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Cost of abating greenhouse gas emissions from UK dairy farms by anaerobic digestion of slurry

by

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Thesis for the degree of Doctor of Philosophy

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ABSTRACT

ENGINEERING AND THE ENVIRONMENT

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COST OF ABATING GREENHOUSE GAS EMISSIONS FROM UK DAIRY FARMS BY ANAEROBIC DIGESTION OF SLURRY

Sarika Jain

As a sector, agriculture in the UK is responsible for 43% of the methane (CH_4) and 80% of the nitrous oxide (N_2O) emissions, greenhouse gases (GHG) with global warming potentials of 21 and 310, respectively. The UK government is providing financial subsidies to reduce GHG emissions, particularly in energy production. These subsidies primarily come in the form of feed-in tariffs (FITs) and renewable heat incentive (RHI) to the renewable energy industry. Given that the traditional, fossil-fuel based energy industry's GHG footprint is 96% in the form of carbon dioxide (CO_2), a policy based on renewable electricity and heat production is primarily rewarding CO_2 abatement and fossil fuel substitution. This is appropriate for most renewable energy technologies except anaerobic digestion (AD) which, besides producing energy, also has the potential to abate substantial amounts of CH_4 and N_2O. Dairy farms produce large quantities of cattle slurry which are suitable for AD but have low energy potential, thus providing poor economic return on capital investment even after claiming the subsidies available. An alternative subsidy could be provided by marginal abatement cost (MAC) which gives a value for GHGs abated. This research shows that after incentives dairy farmers bear a marginal abatement cost of £27 tonne^{-1} CO_2 eq. abated, a key factor in low uptake of on-farm AD in the UK.

Contents

List of tables

List of figures

List of accompanying materials

CD containing spread-sheet model

DECLARATION OF AUTHORSHIP

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declare that the thesis entitled

Cost of abating greenhouse gas emissions from UK dairy farms by anaerobic digestion of slurry

and the work presented in the thesis are both my own, and have been generated by me as the result of my own original research. I confirm that:

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Signed: ……………………………………………………………………..

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Definitions and Abbreviations

AD Anaerobic Digestion

CH_4 Methane

CHP Combined Heat and Power

CO_2 Carbon dioxide

FIT Feed-in Tariff

GHG Greenhouse gas

GWP Global Warming Potential

LU Livestock Unit

MAC Marginal Abatement Cost

N_2O Nitrous Oxide

NH_3 Ammonia

RHI Renewable Heat Incentive

ROC Renewable Obligation Certificate

Nomenclature

º Degree

Σ Greek capital letter sigma, Summation

^ Exponential

β Greek small letter beta

Δ Greek capital letter delta, change in

1. Introduction

> "Warming of the climate system is unequivocal, as is now evident from observations of increase in global average air and ocean temperatures, widespread melting of snow and ice and global average sea level…Other effects of regional climate changes on natural and human environments are emerging, although many are difficult to discern due to adaptation and non-climatic drivers…Most of the observed increase in global average temperatures since the mid-20th century is very likely due to the observed increase in anthropogenic GHG concentrations." (IPCC, 2007).

Governments now offer a range of financial incentives to reduce the emission of greenhouse gases (GHGs) from fossil fuel use, particularly in energy production (DECC, 2012a, DECC, 2012b). In the United Kingdom (UK), electricity is produced mainly through the combustion of fossil fuels (gas (41%) and coal (29%)) (DECC, 2010a) which accounts for 40% of the all carbon dioxide (CO_2) emissions. The agricultural sector is responsible for 43% of methane (CH_4) and 80% of nitrous oxide (N_2O) emissions in the UK (DECC, 2010a), GHG gases with significantly higher global warming potentials (GWP). Livestock enteric emissions, emissions from manure and soil management constitute the bulk of these and are not affected by the majority of renewable energy technologies which are targeted towards energy production from non-fossil fuel sources like sunlight and water. These, however, do not contribute towards reducing wastes and the associated emissions.

