states. If market forces were allowed to influence livestock products then production of each product would shift to the most efficient producers which would reduce animal numbers and reduce methane emissions for a given output. Under the current subsidy system a reduction of livestock numbers would not be a popular choice for farmers as subsidies are paid per head of stock. It is presently not clear what the impact of Agenda 2000 will be on animal numbers and productivity as it is yet to be ratified by member states. There will be a further impact if the EU is enlarged to include the 11 countries presently interested in becoming members.

In order to reduce the 1990 level methane emissions from enteric fermentation within the EU-15 by 15% with existing productivity levels, animal numbers would have to be reduced by 15%. If the reduction was based on methane emissions per unit product then animal numbers would not have to be reduced so significantly.

4.2 Improving rumen fermentation efficiency

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: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. Hoover and Stokes (1991) reviewed the current literature on sources and levels of carbohydrates and proteins that have been found to affect rumen microbial growth. They concluded that the most suitable sources and quantities needed to support maximum growth have not been determined. The major conclusions were that the total population of rumen microbes achieves the highest growth rate on mixtures of peptides, amino acids and ammonia as their nitrogen source and the rate of fermentation of total carbohydrate is directly related to the proportion of starches, pectins and sugars.

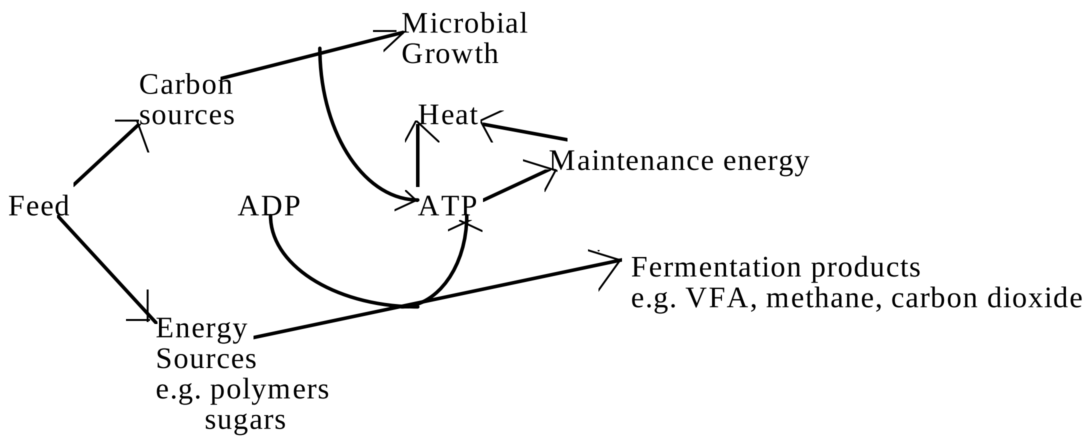
Many workers have developed models to predict microbial growth which prove relatively successful on the developmental data set, but when validated with a blind set of test data their accuracy diminishes. Feeding ruminants on diets containing high levels of readily fermented non-structural carbohydrate can minimise methane production by reducing the protozoal population and lowering rumen pH. This though can give rise to an overall depressed ruminal fermentation. This may lower the conversion of feed energy into animal product and could be detrimental to the animal's health (acidosis, laminitis, poor fertility). Therefore, using diets with extreme nutrient compositions is not a particularly successful or sustainable method to control methane emissions from ruminants.

4.2.1 Hexose partitioning

As already established, during rumen fermentation, feedstuffs are converted into short-chain acids, ammonia, methane, carbon dioxide, cell material and heat. Animal performance is dependent on the balance of these products and this balance is ultimately controlled by the types and activities of micro-organisms in the rumen.

The VFAs are used by the animal as an energy source while the microbes serve as an important source of amino acids for protein synthesis. Ammonia, methane and heat by contrast represent a loss of either nitrogen or energy unavailable to the animal. Microbial energy transformations determine not only the quantity and composition of the fermentation acids, but the amount of biomass present. Figure 4.1 gives a schematic diagram of the partition of energy.

Figure 4.1 A schematic diagram showing the fermentation of feed by rumen micro-organisms and the partition of the feed energy among the various products.



It is the amount of energy available in the form of adenosine triphosphate (ATP) in combination with sufficient precursors necessary for microbial growth, that will influence the efficiency of microbial cell synthesis in the rumen. A useful measure of the efficiency of microbial cell synthesis is the yield of cell DM (g) per mol of ATP from fermented OM (Y_{ATP}) and is often found to be less than 12, whereas the theoretical maximum if all ATP is used for cell synthesis is 20 to 30 depending on the nature of the starting materials and the composition of the cells formed (Hespell and Byrant, 1979). The potential for manipulating the rumen system to increase Y_{ATP} therefore rests in reducing ATP energy dissipation in maintenance, energy spilling or extracellular recycling processes.

A number of workers have described models of microbial fermentation and growth in the rumen (Czerkawski, 1986; Hungate, 1975; Baldwin and Denham, 1979; Leng, 1982). Nolan and Leng (1989) put forward a simple model to simulate feed OM digestion in the rumen and to predict volatile fatty acid and cell yield rates of production of carbon dioxide and methane (Figure 4.2). This model assumes that the molar proportions of the individual VFAs produced remains constant over the range of Y_{ATP} and as a result it can be concluded that the quantity of microbial cells leaving the rumen per unit of carbohydrate consumed may have a large effect on the overall methane production (Table 4.2). Using the relationships provided by Baldwin et al, (1970) for hexose fermentation by different pathways, Beever (1993) concluded that changing the nature of the fermentation would result in an overall reduction in methane production from 17 mol/day (high forage) to 10.6 mol/day (high concentrate) as a consequence of increased propionate production whilst changing the partition of hexose utilisation towards synthesis at the expense of fermentation caused overall estimates to decline from 17.9 to 11.4 mol/day. In both situations it is accepted that biological credibility for such a range of Y_{ATP} or hexose partition has not been established.

Figure 4.2 Simulation of the effect of increasing Y_{ATP} on microbial cell dry matter and volatile fatty acid yield and carbon dioxide and methane production from true digestion (OMTDR) of 1kg of polysaccharide in the rumen (after Nolan and Leng, 1989).

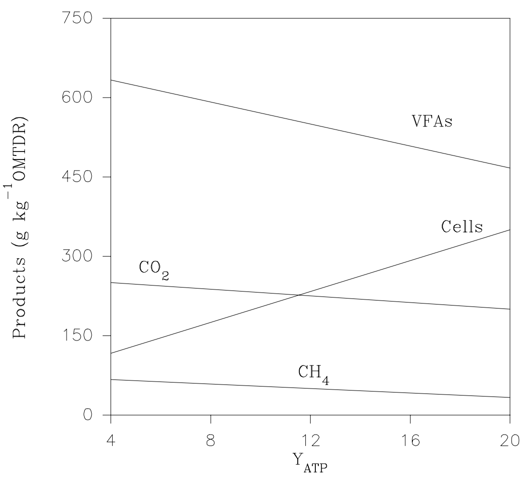


Table 4.2. Effect of the partition of hexose between microbial synthesis and fermentation upon the yield of methane under three contrasting dietary types in the dairy cow (potential hexose available = 40 mol/day) after Beever, 1993).

	Hexose partition					
	0.4:1	0.6:1	0.8:1	1:1	1.2:1	Mean
Hexose fermented (mol/day)	35.0	30.6	27.2	24.5	22.3	
Methane yield (mol/day)from:						
High forage diet	21.4	18.7	16.6	14.9	13.6	17.0
High-cereal diet	13.3	11.6	10.3	9.3	8.5	10.6
High-molasses diet	18.9	16.5	14.7	13.2	12.0	15.1
Mean	*17.9*	*15.6*	*13.9*	*12.5*	*11.4*	

For an average dairy cow using this theoretical basis, annual methane production could for the mean of three diets be reduced from 104.5 kg methane/head/year to 66.6 kg which is a reduction of 36%. At a given hexose partitioning level, changing from a high forage to a high cereal diet would reduce methane emissions from 99 kg/head/year to 62 kg which is a 35% reduction. It can be seen that manipulating the partition of feed carbohydrate directly into microbial growth as opposed to fermentation could have a big impact on methane emissions. The experimental evidence for this in the literature is scarce and needs to be verified with further research.

The status of this option, is that it is still subject to research. The Ministry of Agriculture, Fisheries and Food in the UK is currently funding a research project to investigate *in vitro*, carbohydrate sources that provide improved hexose partitioning and to use this information to design diets with enhanced hexose partitioning for testing *in vivo* to determine the impact on methane emissions.

Theoretically this technology should also enhance protein utilisation and hence reduce ammonia emissions. This option, if proved in practise to reduce methane emissions, would be applicable to all member states though it would be more readily implemented for livestock receiving supplementary concentrates. It would be less applicable to grazing/free-ranging livestock. The cost of implementing the option would be minimal as the overall effect would be increased productivity which would likely offset any additional feed costs associated with the option.

4.2.2 Propionate precursors

Within the rumen, methane represents a terminal hydrogen sink. Propionate production represents an alternative hydrogen sink in normal rumen fermentation, provided sufficient precursors are available. The pathway of carbohydrate fermentation is to glucose units which then enter the citric acid cycle where they are broken down to the main volatile fatty acids (acetate, butyrate, propionate) and other products such as lactate, ethanol and methane. The main precursors to propionate within this cycle are pyruvate, oxaloacetate, malate, fumarate, and succinate, or alternatively directly from pyruvate to propionate via the acrylate pathway (high concentrate diets). Any of these organic acids may promote alternative metabolic pathways to dispose of reducing power and hence reduce methane production.

Existing research shows this to be a very promising approach. Workers in the UK and Spain (Newbold 1998, personal communication) have studied *in vitro* the effect of different concentrations of fumarate in the rumen fermentation of a 50:50 forage:concentrate ration. The addition of fumarate up to 80mM resulted in higher total gas production, with a linear decrease in methane production without the significant accumulation of hydrogen. This reduction accounted for up to a 24% reduction in methane production. This reduction was maintained as long as the addition of fumarate was continued. The decrease in methane production was equivalent to the increase in propionate production in terms of reducing equivalents. An additional benefit to this option was that the number of cellulolytic bacteria were significantly increased and this resulted in an increased degradation of the diet over a 48h period.

Other workers (Callaway and Martin, 1996) have considered the effect of fumarate and malate on rumen fermentation *in vitro*. Malate addition not only acted as an alternative hydrogen sink, like fumarate, but also buffers the ruminal contents by a dual mechanism of reducing lactate accumulation and increasing carbon dioxide production. It is essential, particularly with high concentrate diets that the disposal of ruminal lactate is efficient to avoid a severe decline in rumen pH. Organic acids like malate seem to encourage increased lactate utilisation by the predominant ruminal anaerobe *Selenomonas ruminantium* (Nisbet and Martin, 1990), stimulating its growth *in vitro* (Nisbet and Martin, 1993). There is evidence *in vivo* that addition of malate may improve animal performance in dairy cows (Kung et al, 1982) and in beef cattle (Sanson and Stallcup, 1984; Streeter et al, 1994).

Malate can be purchased in bulk quantities at approximately 2,200 ECU per tonne (80 g per day for feedlot cattle), which would probably preclude it as an economic feed additive for dairy cows. Its use to reduce methane production in the future would be dependent on a reduction in production cost of malate (or any of the other organic acids considered) or a subsidy for its use.

A possible alternative approach is the identification of naturally occurring malate. Plants are a rich source of nutrients that can be utilised by both the ruminant animal and the mixed ruminal microbial population. Intermediates of the citric acid cycle accumulate in plant tissue and may represent as much as 10% of the dry matter of grasses (Stout et al, 1967) and 7.5% of the dry matter of lucerne (Callaway et al, 1997). In all cases the concentration of malate declined with increasing plant maturity. Malate is found primarily in leaf tissue of plants because of the activity of the citric acid cycle (Vickery and Pincher, 1940), thus plant breeders could select for high concentrations of malate by selecting varieties with higher leaf to stem ratios. Concentrations of malate found in forages could be sufficient to provide adequate concentrations of malate to stimulate lactate utilisation by *Selenomonas ruminatium* and hence reduce methane production, but considerable research is still required.

Naturally occurring precursors to propionate production in rumen fermentation would be a more readily acceptable feed additive than antibiotic or chemical agents. It is possible that concentrations of these organic acids could be naturally enhanced in plant material by selection in plant breeding programmes.

It is estimated that if successful, the option could reduce methane emissions by up to 25%, but that there could be other benefits to the livestock industry such as improved feed degradation which would likely reduce feed costs, which as feed accounts for 60% of variable costs for livestock, then any saving in this area would be beneficial. Another possible benefit would be a reduced incidence of acidosis in high producing dairy cows and intensively reared cattle, the current cost of which is not available, but could be considerable.