The UK Low Carbon Transition Plan 2009 aims to cut emissions from waste and farming by 6% of 2008 levels by 2020 through:

- Efficient use of fertilisers and better management of livestock manure;
- Support for Anaerobic Digestion, a technology that turns waste and manure to renewable energy;
- Reducing the amount of waste sent to landfills and better capture of landfill emissions (HM-Government, 2009).

Anaerobic digestion (AD) is a proven technology which breaks down biomass (animal and plant material) in the absence of air to produce biogas and digestate. The biogas produced may be used to generate heat and electricity via a CHP unit or used directly as fuel. In the case of farm based digesters, using cattle slurry as a feedstock, this has the dual benefit of reduction in emissions from the manure and generation of renewable energy (heat and electricity). This not only reduces the operating expenditures of the farm by substitution of imported energy with on-farm produced energy but also adds a revenue stream from any energy exported.

Hence, AD can be used to reduce emissions from manure management whilst also providing a source of renewable energy, and is suitable for farm application at varying scales.

The uptake of on-farm AD has been low in the UK with only 39 farm-sourced digesters operating as of 10/12/2012 (Defra, 2013). This is due to relatively high capital costs which cannot be efficiently recovered under the current financial incentives linked to heat and electricity (Feed in tariff, Renewable Heat Incentive and Renewable Obligation Certificates). The current financial incentives linked to the amount of renewable energy produced are essentially targeting reduction of CO_2 emissions and fossil fuel substitution and not recognising the part AD can play in the reduction of other GHGs, in particular CH_4 and N_2O.

In order to quantify the extent of support for a dairy farm to make AD economically feasible, it is essential to determine the extent of abatement of GHGs resulting from the introduction of a digester and the reduction in farm profit due to additional AD related expenditure. The current farm based economic (Jones, 2010, Kottner *et al.*, 2008, Redman, 2010) and emissions models (Olesen *et al.*, 2006, Rotz and Montes, 2009) do not fully capture the dynamics of costs and the benefits of AD on a dairy farm. This UK based research presents the development of GHG emission and economic models for dairy farms, including the impacts of the introduction of AD. By modelling the amount of GHG emissions abated through the introduction of AD and comparing this with the economic costs and gains, a marginal abatement cost (MAC) is determined.

Every farming situation is different; but through the application of sensitivity analysis and Monte Carlo simulations it is possible to derive MACs for a range of circumstances. MAC of AD can be compared to that of other technologies and also provides a good benchmark to understand the cost that the farmer has to bear for CH_4 and N_2O abatement, which is not incentivised in the current policy framework. The methodology developed can provide input for renewable energy schemes and GHG reduction methods.

1.1 Aims and Objectives

Aims

The aim of this research is to evaluate the potential of on-farm anaerobic digestion in mitigating greenhouse gas emissions from dairy farming in the UK and the associated economic costs/benefits. This aim is fulfilled by completing the following objectives.

Objectives

1. **To quantify the GHG emissions from a typical dairy farm in the UK.**

This objective is met by developing an emissions model quantifying emissions from different sources on a dairy farm operating without and with a digester. This will determine the total greenhouse gas emissions that are abated by anaerobically digesting cattle slurry on a dairy farm.

2. **To quantify the economics of a dairy farm.**

This objective is fulfilled by developing an economic model quantifying the expenditures and revenues added or reduced by anaerobically digesting the slurry collected from a typical dairy farm in the UK. This will quantify the cost of digestion to the farmer. For ease of comparing multiple scenarios both the economic and emission models are developed using Microsoft Excel.

3. **To calculate the marginal abatement cost of AD and identify the input parameters that the model is most sensitive to.**

This objective brings together the emissions and economic models to calculate a marginal abatement cost for GHG abatement for a predefined “Modelled farm”. Sensitivity analysis is conducted by identifying the plausible range of input parameters for a dairy farm in the UK and the corresponding change in the marginal abatement cost. This helps in identification of the most important input parameters and the conditions of environmental and economic profitability.

4. **To calculate the range of marginal abatement cost under varying farming and digester operating conditions expected on dairy farms in the UK.**

This objective is completed by conducting Monte Carlo simulations for plausible values for the most sensitive input parameters identified by sensitivity analysis. This helps in determining the most probable and the expected range of marginal abatement cost to the farmer.

1.2 Contribution to knowledge

There are economic and environmental models available for the evaluation of feasibility of AD; however, these lack transparency, applicability to the UK farming methods, and detail. This research aims to develop detailed and exhaustive emissions and economic models for a dairy farm in the UK that fully capture the costs and benefits associated with AD. By developing a method to evaluate the cost to the farmer for mitigating each tonne of carbon dioxide equivalent emission, this research will bridge the gap between the environmental impacts and economic incentives that may be required in order to encourage the uptake of AD in the UK.

1.3 Outline of the thesis

The thesis is arranged as follows:

Chapter 1 provides the introduction to the thesis, the context in which the research took place, the aims and the objectives.

Chapter 2 reports literature pertinent to this research. Literature available on the practice of dairy farming in the UK is discussed in Section 2.1. Section 2.2 focuses on GHG emissions from dairy farms while Section 2.3 presents the current knowledge on emissions and economics of AD in the UK. The different methods of determining the cost of emission or value of abating carbon are reviewed in Section 2.4.

Chapters 3 reports the farm model which calculates the infrastructure related parameters that are used by the emissions and economic models as input, including but not limited to herd size, manure collection and digester sizing.

Chapter 4 reports the emissions model which calculates the amount of GHG emissions abated from a typical dairy farm in the UK from the introduction of an anaerobic digester. The model allows the study of this difference in emissions at a sub component level e.g. emissions from manure management. The model also accounts for additional emissions generated as a consequence of introduction of AD e.g. from the construction of an anaerobic digester, fugitive biogas emissions, etc.

Chapter 5 presents an economic model for assessment of typical dairy farming activities both pre and post introduction of AD. The model accounts for all the relevant revenue streams like sale of electricity and heat, investment costs (construction, etc.) and running costs (labour, etc.) related to the construction and functioning of an on-farm anaerobic digester.

Chapter 6 reports the calculation of MAC by comparing the emissions abated on introduction of AD with the difference in profit from the same. The method is implemented for study of the modelled farm and sensitivity and Monte Carlo analyses.
Figure 1 presents how the different models and analyses are linked.