The option of the use of the organic acids as daily supplements to reduce methane would only be practicably available to livestock receiving supplementary concentrates in a controlled manner. If concentrations of these organic acids in forages could be increased then the option would be available to all ruminant livestock.

4.2.3 Direct fed microbials

4.2.3.1 Acetogens

In the hindgut of mammals and termites, acetogenic bacteria produce acetic acid by the reduction of carbon dioxide with hydrogen (Ljungdahl, 1986). This process, referred to as reductive acetogenesis, acts as an important alternative hydrogen sink in hindgut fermentation (Demeyer and de Graeve, 1991). Acetogenic bacteria are present in high numbers in the rumen of day old lambs (Morvan et al, 1994) but appear unable to support large populations in the rumen of adult ruminants. Although hydrogen and carbon dioxide utilising acetogenic bacteria have been isolated from the rumen of sheep and cattle, under normal conditions they grow fermentatively on organic substrates (Genthner et al, 1981).

The problem that needs understanding is why acetogenesis outcompetes methanogenesis in the human colon but does not in the rumen. This would require an evaluation of factors which affect the successful competition for hydrogen. Acetogenesis is thermodynamically possible under the *in situ* conditions of the rumen and could theoretically take place (Mackie and Bryant, 1994). The affinity of acetogenic bacteria for hydrogen tends to be lower than that of methanogens and ruminal hydrogen concentrations tend to be too low (Mackie and Bryant, 1994). However the hydrogen threshold value varies widely according to the strain and some acetogenic strains have a hydrogen threshold value comparable to those of methanogenic bacteria (Boccazzi et al, 1994). More research is required to understand the ecology and physiology of acetogenic bacteria and to devise practical solutions to their survival in the rumen and hence the displacement of methanogenic bacteria. This would not only decrease the contribution of ruminants to methane emissions, but would increase the efficiency of ruminant production.

This option is still at the research stage and research groups in Belgium, France, Spain and the UK are active in this area. An alternative approach would be to screen a range of acetogenic bacteria for their activity in rumen fluid and then to circumvent the competition by methanogens in the rumen by introducing the acetogens into the rumen on a daily basis in a manner analogous with current probiotic preparations.

If successful, this option has the potential to eliminate or reduce to a minimum methane emissions from ruminants. Emissions of ammonia may be reduced as a result of more efficient carbohydrate fermentation which requires nitrogen. The option would again be applicable to all ruminants receiving supplements on a controlled and regular basis. The costs associated with isolating, growing and preparation of this type of micro-organism are not clear, but some of these costs will inevitably be offset by improved rumen efficiency.

4.2.3.2 Methane oxidisers

An analogous approach to that reported in the previous section is the use of methane oxidisers as direct-fed microbial preparations. Methanogenesis is a strictly anaerobic process, while methane oxidation has been noted in marine sediments and in some saline inland waters (Iversen and Jogensen, 1985; Oremland et al, 1993), However, aerobic methane oxidation seems to be by far the most important mechanism within known methane sinks (Cicerone and Oremland, 1988).

The rumen is essentially an anaerobic environment though some oxygen does enter the rumen (Czerkawski, 1969) and detectable levels of dissolved ruminal oxygen have been found *in situ* shortly after feeding (Hillman et al, 1985), thus, it is possible that even aerobic methane oxidation occurs in the rumen. Methane-utilising bacteria have been isolated from the rumen, but their role in the rumen was never established. *In vitro*, methane oxidation has been shown to represent 8% of methane synthesis in rumen fluid.

Methane oxidisers from gut and non-gut sources should be screened for their activity in rumen fluid *in vitro*. The interference by methanogens could be circumvented by introducing the methane oxidizers into the rumen on a daily basis in a manner analogous with current probiotic preparations.

If successful, this option has the potential to reduce methane production in the rumen by a minimum of 8%. The option would be available to all ruminants receiving supplements on a controlled and regular basis. The costs associated with isolating, growing and the preparation of this type of micro-organism are not clear.

4.2.4 Genetic engineering

Armstrong and Gilbert (1985) suggested the potential use of recombinant deoxyribonucleic acid (DNA) technology to modify the fermentation characteristics of rumen micro-organisms. Examples of application include an enhanced cellulolytic activity in the rumen biomass for forage fed animals to increase their supply of VFAs and amino acids; and a reduction in methanogenesis accompanied by an alternative hydrogen sink through increasing propionate production. Intensive research programs have been directed towards producing new strains of rumen bacteria (Forsberg et al, 1986) that may lead to greater fermentation of cellulose, hemicellulose and perhaps lignin, bacteria that have a lower maintenance requirement and thus improved efficiencies of growth.

Little progress has been made with the above area of research. Progress is however being made in the direct study of rumen methanogens. The size and species composition of methanogenic populations could be estimated with gene probes. Genes encoding the subunits of CH_3-S-CoM are highly conserved among methanogens (Weil et al, 1989) and are a good choice for DNA probes. Application of further biotechnology techniques would enable the quantification and identification of the methanogenic species present. Once developed, these could be further used to regulate the expression of specific genes in these methanogens which may provide additional means of altering ruminal methane production.

The status of these options is still subject to substantial research. If these options are developed, there may be considerable opposition to the release of genetically engineered organisms into the environment. Currently, genetically modified organisms available such as genetically modified maize grain which contains resistance to an insect pest has been banned by some member states (e.g. Austria) and the whole area has raised considerable debate. Therefore, the acceptance of genetically modified rumen micro-organisms does not at the present time seem realistic.

4.2.5 Immunogenic approaches

A team of researchers at CSIRO in Australia have made an application for a world wide patent under the following title: Immunogenic preparation and method for improving the productivity of ruminant animals.

The patent describes a method of improving the productivity of a ruminant animal by administering to the animal an immunogenic preparation effective to invoke an immune response to at least one rumen protozoan. The removal of one species of protozoan from the rumen will invoke the improvements in productivity associated with defaunation (improved protein: energy ratio of the nutrients available for absorption). It is also believed that by modifying the activity of the rumen protozoan, there will be an indirect effect on the activity of methanogens, due to their commensal relationship with rumen protozoa. Therefore, by reducing the protozoal population, there may also be a corresponding effect on the production of methane. The patent also proposes that a vaccine could be prepared directly incorporating antigens from one or more species of methanogenic bacteria as well as the protozoa. This would therefore reduce the animals production of methane still further.

Data from this work are not yet published but it is anticipated that methane production could be reduced by as much as 70%. The long term prospects of this approach are not yet available but areas to be considered are the longevity of the immunisation and whether other species of protozoa and methanogens will increase their populations to compensate for those species where immunisation has taken place.

If this option develops successfully, it could be applied to the whole ruminant population. The costs associated with the approach could be high initially due to the monopoly associated with patents. The increased protein utilisation associated with defaunation would mean reduced emissions of ammonia and increased animal productivity.

4.3 Increase animal productivity

The concept of increasing animal productivity to reduce methane emissions from ruminants is based on the maintenance of overall production output and as a result, increased production of useful product would mean methane production per unit product would decline. A reduction in total emissions of methane would only result if total output levels (e.g. total milk or beef produced) remained constant and livestock numbers were reduced. Possible options for increasing ruminant productivity are discussed in the following sections.

4.3.1 Probiotics

The most widely used microbial feed additives (live cells and growth medium) are based on *Saccharomyces cerevisiae* (SC) and *Aspergillus oryzae* (AO). Their effect on rumen fermentation and animal productivity are wide ranging and this has been reviewed recently by several authors (Martin and Nisbet, 1992; Newbold, 1992). There is very limited information on their effect on methane production and all of this is *in vitro*. AO has been seen to reduce methane by 50% (Frumholtz et al, 1989) which was directly related to a reduction in the protozoal population (45%). There is a close association of methanogens with ciliate protozoa, so the effect of the AO preparation on rumen protozoa leads to a reduction in methane production. On the other hand, addition of SC to an *in vitro* system reduced the methane production by 10% initially, though this was not sustained (Mutsvangwa et al, 1992). In other experiments with AO and SC, an increase in methane production has been reported (Martin et al, 1989; Martin and Nisbet, 1990).

To date, there are no convincing data relating probiotics and methane production, and hence considerably more research is required before it can be concluded that yeast cultures or AO extracts decrease methane production *in vivo*.

Probiotics are already widely available in the EU to improve animal productivity but the implications of this on methane production are unclear. If research proved probiotics to be effective at reducing methane output, existing users would continue with their use, but further uptake would require incentives, as many producers are already apparently sceptical about the benefits of probiotics. The costs of probiotics are about 2100 ECU per tonne (inclusion 50g per head of adult cattle per day).

The technology and the products are already available within the EU, but the research is required to establish their effect on methane production. Response to probiotics has been recorded in dairy cows and growing cattle. From an analysis of published results from more than 1000 cows, Wallace and Newbold (1993) calculated that yeast culture stimulated milk yield by 7.8% and from 16 trials using growing cattle, they showed an average increase in livewieght gain of 7.5%. If there is little effect on methane production *per se*, there would be a resultant reduction in methane production per unit of production. The benefit of this is only seen if animal numbers are reduced correspondingly.

4.3.2 Ionophores

Ionophores are molecules with "backbones" of various structures that contain strategically spaced oxygen atoms. The backbone is capable of assuming conformations that focus these oxygen atoms about a ring or cavity into which a cation may fit. Hence their biological activity is related to their ability to modulate the movement of cations such as sodium, potassium and calcium across cell membranes (Pressman, 1976). Ionophores tend to act upon the gram-positive bacteria and not directly on methanogens. Ionophores are known to adjust several pathways of fermentation with the direct result of increasing propionate production and they are also known to decrease the degradation of an exogenous amino acid load and to lower rumen ammonia levels.

The main ionophores (monensin, lasalocid, salinomycin) in use have shown improved feed efficiency by reducing feed intake and maintaining weight gain or by maintaining feed intake and increasing weight gain. Monensin, an ionophore used widely in beef fattening, inhibits methane production *in vivo* by an average of about 25% (Van Nevel and Demeyer, 1992). Some long term *in vivo* trials have shown that this inhibition of methanogenesis by monensin and lasalocid is not persistent (Rumpler et al, 1986; Johnson et al, 1991). The effect of salinomycin, however, on methane production seemed more persistent (Wakita et al, 1986).

The use of ionophores gives rise to improved animal productivity (on average an 8% improvement in feed conversion efficiency (Chalupa, 1988)) and a possible direct reduction in methane production. This option is in use throughout the EU for beef animals only as its use is not permitted in dairy cows because the product requires a withdrawal period which is not possible with dairy cows. Its effect is therefore impacting on less than 50% of the methane emissions. As with all measures that reduce methane production by increasing animal productivity, the benefit is only seen if animal numbers are reduced correspondingly.

The use of chemicals/antibiotics to increase animal productivity are increasingly becoming unpopular to the consumers of animal products. It is therefore envisaged that the use of ionophores to reduce methane production is not a sustainable option.

4.3.3 Bovine somatotropin

Bovine somatotropin (BST) is a genetically engineered metabolic modifier approved for use in some countries to enhance milk production from dairy cows. BST does not affect digestibility, maintenance requirements or the partial efficiency of milk synthesis, nor does it act directly on the mammary gland. BST affects mammary tissue indirectly by its action on the liver and the kidney to stimulate production of insulin-like growth factors which act on the mammary gland to increase milk synthesis. Nutrients for increased milk yield are provided by increased intake and co-ordination of metabolism to increase supplies to the mammary gland of glucose, amino acids and fatty acids (Chalupa and Ghalligan, 1989). Given a 15% increase in productivity per animal, there would be a reduction in methane production per unit product. Again, this is not a popular consumer choice for enhancing animal productivity and is actually banned by all EU member states. It is therefore not worthy of pursuance.

4.3.4 Forage type and supplementation

Supplementing forages whether of low or high quality, with energy and protein supplements, is well documented to increase microbial growth efficiency and digestibility (see Moss, 1994 for a review). Milk and meat production will increase as a result. The direct effect on methanogenesis is still variable and unclear, but indirectly, methane production per unit product will decline. The area was recently reviewed (Moss, 1994). Increasing the level of non-structural carbohydrate in the diet (by 25%) would reduce methane production by as much as 20%, but this may result in other detrimental effects e.g. acidosis, laminitis, fertility problems. Also with the implementation of quotas for milk production in the EU, many producers are optimising milk production from home-grown forages in order to reduce feed costs. Supplementing poor quality forages and chemically upgrading them are good options for increasing productivity and in turn reducing methane emissions per unit product. Reductions of total emissions would only result if livestock numbers are reduced correspondingly.

Feeding of ruminants to optimise rumen and animal efficiency is a developing area (new feeding systems) and the efficient deployment of this information to all livestock producers would benefit the environment in terms of both methane and nitrogen emissions. This would lead to best practise information and would require good technology transfer. Many farmers within the EU have to pay for unbiased nutritional advice. If this advice was freely available, there would likely be an increase in productivity and an improvement in the impact of emissions from livestock into the environment. For some of the more productive member states (e.g. Denmark and the Netherlands for milk production) this approach may not be so beneficial.