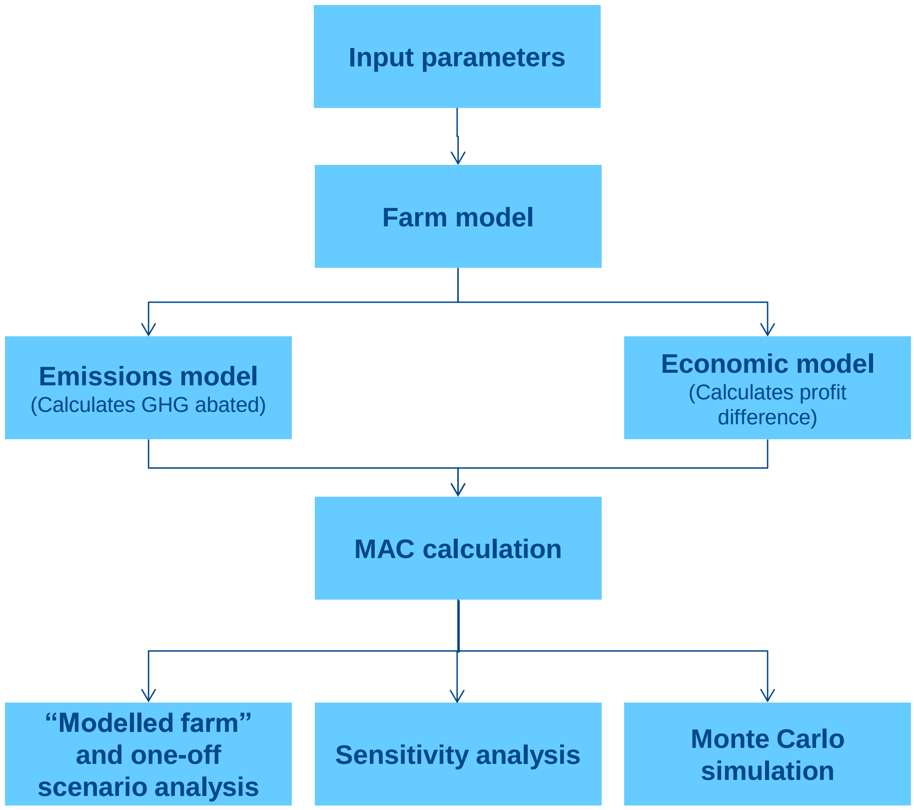


Figure 1 Flowchart of model

Chapters 7, 8, 9 and 10 report the results obtained by implementing the farm, emissions and economic models for a modelled dairy farm. These are followed by results of the sensitivity analysis and the Monte-Carlo simulations when key input parameters are varied one at a time and simultaneously, respectively.

Chapter 11 concludes this research with ideas on future work that may be undertaken to fully capture the industry and fill in the data gaps in the current knowledge.

The appendix contains the code written to carry out the Monte-Carlo simulation. Throughout the thesis pictures have been included showing screen shots of the relevant spread-sheet module.

2. Literature review

This chapter presents literature reviewed in order to build the emissions and economic models for a dairy farm and develop the methods required to determine the MAC. Literature available on the practice of dairy farming in the UK, its current status, some pertinent regulations along with methods to reduce GHG emissions from dairy cows and their slurry have been discussed in Section 2.1. The impact of greenhouse gases on the climate, sources of these on a dairy farm and programs available to model the same have been discussed in Section 2.2. Section 2.3 reviews AD on a dairy farm in the UK. This includes the benefits of AD, its current status in the UK, sources of GHG emissions related to AD, the economics of AD and models available to facilitate the evaluation of economics and, pertinent policies and regulations. The different methods of determining the cost of carbon have been reviewed in Section 2.4.

2.1 Dairy farming in the UK

A dairy farm is defined as a holding on which dairy cows account for more than two thirds of the total standard gross margin for the farm (McHoul *et al.*, 2012).

2.1.1 Current status

Dairy farming is the single largest agricultural sector and accounts for 17% of the UK agricultural production by value (Defra, 2010a). Dairy farming in the UK is concentrated in areas which have an advantage of good grass growing conditions, in particular the South West of England, the lowland areas of south and south-west Wales, the north Midlands and North West of England and the lowland areas of Northern Ireland and of south-west Scotland (Hopkins and Lobley, 2009). Dairy farming has seen a steady decline in the past decade in the UK. The number of dairy farmers in England reduced from 18,695 in 2002 to 10,851 in 2011; a fall of 42%. This number is equivalent to 80 dairy farmers going out of business per month for 9 years (DairyCo, 2012b). Furthermore, a survey by DairyCo (2011) showed that 13% of the dairy farmers interviewed planned to leave the industry in the next 2 years. Lack of successors, low milk prices and high input costs were cited as the greatest concerns. Diversification of farm income can make the farm profitability immune to the price volatility of inputs like feed, fertilisers, fuel, etc. Despite the decline in number of dairy farms and cows, increasing milk yields have kept the milk output stable and the UK is currently the ninth largest milk producer in the world (Hopkins and Lobley, 2009).