4.3.5 High genetic merit dairy cows

Improving the genetic merit of dairy cows has escalated in the last decade with the import of Holstein genetic material from US and Canada for use on the EU native dairy breeds. As a result, average national yields have increased. One of the major improvements is the ability of the cow to partition nutrients into milk preferentially to maintenance and/or growth. This has undoubtedly resulted in increased efficiency. The UK dairy herd has increased its average yield by 8.8% from 1995 to 1997 and the top 10% of herds are averaging 8351 litres per cow. There are additional benefits which include the following:

(i) A cow's lifetime production can be achieved in less lactations, therefore there are less maintenance costs e.g. lifetime production of 30,000 litres achieved as 5 lactations of 6,000 litres or 3 lactations of 10,000 litres

(ii) A 100 cow herd producing average yield of 6,000 litres = 600,000 litres/year or 60 cow herd producing 10,000 litres, therefore less cows to maintain.

(iii) Less replacement heifers to maintain.

This could reduce methane emissions by 20 to 30% through reduced numbers. The genetic merit of livestock within the EU is rapidly improving and this will undoubtedly bring with it increased efficiency. The management of these high genetic merit cows will also become more complex and the overall implementation of this may be stalled by animal welfare implications. High genetic merit cows can have increased problems with fertility, lameness, mastitis and metabolic disorders. All these issues will have to be addressed if genetic progress is to be successfully continued.

5.0 OPTIONS FOR REDUCING METHANE EMISSIONS FROM LIVESTOCK MANURES.

Livestock manures have the potential for anaerobic production of methane, which if fully achieved would exceed the amounts of methane emitted by the rumen by a factor of two (Johnson et al, 1992). Methane production can only occur under strictly anaerobic conditions and is more effective when temperatures are high (~39°C) and moisture levels are sufficient. These factors make the estimation of production levels difficult. Another factor influencing the rate of emission from manure is dietary quality and hence the composition of the manure organic matter which is available as a potential substrate for methane production.

5.1 Impact of reduction of livestock numbers

A reduction of livestock numbers in the EU will directly reduce the quantity of manure and hence methane emissions from this source. The overall impact would be dependent upon the types of livestock whose population diminished, as manure output is directly related to dry matter intake and body size.

5.2 Impact of increased rumen fermentation efficiency and/or animal productivity

All methods which improve rumen fermentation efficiency are likely to increase the digestibility of the diet which in turn means that there is less digestible organic matter in the faeces for methanogenic activity. For example, increasing the digestibility of a ration by 5% units would decrease the volatile substrate by just over 10% according to IPCC methodology. It is also likely that the B_O (methane producing potential) may increase as differences in emission rates between grain and hay fed animals were noted by Lodman et al, (1993) over 7.5 times more methane per kg manure was emitted when animals were fed on grain. Jarvis et al, (1995) showed there was a strong relationship between carbon to nitrogen ratios in the manure and the total amounts of methane emitted (in manure from grazing animals). Methane emissions increased with lower C to N ratios in an exponential fashion. Therefore manure with a high nitrogen content will emit greater levels of methane. Increased rumen fermentation efficiency (hexose partitioning, defaunation, ionophores) will all minimise nitrogen emissions and hence reduce the potential of the manure to emit methane.

Increased animal productivity will reduce animal numbers and reduces the maintenance overhead associated with each unit of product, hence emissions from manure will occur as a result of lower manure output due to number of animals and per unit product.

Research into the composition of livestock manure and its potential methane producing capacity would be beneficial in making improved estimates of methane emissions and to estimate the impact of dietary/efficiency changes on this.

6.0 CONCLUSIONS

This review has considered the impact of various options on reducing methane emissions from enteric fermentation and livestock manures. The possible options considered were reduction of livestock numbers, increased rumen fermentation efficiency and increased animal productivity. There is a general trend in the EU for livestock numbers to reduce, but the degree of certainty of the levels of these reductions by animal species and type and hence the resultant reduction in methane emission, is low. Livestock numbers are indirectly affected by the application of agricultural policy measures other than those directed at livestock, which beyond the year 2000 are yet to be ratified. Reduction of methane emissions by reduced animal numbers in the EU is dependent on no more ruminants being reared elsewhere to compensate for the possible reduced output of milk/meat in the EU. This method of reducing methane emissions is dynamic and dependent on many outside factors, therefore requiring ongoing monitoring.

The second option considered was increased rumen fermentation efficiency. There are numerous possibilities associated with this and most of which require substantial amounts of research and development. The most promising approach is one involving immunisation against strains of both rumen protozoa and rumen methanogens, offering up to 70% reduction in methane emissions. This is at an early stage of development and the longevity of the immunisation is required to be established. If successful this approach could be applied to all ruminants in all member states. The use of propionate precursors and direct fed microbials offer alternative approaches which may allow up to 25% reductions in methane emissions and would be available to all livestock receiving daily supplements. Of the direct fed microbials, acetogens appear to be a more promising option than methane oxidisers.

The third option considered was to increase animal productivity. Many of the options for achieving this require chemical/hormonal additives which are either currently in use throughout the EU or as in the case of hormones are banned within the EU. Other options include chemically upgrading or strategically supplementing forages to improve their utilisation and hence the productivity of the animal. This approach is more suitable to developing countries, but may be of significance if other countries join the EU. Improving the genetic merit of livestock is a very effective way of increasing animal productivity. All of these methods can only be effective in reducing methane emissions if the number of livestock are reduced along side the increased productivity.

All the options considered above to reduce methane emissions from livestock will tend to indirectly reduce methane emissions from livestock manure. Less animals mean less manure, increased rumen efficiency and animal productivity tend to lead to increased digestibility and hence reduced quantities of manure. Some options will possibly change the composition of the manure, particularly nitrogen levels and hence reduce emissions of ammonia.

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8.0 GLOSSARY AND ABBREVIATIONS

Enteric fermentation - Anaerobic microbiological fermentation of polysaccharides and other feed components in the gut (rumen and/or hindgut) of animals

IPCC - Intergovernmental panel for climate change

Feedlot - Intensively reared livestock in yards, fed high concentrate diets

Forage - Fibrous plant material, usually involving the whole plant

Defaunation - Removal of the protozoa population from the rumen

Methanogens - Methane producing archaebacteria

Concentrates - Feed matter which is mainly derived from the seeds of plants or by-products of the seeds after processing

Acidosis - Metabolic or digestive disorder in ruminants where acids accumulate in the rumen, lowering the pH to a level where the bacterial population becomes inactive.

Laminitis - Inflamation of the lamina in the foot

Hexose partitioning - Utilisation of hexose in the rumen by the micro-organisms can be by direct accumulation or by fermentation and the balance between these two can alter affecting the efficiency of the rumen

Probiotics - Microbiological preparations which are beneficial to the gut micro-flora

Acetogens - Micro-organisms which utilise carbon dioxide and hydrogen to produce the short chain fatty acid acetate

Gram-positive bacteria - Bacteria whose cell walls accept Gram stain

Hexose - e.g. glucose $C_6H_{12}O_6$

Pentose - e.g. xylose $C_5H_{10}O_5$

VFA - Volatile fatty acids

mM - milli-molar or millimoles per litre

Appendix 1

Costing Methodology

A1. Costing Methodology

A1.1 INTRODUCTION

There are two general modelling approaches to estimating the cost of pollution abatement strategies: top–down models and bottom–up models. The former are macroeconomic models that attempt to capture the overall economic impact of an abatement strategy; they tend to analyse aggregate behaviour based on economic indices and elasticities. With top–down models, the overall cost of a pollution abatement strategy is normally expressed in terms of a change in gross domestic product (GDP).

In contrast, bottom–up models look at the microeconomic costs of individual mitigation options. Bottom–up models rely on the detailed analysis of technical potential; focusing on the integration of engineering costs and environmental performance data. With bottom-up models, the overall cost of a pollution abatement strategy includes, for example, investment, operation and maintenance, and energy costs.

Past experience has shown that bottom–up models have problems accounting for consumer behaviour and administration costs; whereas top–down models have problems accounting for different rates of technical change. Essentially, top–down models are better at predicting wider economic effects and bottom–up models are better at simulating detailed technological substitution potential.

The current study which AEA Technology is conducting for DGXI aims to examine the technical potential of abatement options at a detailed level, and therefore uses a bottom–up costing model; specifically, the costing model used in this study is the levelised cost/discounting method.

A1.2 THE LEVELISED COST/DISCOUNTING METHOD

The levelised cost/discounting method essentially uses discounted cash flow techniques to reduce the stream of non-recurring and recurring costs associated with each option to a single present value in a given base year. Furthermore, to facilitate comparison between options with different operating lives, the present value of the total cost stream of each option is annualised over the forecast period of plant operating life. As with normal economic project appraisal, this involves determining the equivalent annual cost of each option. An indicator of the cost-effectiveness of each option is then derived by normalising the equivalent annual cost to the resulting emission reduction, e.g. the tonnes of methane abated. As this method uses discounted cash flow techniques, costs and accomplishments are temporally differentiated according to when the pollutants are actually abated.

A1.2.1 The Methodology

The proposed costing methodology comprises the following four steps:

1. collect estimates of the relevant model parameters for each option;
2. determine the present value of each option's total cost stream;
3. annualise the present value of the total cost stream of each option over its operating life;
4. normalise the annualised cost of each option to the resulting emission reduction.

Step 1

The first step is to make estimates of the following model parameters for each mitigation option.

- The non-recurring incremental costs, i.e. the one-off costs incurred to install/implement the mitigation option *k* in period *t*, and the time required to install/implement each option. Where possible, non – recurring costs are disaggregated into the following two categories:

  - purchased equipment costs and.
  - direct and indirect installation costs (i.e. associated costs).

- The annual recurring incremental costs, are disaggregated into the following two categories:

  - the annual incremental costs required to operate mitigation option *k* in period *t*; and, where applicable,
  - the annual savings in recurring costs from operating mitigation option *k* in period *t*.

  Where possible, these two sub–categories of recurring costs will be divided into energy, maintenance and labour costs.

- The operating life of each option, in years (denoted by *t*).

- The appropriate real discount rate, *r*. In agreement with DGXI, a discount rate of 8% was used, to ensure consistency with work carried out on costs of reducing CO_2 emissions.

- The annual quantity of pollutant abated by mitigation option *k* in period *t*.

Examples of the types of costs that might be included within each of the above cost categories is given in Table A1.1.

Where possible, uncertainty in the raw data is addressed through the use of cost and/or performance ranges.

Unless originally quoted in ECU, all of the above costs are converted to ECU using appropriate yearly exchange rate obtained from EUROSTAT (Table A1.2). Costs are converted to 1995 ECUs using the Industrial Producer Price Index for total industry in the EU15 (Table A1.3).

The costs outlined above essentially represent the engineering and financial costs of specific mitigation options. Not included in the cost approach proposed are the macroeconomic and/or social costs of mitigation options. As a result, the impact that an individual mitigation option will have on the level of GDP and human welfare is not addressed in this study.

Table A1.1 Checklist of Capital and Yearly Operating Costs

NON-RECURRING CAPITAL COSTS	
Total Direct Non-recurring Costs	
Purchased Equipment Costs	primary control device auxiliary equipment instrumentation VAT/sales taxes on equipment freight modifications to other equipment
Buildings	
Direct Installation Costs	foundations and supports handling and erection electrical piping insulation painting
Site Preparation	
Total Indirect Non-recurring Costs	
Indirect Installation Costs	engineering construction and field expenses contractor fees start-up performance testing contingencies
Other Non-recurring Costs	
Land – green-field site Working Capital Off-site Facilities	
ANNUAL RECURRING COSTS	
Variable (Direct) Costs	
Non-fuel Operating Costs	maintenance materials operating, supervisory and maintenance labour staff training replacement parts water treatment waste treatment and disposal
Fuel Operating Costs	electricity, oil, gas etc.
Fixed Operating Costs	
Overheads Rates Insurance Administrative Charges	
Positive Side Effects	
Savings	energy use reductions in quantities of chemicals or solvents
Improved Product Quality Useful By-products	

Table A1.2 ECU Exchange Rates: 1 ECU =

Country	1990[1]	1991[1]	1992[1]	1993[1]	1994[1]	1995[1]	1996[2]	1997[3]
USA	1.27	1.24	1.30	1.17	1.19	1.31	1.28	1.25
Germany	2.05	2.05	2.02	1.94	1.92	1.87	1.91	1.95
Netherlands	2.31	2.31	2.27	2.18	2.16	2.10	2.14	2.18
UK	0.71	0.70	0.74	0.78	0.78	0.83	0.82	0.74

[1] Source: EUROSTAT (1997) 96 Yearbook.
[2] Based on monthly average for the year.
[3] Based on average for January only.