2.1.2 Dairy farming systems

Dairy cows in the UK are reared on specialist farms that have adopted relatively intensive farming methods. Dairy farming in the lowlands is based on efficient management of improved grassland forage and supplementation with predominantly UK-sourced feeds. The grass (predominantly ryegrass) receives moderate to high rates of mineral nitrogen fertilisers (mean rates of about 120 kg N ha^{-1} on dairy swards). Feeding is based on grazing herbage that is leafy and of high digestibility with surplus herbage from spring and summer conserved as silage for indoor winter feeding. Most dairy herds are kept indoors during winter for up to 6 months but this may vary depending on soil types and weather conditions (Hopkins and Lobley, 2009). In the UK, confined dairy cows may be housed in cubicles, straw yards or kennels depending on the economics and the availability of bedding material. Bedding may consist of straw, sawdust, sand, recycled paper, lime ash or gypsum. Bedded area of at least 7.5 m^2 and loafing area of 3 m^2 per cow is required for housed cows (Dairyco, 2012a). Dairy cows are moved from their stalls to the milking parlour twice in a day for milking. 10 to 15% of the manure is deposited in the milking parlour and the holding area.

The manure deposited in the barn and the parlour may be flushed or scraped. While flushed systems are economical, effective and require less labour, they dilute the manure substantially making it unsuitable for digestion. Tractor mounted scrape systems are time consuming and can be operated only when the cows are away. Keeping dairy cows on a slatted floor with underground slurry storage is another option. The type and shape of slats play an important role in the comfort and health of the cow (Dairyco, 2012a). The slurry collected may be stored in: clay or HDPE lined lagoons, slurry bags, steel tower or concrete store.

The average herd size has increased from 96 dairy cows and heifers in milk per farm in England in 2005 to 145 dairy cows per herd in 2010 (Defra, 2011a). 32% of the cattle in Great Britain are black and white which includes Holstein, Friesian and cross-bred animals of these breeds (Defra, 2008). Holstein Friesians yield 6000 to 9000 litres milk per year depending on the intensity of farming (Nix, 2012). The average milk yield for a conventionally reared lowland dairy cow is 7406 litres year^{-1} (McHoul *et al.*, 2012).

2.1.3 Nitrogen Vulnerable Zones

In response to the European Union's Nitrate Directive (91/676/EEC) (The European Parliament and the Council of the European Union, 2009), most countries in the EU have a nitrogen limiting system for nutrient management which has been implemented by identifying Nitrogen Vulnerable Zones (NVZ). In some countries this is applied on a regional basis; some, including Austria, Denmark, Finland, Germany, the Netherlands and Luxembourg have designated the whole country Nitrogen Vulnerable (European Commission, 2002).

In order to limit the loss of nitrogen to water, Defra has designated certain areas of the UK as Nitrogen Vulnerable Zones (NVZ). NVZ are areas of land draining into waters which have the potential to be polluted by nitrates from agriculture. About 68% of England now lies in Nitrogen Vulnerable Zones. Farms lying within these zones are required to follow certain manure and nutrient management guidelines to minimise leaching losses (Defra, 2009). There are a number of manure storage and application restrictions that must be adhered to and have been listed below.

2.1.3.1 Manure storage requirements

Manure storage requirements have been specified by European countries in order to minimise the emissions from application to the field. Table 1 summarises the manure storage requirements in some EU countries.

Table 1 Manure storage requirements in European countries (Jakobsson *et al.*, 2002)

Country	Storage Capacity (Lowest minimum requirement), months	Storage Capacity (Highest minimum requirement), months
Austria	6	6
Belgium	4	4.5
Denmark	6	9
Finland	12	12
France	4	6
Germany	6	6
Greece	3	6
Ireland	2	6
Italy	3	6
Norway	8	-
Portugal	1	3
Spain	2	4
Sweden	6	10
Switzerland	3	7
The Netherlands	5	-
The UK	5	

2.1.3.2 Spreading bans

Almost all countries in the EU have restrictions on the winter spreading of manure/slurry to land. Bans on application of organic fertiliser to snow-covered, deeply frozen or saturated soil are also in place throughout the EU. In the UK, in designated NVZs, application of organic manures is banned in the periods presented in Table 2. Use of high trajectory slurry spreaders is also banned.