Table A1.3 Industrial Producer Price Index: Total Industry (1990 = 100)

Year	1992	1993	1994	1995	1996	1997
Index	104.4	106.1	108.2	112.4	113.3	114.2

Source: EUROSTAT (1997) Eurostatistics: data for short-term economic analysis.

Step 2
Using the relevant parameter values identified in step 1, the second step is to determine, for a suitable base year (denoted as t = 0), the present value of each option's total cost stream. This is given by equation 1.

$$\mathrm{PVC}_0^k = \sum_{t=0}^{\mathrm{T}^k} \left[\mathrm{NRC}_t^k + \mathrm{RC}_t^k\right] \circ \left[1 + r\right]^{-t} \qquad [1]$$

where

PVC_0^k = the present value of the total cost stream for mitigation option *k* in year zero,

NRC_t^k = the non-recurring cost of mitigation option *k* in period *t*,

RC_t^k = the recurring costs to operate mitigation option *k* in period *t*,

T^k = the operating life of mitigation option *k*, and

r = the appropriate discount rate.

Step 3
From the present value of the total cost stream, the third step is to calculate the equivalent annual cost of each mitigation option (i.e. the value of an equal annual payment throughout the option's life, which yields the same present value). This is also referred to as levelising/annualising the present value of the total cost stream, and is computed using equation 2.[1]

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[1] When the annual recurring costs are assumed to remain constant in real terms over the operating life of the measure, the equivalent annual cost can be determined by dividing the non-recurring cost by the appropriate annuity factor and adding this to the annual recurring costs. However, this approach is more difficult to apply when the build time exceeds one year. Hence, it is not used here.

$$\mathrm{EAC}^k = \left[\mathrm{PVC}_0^k\right] \circ \left[r^{-1}\left\{1-(1+r)^{-t}\right\}\right]^{-1} \qquad [2]$$

where

EAC^k = the equivalent annual cost of mitigation option *k*, and

$\left[r^{-1}\left\{1-(1+r)^{-t}\right\}\right]$ = the *t* - year annuity factor for a discount rate of *r*.

Alternatively, the equivalent annual cost of environmental protection measure *k* can be found by multiplying the present value of costs by the appropriate capital recovery factor, i.e.

$$\mathrm{EAC}^k = \left[\mathrm{PVC}^k\right] \circ \left[r(1+r)^t\right] \circ \left[(1+r)^t - 1\right]^{-1}$$

where

$\left[r(1+r)^t\right] \circ \left[(1+r)^t - 1\right]^{-1}$ = the *t* - year capital recovery factor for a discount rate of *r*.

Step 4

The fourth and final step is to normalise the annualised cost of each option to the resulting emission reduction. This will provide an indicator of the cost-effectiveness of the proposed option (it may loosely be interpreted as the marginal cost of the option), and is computed as follows

$$\mathrm{MC}^k = \frac{\mathrm{EAC}^k}{\mathrm{QA}_t^k}$$

where

QA_t^k = the quantity of pollutant abated by mitigation option *k* in period *t*.

That is,

$$\mathrm{MC}^k = \frac{\left\{\sum_{t=0}^{\mathrm{T}^k}\left[\mathrm{NRC}_t^k + \mathrm{RC}_t^k\right] \circ \left[1+r\right]^{-t}\right\} \circ \left\{r^{-1}\left[1-(1+r)^{-t}\right]\right\}^{-1}}{\mathrm{QA}_t^k} \qquad [3]$$

Subject to certain assumptions, other methods exist that will produce identical results. One of these involves discounting the quantity of pollutant abated over the operating life of an option back to a summary statistic, referred to as the present tonnes equivalent (PTE), and dividing the present value of the total cost stream by the PTE. Using this method, the marginal cost of abatement is computed using equation 4.

$$\mathrm{MC}^{k} = \frac{\left\{ \sum_{t=0}^{\mathrm{T}^{k}} \left[\mathrm{NRC}_{t}^{k} + \mathrm{RC}_{t}^{k} \right] \circ \left[1 + r \right]^{-t} \right\}}{\sum_{t=\mathrm{x}}^{\mathrm{T}^{k}} \left[\mathrm{QPA}_{t}^{k} \right] \circ \left[1 + r \right]^{-t}} \qquad [4]$$

where x is the year in which pollution abatement option k is implemented or becomes operational, and the denominator is the PTE.

Both approaches, i.e. equations 3 and 4, yield the same result. The second approach has the advantage over the first in that it allows one to account for variations in the quantity of pollutant abated over time. In this study, however, the quantity of pollutant abated by each measure is assumed to remain constant over time. Consequently, we have opted for the use of the first approach, i.e. equation 3. Moreover, the first approach is more intuitively appealing in that it avoids the need to discount annual emission reductions.

A1.3 VARIATIONS IN COSTS ACROSS EUROPE

To gain an insight into how the cost-effectiveness of the various measures might vary across the EU15, each component of the base data has been adjusted to take into account known differences in relative factor prices between the base country, for which detailed cost data exists, and other members of the EU15, for which cost data needs to be estimated. To do this an appropriate relative price index has been constructed for each significant cost component. Table A1.4 summarises those factors used to adjust the individual costs components and Table A1.5 lists the indices used.

Table A1.4 Cost Adjustment Factors

Cost Component	Adjustment Factor
Non-Recurring Costs:	
Purchased Equipment	Purchasing power parities
Associated Installation Costs	Construction and civil engineering costs
Recurring Costs:	
Energy Inputs/Savings	Electricity costs, Gas costs and Fuel costs
Materials Inputs/Savings	Purchasing power parities
Labour Inputs	Average hourly labour costs

Table A1.5 Price Indices (UK = 100)

Country	PPPs	Electricity Prices	Gas Prices	Fuel Costs	Construction Costs	Labour Costs
Austria	134	118	154	130	115	151
Belgium	127	99	112	96	115	162
Denmark	160	101	116	111	107	147
Finland	148	92	116	105	124	134
France	133	88	128	100	98	146
Germany	137	147	162	99	147	177
Greece	95	90	N/A	38	86	53
Ireland	103	96	252	137	88	98
Italy	98	135	138	116	101	138
Luxembourg	122	90	112	69	150	131
Netherlands	128	109	117	90	100	147
Portugal	84	178	N/A	89	49	42
Spain	102	118	126	85	53	115
Sweden	152	57	N/A	75	117	145
UK	100	100	100	100	100	100
United States	95	68	81	75	50	114

Source: OECD (1997) Main Economic Indicators; HMSO (1997) DUKES; EUROSTAT (1997) Yearbook 96; Spoons (1995) European Construction Costs Handbook.

The table should be read vertically. The columns show the number of monetary units needed in each country to purchase some quantity of the specified commodity relative to the cost of purchasing that quantity in the UK. For the individual mitigation options considered, the above indices are re-based to correspond to the currency in which the base cost data was originally sourced. For example, the cost of the some of the measures to abate emissions from gas compressors was based on Dutch data. Therefore, the above indices are re-based to the Netherlands (i.e. NL = 100) and the re-based indices are then used to estimate the costs of that measure in other members of the EU15. Where the base cost data has not been split between the various non-recurring and recurring cost components used in the analysis, a weighted average index has been used, assuming that the recurring and non-recurring costs are allocated equally to each component.

Appendix 2

CORINAIR Data

Table A2.1 Estimated Emissions of CH_4 in EU Member States in 1990 (kt)

	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
1 Public power, cogeneration and district heating	0	0	1	1	1	6	1	0	4	0	1	0	9	1	3	28
2 Commercial, institutional and residential combustion	11	4	6	1	150	112	0	4	17	0	2	7	44	10	1	369
3 Industrial combustion	1	5	1	1	7	10	0	0	10	0	2	3	7	4	0	51
4 Production processes	NE	14	0	AZ	6	6	1	0	8	0	8	2	4	0	0	49
5 Extraction and distribution of fossil fuels	119	47	12	191	310	1,547	364	10	347	2	1	2	684	0	1,211	4,846
6 Solvent use	0	0	0	NE	0	0	0	0	0	0	0	0	0	NE	0	0
7 Road transport	3	9	2	2	22	69	4	1	25	0	6	1	11	13	10	180
8 Other mobile sources and machinery	IE	0	1	2	1	2	1	0	8	0	0	0	2	3	0	20
9 Waste treatment and disposal	161	5	122	66	739	2,249	202	138	1,454	4	378	35	507	180	1,088	7,328
10 Agriculture	356	270	263	160	1,611	2,064	363	643	1,764	18	520	204	874	205	1,076	10,391
Total Anthropogenic Emissions	**651**	**355**	**407**	**424**	**2,847**	**6,066**	**936**	**796**	**3,637**	**24**	**918**	**254**	**2,142**	**416**	**3,389**	**23,261**
11 Nature	204	15	354	566	191	0	4,572	54	291	1	123	137	856	1,691	0	9,055
Total	**855**	**370**	**761**	**990**	**3,038**	**6,066**	**5,508**	**850**	**3,928**	**25**	**1,040**	**391**	**2,998**	**2,106**	**3,389**	**32,317**

Source: CORINAIR90

Table A2.2 Estimated Emissions of CH_4 in EU Member States in 1994 (kt)

	A	B	DK	FIN	FR	GER	GRE	IRE	IT	LUX	NLS	P	SP	SW	UK	EU15
1 Public power, cogeneration and district heating	0	0	1	2	2	9	1	0	6	0	1	1	9	1	12	45
2 Commercial, institutional and residential combustion	21	5	6	9	153	60	0	3	18	1	22	7	41	0	46	392
3 Industrial combustion	0	2	1	2	5	12	1	0	6	0	3	2	7	1	6	49
4 Production processes	0	2	1	4	6	9	0	0	10	0	7	2	4	0	0	44
5 Extraction and distribution of fossil fuels	5	39	14	0	342	1,161	49	11	393	2	187	0	653	0	811	3,667
6 Solvent use	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
7 Road transport	4	10	2	3	18	36	4	1	29	0	6	2	13	13	25	165
8 Other mobile sources and machinery	0	0	1	0	0	2	0	0	5	0	0	0	2	2	3	18
9 Waste treatment and disposal	81	90	78	122	690	1,900	115	136	1,766	3	288	35	651	40	1,828	7,824
10 Agriculture	463	272	327	95	1,626	1,660	280	657	1,898	17	563	204	928	216	1,116	10,321
Total Anthropogenic Emissions	**575**	**421**	**431**	**237**	**2,842**	**4,848**	**449**	**808**	**4,132**	**22**	**1,078**	**253**	**2,309**	**274**	**3,848**	**22,525**
11 Nature	57	13	354	0	95	0	34	37	72	1	112	130	854	1,691	0	3,451
Total	**632**	**434**	**785**	**237**	**2,937**	**4,848**	**483**	**845**	**4,204**	**23**	**1,190**	**383**	**3,164**	**1,964**	**3,848**	**25,976**

Source: CORINAIR 94

Appendix 3

Alternative Projections of Coal Production in the EU

A3. Alternative Projections of Coal Production in the EU

A3.1 INTRODUCTION

The emission projections used in Section 6 are taken from the European Commission's, DGXVII Energy in Europe 2020 study, however alternative projections of coal production are made on an annual basis by the IEA and a recent study by CSIRO of Australia for the IEA Greenhouse Gas Programme (Williams et al, 1996) has also addressed coal production and the associated level of methane emissions world-wide. In this Appendix, the production projections from the CSIRO study and the IEA are compared with those from the European work. Differences between the sets of projections are reviewed and discussed. Then based on the three sets of projections, together with other additional information concerning the European coal industry, an alternative projection for the production of coal in the EU has been derived.

A3.2 CURRENT EMISSIONS FROM COAL MINING

The CSIRO study for the IEA Greenhouse Gas Programme estimated methane emissions world-wide based on projections of coal production and the theory of methane emission to determine the amounts of gas that could be emitted at different stages of the coal extraction/use cycle. The calculated emissions were then compared with collected data where available. Having 'proved' the technique for those countries where reasonable data was available it was possible to 'extrapolate' it for other countries, and so build up a global figure.

The study resulted in an average specific methane emission figure for underground mines in each coal producing country; together with a similar, but lower figure for surface mines.

This methodology used has certain limitations, but it is an improvement on previous work, which has generally multiplied production by an empirically derived emission rate on a country by country basis. Readers with an in-depth knowledge of a specific country could no doubt make detailed criticisms of some items of the data, but this is unlikely to significantly affect the global total.

For 1993 (the most recent production data then available) CSIRO estimated the total saleable black coal production as 3384.8 Mt and the corresponding methane release as 31,949 Mm^3. The three largest producers were China, USA and CIS. Data for these, plus the four EU coal producing countries are given in Table A3.1.

These results show that in 1993, the EU produced 4.7% of world hard coal production, but emitted 8.1% of the coal-related methane. The relatively high methane emissions can be explained in two ways. EU emissions are known to be comparatively high, the mines are deeper than average and the in situ methane concentration tends to increase with depth. Also EU reporting tends to be better than average and better reporting often leads to higher reported emissions. All these emissions are 'gross', that is they include any mine methane which is currently beneficially utilised, but this is less than 5% on a global basis.

Table A3.1 1993 Coal Production and Related Methane Release for Selected Countries

Country	Coal production (Mt)	Methane release (Mm^3)
China	1 047.0	11 238
USA	774.2	6 387
CIS	420.4	6 028
France	9.0	176
Germany	64.2	1 553
Spain	18.2	83
UK	67.5	790
Sub-total EU	*158.9*	*2 602*

One important point that emerges in the table is the importance, both in coal production and methane emission, of China and CIS. These are two countries where the available data is limited or suspect and hence there are large uncertainties associated with these values.

In addition to hard coal production there was 986.7 Mt of lignite/brown coal produced in 1993. This is generally in thick, near-surface, seams and the normal assumption is that this does not lead to any emission of methane. This is, doubtless, an over simplification, but is reasonable given the other approximations made.

A3.3 FUTURE PROJECTIONS

A3.3.1 Sources of Data

As discussed above three main sets of projections have been considered, these are:

1. The CSIRO report which presents projected coal production and coal-related methane emissions for the ten largest coal producing countries and globally for the years 2000, 2005 and 2010. Hence, figures are given for Germany and the UK, but not for France or Spain. The global figures are given in Table A3.2. Detailed data for the UK and Germany are discussed below.

Table A3.2 Global Projected Coal Production and Related Methane Emissions

Year	Coal production (Mt)	Methane release (Mm^3)
2000	3 968.8	37 462
2005	4 446.7	41 973
2010	4 982.2	47 027

2. The “IEA Coal Information” series of books (‘IEA-CI’). The most recent of these, the 1996 edition, was published in July 1997. For most countries this gives actual production up to 1995 and estimated production for 1996. The series of publications gives historical coal production, subdivided into hard coal and brown coal/lignite for each individual OECD country. It also gives projections, again subdivided into hard coal and brown coal/lignite, but in terms of ‘tonnes of coal equivalent’ (tce) for 2000 and 2005. Using the historical data it is possible to convert these to ‘real’ tonnes.

3. Energy in Europe - European Energy to 2020 (‘EE2010’). This gives 1990 (historical) and 2000, 2010 and 2020 (projected) solid fuel production by country for a range of scenarios.

The ‘conventional wisdom’ scenario has been used in the present evaluation. Figures are in ‘tonnes of oil equivalent’ and are not subdivided into coal types. These data are converted to tce and compared with the “IEA Coal Information” data in the following section.

A3.3.2 Projected coal production in the EU

Within the EU, France, Germany, Spain and the UK are the only countries currently mining hard coal in commercial quantities. The Netherlands ceased hard coal mining in the 1970s, Belgium in 1992 and Italy in 1993. There were some very small operations in Ireland, typically less than 10,000t/y in total, until recently. There is no realistic prospect of any of these other countries resuming conventional coal mining on a commercial scale, given the current, and likely future, prices of imported coal.

The four EU coal producers are examined in the following sections and a set of alternative projections are derived based on the previous projections and additional country specific information.

France

A comparison of historical and projected coal production data for France, using the three sources, quoted above is given in Table A3.3.

Table A3.3 Projected Coal Production for France

	1993	1995	2000	2005	2010	2020
EE2020						
Solid fuel (Mtce)		n/a	6.10	n/a	1.36	1.43
IEA-CI						
Total coal (Mtce)	9.32	8.64	7.00	5.43	n/a	n/a
Hard coal (Mtce)	8.29	7.81	7.00	5.43	n/a	n/a
Brown coal/lignite (Mtce)	1.02	0.82	0	0	n/a	n/a
Hard coal (Mt)	8.99	8.50	n/a	n/a	n/a	n/a
CSIRO						
Hard coal (Mt)	9.0	n/a	n/a	n/a	n/a	n/a

The ‘EE2020’ projections are lower than those prepared by the IEA ‘IEA-CI’. However, both these sources may be high; a ‘national coal pact’ has been signed between Charbonages de France and the unions for the termination of all mining by the year 2005. On this basis it would be reasonable to accept the ‘EE2020’ figure of 6.10 Mtce (6.64 Mt) and zero for 2005 and beyond. CSIRO assumed all French production in 1993 was from underground mines. This is not strictly correct, but the projected tonnages are so small that it is a reasonable simplifying assumption. no forward projections were made.

Germany

The equivalent figures for Germany are given in Table A3.4.

Table A3.4 Projected Coal Production for Germany

	1993	1995	2000	2005	2010	2020
EE2020						
Solid fuel (Mtce)	n/a	n/a	71.00	n/a	60.45	48.54
IEA-CI						
Total coal (Mtce)	125.40	112.64	113.81	108.83	n/a	n/a
Hard coal (Mtce)	59.18	54.38	49.53	49.97	n/a	n/a
Brown coal/lignite (Mtce)	66.16	58.13	64.29	58.86	n/a	n/a
Hard coal (Mt)	64.18	58.86	n/a	n/a	n/a	n/a
CSIRO						
Hard coal (Mt)	64.2	n/a	38.2	26.4	18.2	n/a

Germany is currently the largest hard coal producer in the EU, but mining costs are very high and the industry is heavily subsidised. Continuation of the industry is on political and social grounds. It is therefore very difficult to make future projections. The CSIRO figures are lower than the other sources. Their report states that its figures were "provided from local sources or on information from IEA" (referenced to IEA-CI report). On balance it is proposed to accept these CSIRO figures as being likely to be the most recent and realistic ones and to extrapolate these for 2020. This has been done by assuming the same annual percentage decline projected by CSIRO for the period 1995 to 2010.

Spain

The equivalent figures for Spain are given in Table A3.5.

Table A3.5 Projected Coal Productions for Spain

	1993	1995	2000	2005	2010	2020
EE2020						
Solid fuel (Mtce)	n/a	n/a	13.26	n/a	8.60	4.87
IEA-CI						
Total coal (Mtce)	15.54	14.61	13.29	n/a	n/a	n/a
Hard coal (Mtce)	10.29	9.95	9.29	n/a	n/a	n/a
Brown coal/lignite (Mtce)	5.35	4.66	4.00	n/a	n/a	n/a
Hard coal (Mt)	14.13	13.60	n/a	n/a	n/a	n/a
CSIRO						
Hard coal (Mt)	14.1	n/a	n/a	n/a	n/a	n/a

Spain also has high mining costs. The geology is difficult and some of the seams being mined are nearly vertical. As CSIRO does not give a projection it is proposed to accept the 'EE2020' ones, interpolating for 2005. The actual tonnages are calculated by assuming the same ratios for equivalent to actual tonnages as for 1995, as done previously for Germany. In addition the ratio between underground and surface mining is assumed to remain the same. This is a reasonable assumption in view of the comparatively small tonnages involved.

United Kingdom

The equivalent figures for the United Kingdom are given in Table A3.6.

Table A3.6 Projected Coal Productions for the United Kingdom

	1993	1995	2000	2005	2010	2020
EE2020						
Solid fuel (Mtce)	n/a	n/a	29.33	n/a	18.77	12.50
IEA-CI						
Total coal (Mtce)	56.41	43.08	29.57	25.71	n/a	n/a
Hard coal (Mtce)	56.41	43.08	29.57	25.71	n/a	n/a
Brown coal/lignite (Mtce)	0	0	0	0	n/a	n/a
Hard coal (Mt)	68.20	51.47	n/a	n/a	n/a	n/a
CSIRO						
Hard coal (Mt)	67.5	n/a	30.0	16.8	9.4	n/a

The UK is in the enviable position of being the only EU coal industry which can operate without a government subsidy, although margins are very tight. The industry was privatised in 1995 and the industry has the benefit of contracts with the electricity companies which were negotiated prior to privatisation. These will end in 1998, when there is likely to be some hard negotiating. Indications are that UK coal can compete with imports, particularly when port unloading and rail transport costs are taken into account. Over recent years there has been a trend towards gas powered generation in the UK; however, the UK Government is currently undertaking a review of energy sources for power generation which is considering issues such as fuel security and diversity.

The most likely scenario is an ongoing, but reduced, market for coal up to 2010, some of which would be met by imports. Although, some forecasters think coal demand could actually increase after 2010, as North Sea gas becomes depleted.

The UK coal position is complicated, as far as methane emissions are concerned, by the fact that a significant part of the production is from surface mines. There are increasing environmental objections to new surface mines, so production from this source is likely to fall in absolute terms; although the industry would like to maintain this form of output for cost and coal quality reasons.

It is proposed to accept the 'EE2020' figures for 2010 and 2020, and interpolate for 2005, assuming the current ratio between equivalent and actual tonnes and the underground/surface mine split remains at its present ratio(approximately 2:1). The reason for preferring the 'EE2020' figures to those from the CSIRO study, although the latter were accepted for Germany, is that UK mining costs are still decreasing. On this basis the CSIRO figures seem rather pessimistic.

Alternative Projections for EU-15

The projections for coal production in France, Spain, Germany and the UK derived above are summarised in Table A3.7. Figures for 1990 have also been included, since this is often used as a 'base case' for various emissions.

Table A3.7 Projected Coal Production for EU-15 Coal Producers (Mt/y)

Country	1990	1995	2000	2005	2010	2020
France	11.2	8.5	6.1	0	0	0
Germany	77.0	58.9	38.2	26.4	18.2	8.1
Spain	14.9	14.2	12.9	10.6	5.9	4.7
UK	91.8	51.5	35.0	28.7	22.4	14.9
Total EU	*194.9	133.1	92.2	65.7	46.5	27.7

* Total for the four EU countries listed above. Other countries, within the EU-15, which have since ceased coal production, produced an additional 1.4 Mt in 1990. This difference can be neglected for all practical purposes.

Experience over the last 20 years has shown that EU coal production projections tend to overestimate future production. However, there are some who predict that there is unlikely to be any coal mining in the EU-15 countries in 2020.

Appendix 4

Transport Emissions Model

A4. Transport Emissions Model

A4.1 ROAD VEHICLES

For cars, new vehicle registrations and diesel car registrations up to 1995 were obtained from SMMT World Automotive Statistics. Extra figures for 1996 were obtained from National Automotive Trade Organisations where possible. In addition, figures for total national car parcs (including diesel car numbers) were obtained and used to check predictions where possible.

A4.1.1 Catalytic Converter Penetration

To model the increase of catalytic converters in the petrol car fleet, it is assumed all petrol cars built from 1993 include a three-way catalytic converter, as required under EU Directive. Years preceding this date have various penetration levels for converters, according to historic data for the particular Member State.

A4.1.2 Diesel Engine Penetration

Projected diesel penetration (% of registrations) was calculated using previous penetration levels and the minimum and maximum levels found in mature markets (UK, 20% and France, 45% respectively). Most Member State markets are predicted to reach 20% diesel fleet penetration by 2005. (Source: Automotive Environment Analyst, No. 25, Feb. 1997)

Country	Projected Diesel Penetration
Austria	43% current level maintained to 2020
Belgium	47% current level maintained at 45% to 2020
Denmark	Linear increase to 20% in 2005
Denmark	Linear increase to 20% in 2005
France	47% current level maintained to 2020
Germany	15% in 1995, 20% by 2000 and beyond.
Greece	No data on diesel registrations - Linear increase to 20% in 2005
Ireland	Linear increase to 20% in 2005
Italy	Increase to 20% in 2000
Luxembourg	Current 30% level maintained
Netherlands	Linear increase to 20% - 2005
Portugal	Linear increase to 20% - 2005
Spain	Current 33% level maintained
Sweden	Linear increase to 20% 2005
UK	Current 20% level maintained

A4.1.3 New Car Registration Predictions

Future new car registrations are predicted on the basis of a GDP scaling factor (from a 1995 base) derived for each Member State from figures used for the 'Energy in Europe' study. These scaling factors are shown in Table 1 below. The justification for the use of this scaling factor is that the number of new car purchases is linked to individual wealth, which in turn is related, albeit indirectly, to a country's GDP performance. It may also be expected that the rate of renewal of private cars will increase with individual wealth, thereby affecting the overall vehicle turnover and stock. However, the calculation of this effect is beyond the scope of this exercise.

Table 1 GDP scaling factors used to predict new car registrations

Year	1995	2000	2005	2010	2015	2020
Austria	1.00	1.12	1.24	1.38	1.51	1.63
Belgium	1.00	1.13	1.26	1.41	1.54	1.68
Denmark	1.00	1.15	1.28	1.42	1.54	1.65
Finland	1.00	1.17	1.32	1.48	1.62	1.75
France	1.00	1.13	1.26	1.41	1.54	1.67
Germany	1.00	1.13	1.28	1.43	1.57	1.70
Greece	1.00	1.15	1.36	1.60	1.86	2.15
Ireland	1.00	1.31	1.58	1.81	2.01	2.22
Italy	1.00	1.11	1.24	1.37	1.50	1.62
Luxembourg*	1.00	1.13	1.28	1.43	1.58	1.72
Netherlands	1.00	1.15	1.31	1.49	1.65	1.81
Portugal	1.00	1.17	1.40	1.67	1.95	2.24
Spain	1.00	1.16	1.34	1.54	1.74	1.94
Sweden	1.00	1.13	1.26	1.40	1.52	1.63
UK	1.00	1.15	1.31	1.48	1.63	1.78

*No data available: an average of Netherlands, Belgium, France and Germany used (Neighbouring countries geographically and economically)

A4.1.3.1 Survival Rates

Survival rates for new cars were calculated for the UK as a reference case using detailed data for this country. Survival rates for the other EU15 countries were developed by adapting these rates to fit predicted stock numbers with actual figures for the years 1992 - 1995.

The spreadsheet model developed to calculate vehicle numbers uses all the above information to predict vehicle numbers for:

- Petrol non-catalytic converter
- Petrol with catalytic converter
- Diesel
- Total cars for each member state.

These are summarised for EU15 in Figure 1 below.

A4.1.4 Vehicle Use Demand

The use of private cars for future years was predicted by extending previous years trends for the EU15 countries. Data for the years 1990 to 1994 (source: UN Annual Transport Statistics) on vehicle kilometres driven were extended to 2020 using a least-squares trend. For several countries, insufficient data were available and an overall growth rate, based on the average of the countries for which data were available was used to predict demand. This was the case for Austria, Belgium, Greece, Luxembourg and Portugal. For Italy and Spain, for which data are also sparse, a growth factor based on the other three 'major' EU countries, Germany, France and the UK, was used. The vehicle km demand projections are shown in Figure 2.

Figure 1 Predicted vehicle stock, by type for EU15 to 2020

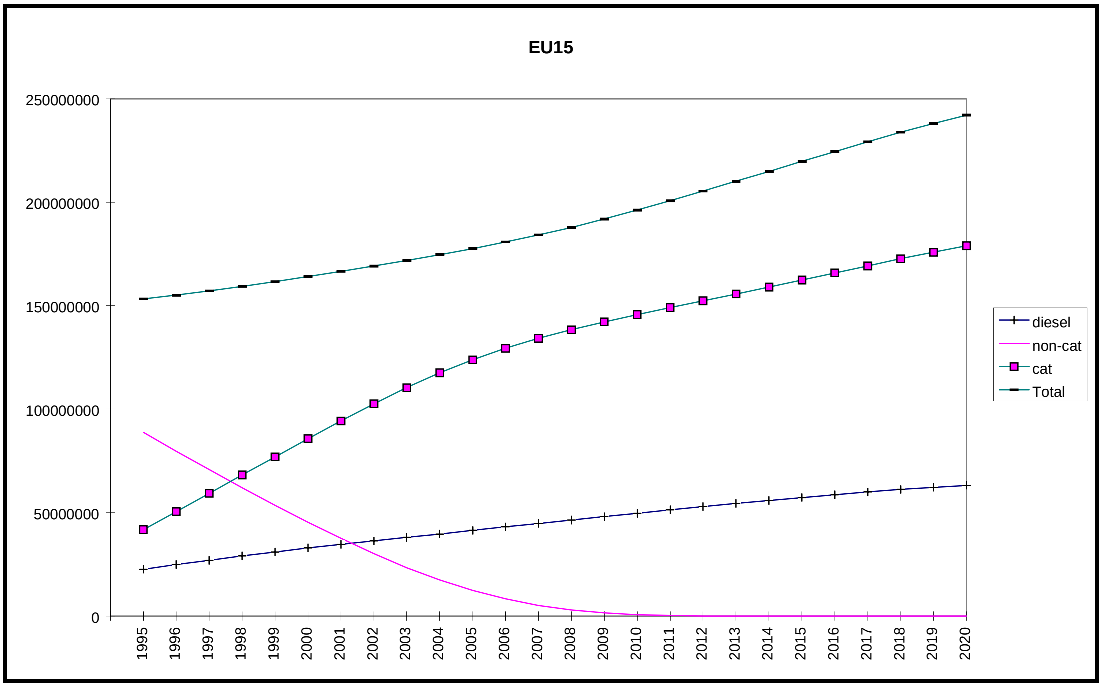
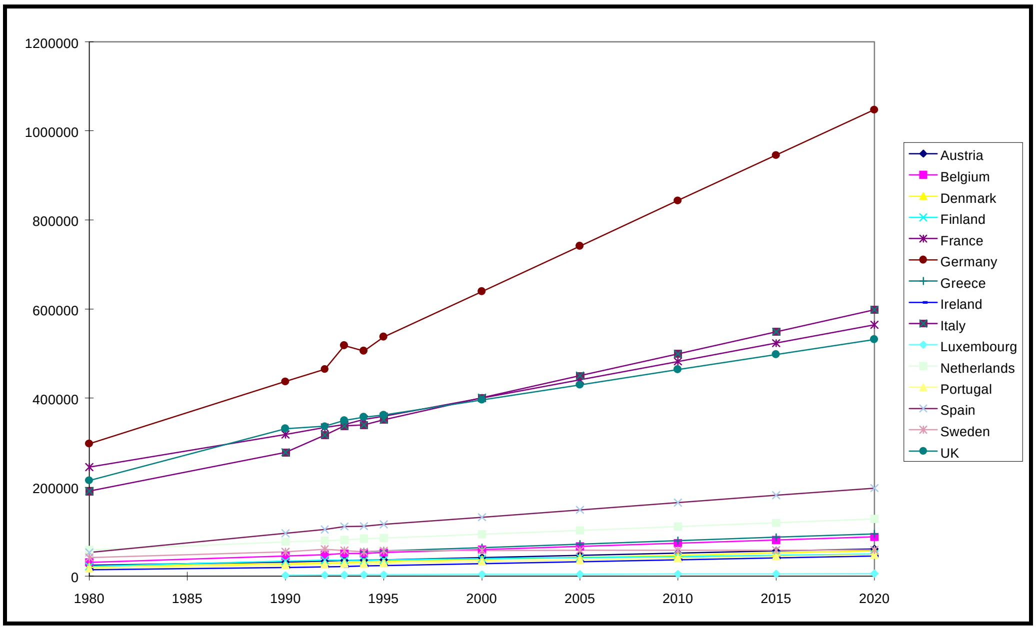


Figure 2 Vehicle km projections for EU15 to 2020



The results of the above stock modelling activities were used to make forecasts of a stock-weighted emission and vehicle km scaling factor to 2020 that was applied to the car segment of 1994 CORINAIR methane emissions. Emissions from non-car road vehicles, for which detailed information was not available, were forecast using the CORINAIR 1994 emissions and a forecast GDP inflator for each Member State. This inflator is thought to adequately represent growth in commercial road vehicle emissions. Other sectors make only negligible contributions to the non-car sector.

A4.2 NON-ROAD MOBILE SOURCES (OTHER TRANSPORT)

In accordance with the CORINAIR nomenclature for emission sources, these are broken down into:

- Off-road vehicles and machines:
    - Agriculture
    - Forestry
    - Industry
    - Military
    - Household
- Railways
- Inland Waterways
- Maritime Activities
- Airports

A4.2.1 Off-Road Vehicles and Machines

The proposed EU directive on Emissions from Non-road mobile sources has, and the latest 'Common Position' adopted by the EU member states will have a negligible effect on methane emissions from these sources[1]. This enabled a simplified methodology to be applied to forecast future emissions.

For future years, forecast is made for all the sectors activities using a single energy use factor (from Energy in Europe studies) that will simply scale vehicle population and therefore emissions. Hours of use, machinery power ratings and load factors are assumed to remain unchanged to 2020.

For the agricultural sector, a specific agricultural growth index was not available. In addition, the stock of non-road machinery is used in a wide range of specific agricultural applications, which may be subject to growth or decline over the period of the forecast. Information on the use of machinery within these agricultural sectors is not available. Therefore, a single energy use factor (again from Energy in Europe studies) was used for scaling future emissions.

As disaggregated data on stock at the Member State level are patchy and incomplete, an overall forecast for EU15 has been completed for these sources.

A4.2.2 Railways

Methane emissions from railway activities are a small proportion of total non-road emissions and are expected to rise in proportion to energy use, assuming there is no increase in electrification (which is already the case across most of EU15). Emissions in 1994 (from CORINAIR data) have been scaled using the Energy in Europe energy use factor for each country. Where 1994 CORINAIR data were incomplete, an estimate of 1994 emissions has been made using data on national totals of diesel locomotive power.

[1] Samaras and Zierock, Estimation of emissions from Other mobile sources and machinery.

A4.2.3 Inland waterways

Inland waterway vessels are a significant source of emissions within the non-road sector. A proposed EU directive on boat emissions will probably be aimed at recreational vessels under 24m in length, and then only for new craft. Existing diesel-powered freight vessels, which make the largest contribution to emissions will not be affected. In addition, CH_4 emissions are likely to remain unaffected. From 2010, new propulsion systems may be employed on inland vessels which demonstrate improved fuel efficiency of 10-30%. However, the slow stock turnover of this sector would indicate that such vessels will have a negligible effect on methane emissions before 2020.

Emissions in 1994 (from CORINAIR data) have been scaled using the Energy in Europe energy use factor for each country. Where CORINAIR data were incomplete, an estimate of 1994 emissions has been made using data on national totals of tonne-km total carried on inland waterways (in 1993).

A4.2.4 Aircraft

In the air sector, a projection based on commercial fleet energy use to 2010 has been extended to 2020 and applied to 1994 CORINAIR figures for LTO cycles and ground activities for each Member State. It is assumed that this forecast, despite being for overall energy use can be applied to these LTO cycles. Methane emissions increase in proportion to fuel use, and NO_x reduction measures are assumed to have negligible effects on CH_4 emissions.

A4.2.5 Marine

To simplify forecast of this sector, a simple scaling has been applied using the Energy in Europe energy use factors for each Member State. CORINAIR data (1994) for methane emissions is patchy and incomplete. Estimates for several member states were made on the basis of NMVOC data (which were more complete) and also Total goods thoughput of national seaports.

Various new technologies are envisaged in the marine sector (including all-electric ships an d fuel-cell powered vessels). However, the long-term development still required and the slow stock turnover means that negligible effects on emissions are expected.

Appendix 5

Comparison of CH_4 Reduction Studies

Paper by Dominique Gusbin, Coherence, Louvain la Neuve, Belgium, November 1998

Contribution of non-CO_2 greenhouse gases to the EU Kyoto target: Evaluation of the reduction potential and costs

1. Background

At the conference of the parties in Kyoto in December 1997, the EU agreed to reduce emissions of the six greenhouse gases (carbon dioxide, methane, nitrous oxide, hydrofluorocarbons, perfluorocarbons and sulphur hexafluoride) by 8% of 1990 levels by 2010[1]. The target applies to emissions weighted by their (100 year) global warming potential.

Until recently, studies and strategies for addressing climate change mitigation have principally been focused on reducing emissions of carbon dioxide, but the importance of other greenhouse gases and opportunities for their abatement have been increasingly recognised in the last couple of years. In particular, DGXI of the European Commission has lauched three studies considering non-CO_2 greenhouse gases and examining their reduction potential and costs. These studies are:

(1) Economic evaluation of quantitative objectives for climate change, COHERENCE (ongoing); in the framework of this study ECOFYS produced a report in June 1998 on Emission reduction potential and costs for methane and nitrous oxide in the EU-15;
(2) Reductions of the emissions of HFC's, PFC's and SF6 in the European Union, ECOFYS, June 1998;
(3) Options to reduce methane emissions, AEA Technology Environment, September 1998, and
Options to reduce nitrous oxide emissions, AEA Technology Environment, September 1998.

[1] More precisely, Member States have the choice of a 1990 or 1995 baseline for HFCs, PFCs and SF6, and a limited allowance can be made for sinks in calculating the 8% reduction.

This paper summarises the major findings from the above studies as to the reduction potential and costs of non-CO_2 greenhouse gases emissions in 2010. It also addresses the uncertainties in emissions and costs estimates for some sources and mitigation options. The reduction potential of each gas is estimated in comparison with a business-as-usual scenario to 2010. It is provided both in ktonne of gas considered and in ktonne of CO_2-equivalent using the global warming potential of the gases (100 years). Costs of reduction options or of packages of reduction options are provided in ECU (1995) per tonne of CO2-equivalent abated.

2. Emissions in 1990/1995

Emissions of the six greenhouse gases in the EU are shown in Table 1. Emissions of the three main greenhouse gases in 1990 are those estimated by the Member States and reported in the EU Second Communication to the FCCC. Emissions of the three halogenated gases in 1995 (the reference year for these gases) are those reported and discussed by the Member States at Expert Group meetings [1].

GHG	Emissions in 1990 (Mt)	GWP (100 years)	Emissions in 1990/1995 (*) (Mt CO2-equiv)
CO2	3365	1	3365
CH4	23	21	489
N2O	1	310	315
HFC's		[1000-3000]	37
PFC's		6500; 9200	7
SF6		23900	14
Basket of six			4227
EU Kyoto target in 2010		(% of 1990)	-8%
		(kt CO2 equ.)	3889

(*) 1995 for halogenated gases.

Table 1: Anthropogenic emissions of CO_2, CH_4, N_2O, HFC's, PFC's and SF6 in the EU in 1990/1995

After allowing for the global warming potential of the gases (100 years), it is clear that non-CO_2 greenhouse gases are significant contributors to greenhouse gases emissions. Methane and nitrous oxide emissions in 1990 are equivalent to 24% of

CO_2 emissions. Emissions of halogenated gases in 1995 are equivalent to slightly less than 2% of CO_2 emissions in 1990.

The main sources of anthropogenic emissions of non-CO_2 greenhouse gases in 1990/1995 are shown in Table 2. It should be noted that the uncertainty in emissions estimates for some sources is significant [2]. This is particularly the case for landfill gas emissions, for N_2O emissions from agricultural soils and for halogenated gases emissions.

The achievement of the EU Kyoto target means that total EU emissions of the six gases in 2010 should not exceed 3889 Mt CO_2-equivalent.

3. Emissions in 2010 under a Business as Usual scenario

To allow the calculation of achievable emission reductions in 2010, business-as-usual projections are calculated for each gas and each of the main sectors.

Emissions under a business-as-usual scenario were estimated in the above studies using similar assumptions regarding background trends in activity indicators (e.g. fuel production and consumption, crop areas, livestock numbers, waste production, industrial production) and management practices (e.g. manure management, reduction in fertiliser use, lower leakage rates from new gas pipelines) [3]. These assumptions are described and discussed in the study reports.

However, business-as-usual projections have been made using sometimes different approaches as to the inclusion or not of the effect of existing policies and measures to reduce greenhouse gases emissions [4]. There is no single approach for dealing with this issue, so the general approach adopted in this paper is not to take into account specific policies and measures which the EU and Member States may have already put in place to reduce emissions. These reductions will be included in the reduction potential of the corresponding measure. Consequently, emission projections may differ from those reported in the above studies and by the Member States because the latter include the effect of some measures already implemented and/or planned [5] [6].

Non-CO_2 greenhouse gases emissions under a business-as-usual scenario are shown by sector in Table 2.

GHG	Sources	1990/1995 (*)	2010 (**)	2010 incr/decr (%)	2010 (in Mt CO2-equ.)
CH4	**Total**	**23309**	**21469**	**-8%**	**451**
	Agriculture	**9946**	**9248**	**-7%**	
	enteric fermentation	7054	6463	-8%	
	animal manures	2022	1928	-5%	
	other	870	857	-2%	
	Waste	**7991**	**8499**	**6%**	
	landfill	6641	7043	6%	
	waste water	1350	1456	8%	
	Energy	**5350**	**3697**	**-31%**	
	coal production	2936	822	-72%	
	oil and gas sectors	1561	2139	37%	
	fuel combustion	853	736	-14%	
	Other	**22**	**25**	**12%**	
N2O	**Total**	**1015**	**1093**	**8%**	**339**
	Agriculture	**417**	**377**	**-9%**	
	Waste	**11**	**11**	**0%**	
	Energy (combustion)	**175**	**268**	**53%**	
	Transport	41	146	256%	
	Other	134	122	-9%	
	Industrial processes	**357**	**381**	**7%**	
	Other	**55**	**55**	**0%**	
Halogenated gases		**58**	**82**	**41%**	**82**
	HFC's	**37**	**65**	**76%**	
	PFC's	**7**	**5**	**-29%**	
	SF6	**14**	**12**	**-14%**	
Basket of non-CO2 gases					**872**

(*) 1990 and in kt for CH4 and N2O; 1995 and in Mt CO2-equ. for HFC, PFC and SF6
(**) in kt for CH4 and N2O; in Mt CO2-equ. for HFC, PFC and SF6

Table 2: Non-CO_2 greenhouse gases in the EU under a business-as-usual scenario (kt/year)

By 2010, methane emissions would fall by 1.8 Mt to 8% below 1990 levels due predominantly to a dramatic decline in emissions from coal mining as coal production falls. There is also a significant reduction in livestock related emissions as cattle numbers are predicted to fall. These reductions are partially offset by increased emissions from landfills due to large volumes of waste produced and hence disposed to landfill and a small rise in emissions from the oil and gas sector due essentially to increased gas consumption levels.

By 2010, nitrous oxide emissions are projected to increase by 8% (78 kt) from 1990. This is mainly due to an increase in emissions (of 93 kt) from road transport, due to the increased penetration of catalytic converters. There is also an increase (of 24 kt) in emissions from production processes which is partially offset by a fall (of 40 kt) in agricultural emissions. The business-as-usual projections do not include reductions of around 240 kt (in 2010) resulting from abatement plans installed or due to be installed at the main adipic acid EU production plants.

Finally, emissions of the three halogenated gases are projected to increase by 41% in 2010 compared to 1990 levels due essentially to a significant increase in HFC's emissions (78%); PFC's and SF_6 emissions are projected to decrease by 29% and 14% respectively over the same period due to a decrease in aluminium production and in the use of high-voltage switches in the EU.

Table 3 summarises the emission projections under a business-as-usual scenario for the basket of six gases. Overall, emissions are projected to increase by 6% in 2010 compared to 1990 levels.

	1990 1995 for halog.	2010 BAU (*)	2010 % of 1990
CO2	3365	3617	7%
CH4	489	451	-8%
N2O	315	339	8%
HFC+PFC+SF6	58	82	41%
Basket of six	4227	4489	6%
Kyoto target		3889	
Reduction needed		600	

(*) CO2 emission projections are calculated on the basis of the pre-Kyoto scenario which assumes an 8% increase in (energy-related) CO2 emissions by 2010 compared to 1990. Other CO2 emissions are assumed to remain constant.

Table 3: Business-as-usual projections for the basket of six gases (in Mt CO_2 equivalent)

The above projections should be regarded with caution; uncertainty in emission estimates for some sources is significant, a new baseline energy scenario is under development, forecasts for the three industrial gases are hampered by inconsistencies, lack of data for some countries and several source categories. For these reasons, Table 3 only gives an order of magnitude of the challenge.

The emission reduction required compared to business-as-usual (as defined in Table 3) is estimated to be about 600 Mt CO_2 equivalent for the six gases. Because this overall challenge excludes the effect of current policies equivalent to around 140 Mt of CO_2 [7], it is comparable to that under the pre-Kyoto communication (COM(97)481 final) where around 500 Mt of additional emission reduction was estimated to be necessary to meet the EU Kyoto target.

4. Emission reduction measures, potential and costs in 2010

Measures have been identified to reduce non-CO_2 emissions in all of the main sectors, together with estimates of the achievable reductions in 2010 compared to the business-as-usual scenario and mitigation costs.

In general, similar mitigation options have been considered in the above studies (see list in Annex 1). The major difference concerns mitigation options for methane in the agricultural sector. On the basis of a recent study by the Agricultural University of Wageningen (Gerbens, 1998), further measures have been considered in the Ecofys study; these measures aim at improving feed conversion efficiency by adjusting animal diets. Another difference, which has however a lower impact on emission reductions, concerns mitigation options for N_2O in the energy sector. The AEAT study has estimated the impact on N_2O emissions of several CO_2 reduction measures such as the introduction of energy efficiency measures and renewables in the energy supply sector, and inter-modal shift in the transport sector.

Although different assumptions have sometimes been made as to the applicability of individual measures, overall both studies have estimated comparable achievable reductions for methane and nitrous oxide emissions provided the same mitigation options are considered.

The reduction costs of the different measures or packages of measures have been estimated on the basis of direct (private) resource costs (i.e. investment costs, operation and maintenance costs, and potential cost savings).

Large cost ranges for some measures and small differences in costing methodology and approach [8] have led to sometimes big differences in cost estimates between the two studies. Nevertheless, a number of measures have been identified in the same cost ranges. Three cost categories are thus considered in this analysis that correspond to different cost ranges, and mitigation options are ranked according to these cost categories. The "low cost" category corresponds to measures identified as having "negative costs"[2] (or as being "cost-positive" according to AEAT study). The "medium cost" category encompasses measures with a marginal reduction cost ranging from 0 to 50 ECU per tonne CO_2 equivalent abated. Finally, the "high cost" category characterises the measures which fall above the limit of 50 ECU per tonne CO_2 equivalent abated.

The potential reductions and costs of non-CO_2 mitigation measures are summarised below.

For methane, the set of options is estimated to offer total reductions of 9 Mt (or 189 Mt CO_2 equivalent) in 2010 and would bring methane emissions down to 53% of 1990 levels. In other words, methane emissions could be reduced by 47% in 2010 compared to 1990 levels. The largest reductions (Table 4.1) arise from measures directed at landfilling of waste (4.7 Mt), although significant reductions are also available in the agricultural sector (2.9 Mt) and in the oil and gas sector (1.3 Mt).

Total reductions include methane reductions from landfills as resulting from the implementation of Member States' current and planned measures (i.e. utilisation or flaring of landfill gas and diversion of organic waste from landfill). These measures are estimated to deliver reductions of 3.2 Mt by 2010. Their implementation is also suggested under the proposed Landfill Directive.

Reductions in enteric fermentation emissions have been estimated on the basis of a single and recent study (1998). Further studies are maybe required to confirm the estimated reduction potential of around 1.6 Mt, which represent more than 50% of reductions estimated in the agricultural sector.

A number of mitigation options were also identified in the energy sector (some recovery of mine methane, various measures in the oil and gas sectors), their potential reductions in 2010 have been estimated at around 1.7 Mt

[2] Cost savings more than offset the cost of the measure.

80% of total estimated reductions (7.2 Mt) could be reduced at a cost below 50 ECU/tonne CO_2 equivalent.

Sources	Reduction in 2010		Cost range	red. 2010/1990
	(kt)	(Mt CO2-equ.)	(ECU/t CO2-equ.)	(%) (*)
Total	**8999**	**189**		**-47%**
(BaU)				(-8%)
low cost measures	2883	61	< 0	-20%
medium cost measures	4337	91	0-50	-39%
high cost measures	1779	37	> 50	-47%
Agriculture	2874	60		
enteric fermentation	1600	34	< 0	
	83	2	> 50	
animal manures	969	20	0-50	
	222	5	> 50	
Waste				
landfill	4655	98		
	1110	23	< 0	
	2867	60	0-50	
	678	14	> 50	
Energy				
coal production	140	3	< 50	
oil and gas sectors	1330	28		
	173	4	< 0	
	361	8	0-50	
	796	17	> 50	

(*) cumulative, i.e. percentage reduction accounts for lower cost measures.

Table 4.1: Reductions and costs of CH_4 mitigation options

With all measures implemented, nitrous oxide emissions could be reduced by 385 kt (or 119 Mt CO_2 equivalent) in 2010. This would bring emissions 30% below 1990 levels. The largest reductions (Table 4.2) arise from measures directed at adipic and nitric acid production. Reductions of 237 kt result from the installation of abatement equipment at the main adipic acid manufacturing plants in the EU; further reductions of 39 kt are possible by abating emissions at the remaining adipic acid plants and at nitric acid plants.

The "package of options" identified for the agricultural sector offer the second greatest savings, with a reduction potential of 79 kt [9]. Measures identified in the other sector would contribute to less than 8% of total reductions.

Sources	Reduction in 2010		Cost range	red. 2010/1990
	(kt)	(Mt CO2-equ.)	(ECU/t CO2-equ.)	(%) (*)
Total	**385**	**119**		**-30%**
(BaU)				(8%)
low cost measures	3	1	< 0	7%
medium cost measures	277	86	0-50	-20%
agriculture	79	24		
by-product of CO2 red.	26	8		
Agriculture	79	24		
Waste	3	1	< 0	
Energy (combustion)				
Transport	10	3	(**)	
Other	16	5	(**)	
Industrial processes				
adipic acid production	261	81	0-50	
nitric acid production	16	5	0-50	

(*) cumulative, i.e. percentage reduction accounts for lower cost measures.
(**) costs not estimated; by-product of CO2 reduction policies

Table 4.2: Reductions and costs of N_2O mitigation options

The preliminary estimate of potential reductions of halogenated gases emissions reported in the Ecofys study, suggests that emission reductions of 72 Mt CO_2 equivalent would be feasible in 2010 and would bring total emissions of the three gases 82% below 1990 levels. The largest reductions (Table 4.3) arise from measures aimed at HFC reduction (61 Mt CO_2 equivalent).

More than 80% of total estimated reductions have a cost below 50 ECU/tonne CO_2.

Sources	Red. in 2010 (Mt CO2-equ.)	Cost range (ECU/t CO2-equ.)	red. 2010/1990 (%) (*)
Total halogenated gases	**72**		**-82%**
(BaU)			(41%)
low cost measures	0	< 0	41%
medium cost measures	59	0-50	-61%
high cost measures	12	> 50	-82%
HCF's			
HCFC-22 production	9	0-50	
refrigeration	9	0-50	
	12	> 50	
Other	30	0-50	
PFC's			
aluminium production	4	0-50	
SF6			
package of measures	7	0-50	

(*) cumulative, i.e. percentage reduction accounts for lower cost measures.

Table 4.3: Reductions and costs of mitigation options for the three halogenated gases

5. Conclusions

Emissions of non-CO_2 greenhouse gases under a business-as-usual scenario are projected to increase by 1% in 2010 compared to 1990/1995 levels. This is the result of the combination of a downward trend for methane emissions (-8%) and upward trends for nitrous oxide and the three halogenated gases emissions (+11% and +41% respectively).

This examination of measures to reduce non-CO_2 greenhouse gases emissions shows that such measures could make a substantial contribution to the achievement of the EU's Kyoto target.

The implementation of all measures identified would lead to a reduction of 380 Mt CO_2 equivalent in 2010. This would bring total non-CO_2 emissions to 43% (370 Mt CO_2 equivalent) below 1990 levels by 2010.

The EU six gas basket of emissions in 1990 is estimated to be about 4227 Mt CO_2 and a reduction of 600 Mt of CO_2 would be required to meet the EU's Kyoto target.

The identified reduction in non-CO_2 emissions is equivalent to 63% of total reduction needed, and would bring emissions of the six gas basket to 2.5% below 1990 levels by 2010.

Reductions from agricultural measures have been estimated at 85 Mt CO_2 (Table 5), these measures are potentially the most difficult to implement and estimates of their applicability and impact have still high level of uncertainty (this is particularly true for nitrous oxide and for methane from enteric fermentation).

Reductions from non-agricultural measures have been estimated at 295 Mt CO_2 (Table 5), of which half corresponds to reductions resulting from the implementation of existing and planned measures directed at landfilling of waste and adipic acid manufacturing plants. The cost-effectiveness analysis shows that 252 Mt CO_2 can be reduced at a cost below 50 ECU/tonne CO_2. With only non-agricultural measures implemented, emissions of the six gas basket are projected to stabilise at 1990 levels by 2010.

The emissions projections and reductions from measures reported in the two studies on CH4 and N2O and in this synthesis paper are compared in more detail in Annex 2.

Agricultural measures

	Em. reductions in 2010 (Mt CO2 equ)	Costs (ECU/t CO2 equ)
CH4		
	34	< 0
	20	0-50
	7	> 50
N2O		
	24	
Total non-CO2	**85**	

Non agricultural measures

	Em. reductions in 2010 (Mt CO2 equ)	Costs (ECU/t CO2 equ)
CH4		
	27	< 0
	71	0-50
	31	> 50
N2O		
	9	< 0
	86	0-50
HFC, PFC, SF6		
	0	< 0
	59	0-50
	12	> 50
Total non-CO2	**295**	
of which	36	< 0
	216	0-50
	43	> 50
Measures in place	146	
Additional measures	149	

Table 5: Synthesis of reductions and costs of non-CO_2 mitigation options

6. Comments

[1] Expert Group on Climate Change, "Community target - adjustments for extra gases and sinks resulting from Kyoto protocol", fax from DETR dated 19 February 1998; and
Expert Group on Climate Change, Comments to the letter "Burden sharing: data on greenhouse gas emissions and removal by sinks", fax from DETR dated 19 January 1998.

[2] In the AEAT nitrous oxide report, emissions from the agricultural sector were modelled using the revised IPCC methodology (IPCC, 1997) and for the EU were found to be 20% higher than those reported by Member States in their second national communications. This is believed to be due to the fact that not all Member States had yet adopted the revised IPCC methodology. In any case, N_2O emission estimates from agricultural soils have still very high levels of uncertainty.

[3] The only major difference between the studies concerns methane emissions from gas pipeline leakages. Although both studies based their calculation on the pre-Kyoto energy scenario (DGXVII, July 1997), AEAT assumes an overall better improvement of gas pipeline systems due to significantly lower leakage rates from new polyethylene gas distibution pipes, based on UK figures. Overall, methane emissions from oil and gas sectors are estimated to increase by only 10% between 1990 and 2010 in the AEAT study, compared to 37% in the Ecofys study. Gas consumption in the EU would increase by 98% over the same period. The above difference in BAU emission trend has little impact on the calculation of the reduction potential which is based on mitigation options independent of the leakage rate of new gas pipelines. In this paper, the Ecofys BAU emission trend was chosen.

[4] The AEAT business-as-usual projections of nitrous oxide emissions take into account the abatement measures taken at the main adipic acid manufacturing plants, while the Ecofys business-as-usual projections of nitrous oxide emissions do not. On the contrary, the Ecofys business-as-usual projections of methane emissions from landfills take into account the abatement measures taken or planned by the Member States (according to the *Expert Group's work on EU common and coordinated measures (landfill emissions)),* while the AEAT business-as-usual projections do not.

[5] Business-as-usual projections of methane emissions are similar to the ones reported in the AEAT report (a decrease by 9% in 2010 below 1990 levels), the difference comes from the oil and gas sector where emissions are predicted to increase less (see [3]); they are however far different from Ecofys' projections (a decrease by 26% in 2010 below 1990 levels) because the latter include the effect of existing and planned policies and measures to reduce emissions from landfills. If these measures were taken into account in the BaU projections reported here, methane emissions would have fallen by 22% in 2010 below 1990 levels.

Business-as-usual projections of nitrous oxide emissions are similar to the ones reported in the Ecofys report (an increase by 9% in 2010 above1990 levels), the small difference comes from the agricultural sector where emissions are predicted to decrease less; they are however far different from AEAT projections (a decrease by 15% in 2010 below 1990 levels) because these include the effect of existing measures to reduce emissions from adipic acid manufacturing plants (industrial processes). If these measures were taken into account in the BaU projections reported here, nitrous oxide emissions would have fallen by 16% in 2010 below 1990 levels.

[6] Emissions in 1990 provided in the above studies are not always in close agreement with emissions provided by the Member States and reported the EU Second Communication. This is believed to be due to the fact that Member States, Ecofys and AEAT have not used the same methodology for estimating emissions. In order to ensure consistency with national estimates, emission projections have been scaled by the difference in the 1990 emission estimates.

[7] According to AEAT and Ecofys' studies, current Member States policies would allow to reduce methane emissions from landfills by around 73 Mt of CO_2 and nitrous oxide emissions from adipic acid production by 73 Mt of CO_2.

[8] Both studies have used a bottom-up costing approach to estimate the cost-effectiveness of the various measures: only direct resource costs are taken into account, namely investment, operation and maintenance, energy costs and cost savings. The assessment does not include any positive or negative macroeconomic impacts that might occur if certain mainly economic instruments are used to implement the technical measures, nor any estimate of social resource costs or secondary environmental benefits. Costs are calculated as the ratio between the yearly costs of the measure and the resulting yearly emission reduction. The calculation of the yearly costs differ however slightly in the two

studies; in the AEAT study, a levelised cost/discounting method is used with a discounting rate of 8%; in the Ecofys study the total cost stream is only annualised over the lifetime of the equipment, the annuity factor is calculated with a discounting rate of 15%. Differences in methodology are however small compared to differences in costs associated with some mitigation options.

[9] The reduction potential is derived by scaling AEAT's reduction estimate of 114 kt by the ratio between AEAT's estimate of 1990 agricultural N2O emissions (using a revised methodology) and the value for 1990 reported in this paper in Table 2.

Annex

Comparison between figures provided in the studies and in the synthesis paper: CH4 and N2O

METHANE

	1990	2010 BAU		2010 without agr. measures		2010 with all measures included		reduction 2010 without agr.	reduction 2010 all measures	reduction 2010 from agr.
	Mt CO2	Mt CO2	% of 1990	Mt CO2	% of 1990	Mt CO2	% of 1990	Mt CO2	Mt CO2	Mt CO2
Ecofys (Coherence report)	492	364	-26%	290	-41%	239	-51%	74	125	50
AEAT (*)	490	443	-10%	314	-36%	294	-40%	129	149	20
Paper of 20 Oct.98	489	451	-8%	322	-34%	262	-46%	129	189	60
		(1) (2)		(3)						(4)

NITROUS OXIDE

	1990	2010 BAU		2010 without agr. measures		2010 with all measures included		reduction 2010 without agr.	reduction 2010 all measures	reduction 2010 from agr.
	Mt CO2	Mt CO2	% of 1990	Mt CO2	% of 1990	Mt CO2	% of 1990	Mt CO2	Mt CO2	Mt CO2
Ecofys (Coherence report)	313	342	9%	250	-20%	229	-27%	92	113	21
AEAT	377	322	-15%	302	-20%	266	-29%	20	56	36
Paper of 20 Oct.98	315	339	8%	244	-23%	220	-30%	95	119	24
	(5)		(6)				(7)	(6)		

(*) Figures for 2010 have been revised from the September report.

EU's Kyoto target

	Basket of six 1990/1995 Mt CO2	Basket of six 2010 BAU Mt CO2	Mt CO2 Reduction needed in 2010
Ecofys (Coherence report)	4228	4405	515
AEAT	4290	4464	517
Paper of 20 Oct.98	4227	4489	600
	(8)	(8)	

Contribution of GHG to EU's Kyoto target

	Reduction needed in 2010	CH4 non-agr.	CH4 agr.	N2O non-agr.	N2O agr.	halog. gases	CO2
Ecofys (Coherence report)	515	74	50	92	21	72	205
AEAT	517	129	20	20	36	72	240
Paper of 20 Oct.98	600	129	60	95	24	72	220
						(9)	(10)

Explanation of major differences

(1): Ecofys' BAU scenario includes existing and planned measures to reduce CH_4 from landfills. The reduction potential in 2010 is therefore lower than in AEAT study.
AEAT study and the paper do not incorporate in the BAU scenario CH_4 abatement measures in place.

(2): AEAT's BAU assumes significant reductions from new gas distribution pipes based on UK figures (leakage rate from new pipes is calculated as a percentage of leakage rate of old distribution pipelines in the UK; the same percentage is applied in all MS despite the fact that MS other than UK have average leakage rates of existing distribution network far lower than in the UK). However, the AEAT methodology has the advantage to distinguish between the transmission and distribution systems for additional gas consumption, which is important as the latter is the main source of methane emissions.
The Ecofys approach is more global in the sense that methane emissions are estimated on the basis of total gas throughput (or inland consumption) and an average "target" leakage rate for the overall gas network of at most 0.5% in 2010 (the average leakage rates in the MS ranged from 0.5% to 2% in 1990). One may however argue that it is too approximative as it does not account for the characteristics of the national gas networks (e.g. operating pressure, length of transmission network versus length of distribution network, length according to the material of the pipes etc).

(3): The level of emissions is lower in Ecofys' study because it assumes higher reductions from measures in the gas sector (compressors and replacement of old grey cast iron pipes)

(4): Agricultural measures are different in Ecofys and AEAT syudies. The reduction potential is higher in the synthesis paper because it sums up the effect of measures in both studies.

(5): In the AEAT study, agricultural emissions for 1990 and future years have been calculated using the revised IPCC methodology; for the EU they were found to be 20% higher than those reported in the EU second communication and used by Ecofys and in the synthesis paper.

(6): AEAT' s BAU scenario includes existing N_2O abatement plans at the main adipic acid manufacturing plants; so BAU emissions are lower than in the two other analyses.
The reduction potential in 2010 is therefore lower than in the other two analyses. Ecofys' study and the paper do not incorporate in the BAU scenario N_2O abatement measures in place.

(7): Emission projections with all measures included are lower in the paper than in the other two studies because it sums up the effect of measures considered in the two studies (no measure in the waste water sector in AEAT's study (reduction of 1 Mt CO_2); no measure in the energy sector in Ecofys'study (reduction of 8 Mt CO_2)).

(8): CO_2 and halogenated gases emissions as in Table 3 of the paper.

(9): Reduction potential taken from Table 4.3 of the paper.

(10): CO_2 contribution to the EU's Kyoto target if all measures identified for non-CO_2 greenhouse gases are implemented irrespective of their cost and political applicability.
If only abatement measures with a cost lower than 50 ECU/t CO_2 were implemented, CO_2 contribution would be higher by 20 to 30% (270 to 300 Mt).