Table 2 Periods of non-application of organic manures in NVZs (Defra, 2009)

Grassland		Tillage land	
Sandy or shallow soils	All other soils	Sandy or shallow soils	All other soils
1 Sep – 31 Dec	15 Oct – 15 Jan	1 Aug – 31 Dec	1 Oct – 15 Jan

Manufactured nitrogen fertiliser may not be applied to land during the periods presented in Table 3.

Table 3 Periods of non-application of fertilisers in NVZs

Grassland	Tillage Land
15 Sep – 15 Jan	1 Sep – 15 Jan

2.1.3.3 Maximum application rates

Maximum application rates of manure/slurry or mineral fertilisers exist in several countries in the EU. In the UK the maximum loading is 170 kg per hectare of total N produced by livestock in each calendar year averaged over the area of the holding or land. Farmers with more than 80% of the farm as grassland may be able to operate at a higher limit of 250 kg of total N under a derogation approved by the European Commission (European Commission, 2002).

2.1.4 GHG emission reduction from dairy cows

There are multiple ways in which a dairy farmer can reduce the GHG emissions from his enterprise. Some of these are animal management methods while others are related to management of manure. Some of the animal management methods have been listed below:

- Improving feed conversion efficiency – There are a number of ways of improving the feed conversion efficiency in a dairy cow which leads to higher milk yield per cow and reduced emissions per unit of milk. Also, fewer dairy cows are required to obtain the same amount of product leading to further reduction in emissions. Feed conversion efficiency may be improved by increasing concentrates in the diet and increasing the proportion of maize silage in the diet.

- Probiotics divert hydrogen from methanogenesis to acetogenesis which increases the amount of acetate and reduces the amount of methane emitted from enteric fermentation.
- Ionophores improve the efficiency by decreasing the dry matter intake of the animal and increasing milk production. The use of ionophores has been banned in dairy farming as these are used in human medicines and continued use in farming may compromise their effectiveness as a medicine.
- Bovine Somatotrophin (bST) has been shown to decrease CH_4 emissions but its use is widely unacceptable to European consumers due to potential detrimental impact on animal health.
- Breeding for improved efficiency - Genetic improvement of milk yield, fertility and other desirable traits in the dairy cow and transgenic offspring may be another method of improving efficiency and reducing the number of animals and thus the emissions (Moran *et al.*, 2008).

2.1.5 Methods of reducing emissions from slurry management

Emissions from the management and use of slurry on a farm may be reduced by implementing the following measures as recommended by Defra (2009)

1. Have a nutrient management plan to apply fertilisers to meet and not exceed the crop requirement.
2. Spread organic manure such that application coincides with period of growth of plant and uptake of nitrogen.
3. Do not apply organic manure in periods when risk of run off is high, i.e. in winter and when the ground is saturated or frozen.
4. Avoid windy days for organic manure application as these lead to higher ammonia losses.
5. In order to meet the requirement of the crops, conduct field experiments to assess the quality of soil, nutrient requirement for the crop grown and the nutrient composition of slurry.

6. Using equipment that has a low spreading trajectory reduces emissions from volatilisation of nitrogen. Using band spreaders and shallow slurry injectors where possible to reduce emissions and increase uptake of nitrogen.
7. By incorporating organic manure into the soil as soon as it is practical (within 24 hours) when applied to bare land or stubble.
8. Allowing ample time between slurry and mineral fertiliser application (Defra, 2009).
9. Covering slurry tanks and lagoons (Moran *et al.*, 2008).
10. Use of aerobic tanks and lagoons reduces the methane emissions from storage slurry (Moran *et al.*, 2008) but may lead to higher nitrous oxide emissions.

2.2 GHG emissions

2.2.1 Climate change and greenhouse gases

The surface temperature of the Earth is determined by the balance of the incoming solar energy absorbed by the Earth's surface and the energy re-emitted in the form of infra-red radiation. This re-emission has a cooling effect on the Earth. GHGs trap some of this radiation, however, which results in warming the surface of the Earth and lowering atmospheric temperature. This effect, known as the Greenhouse effect, has been in operation for millions of years. The accumulation of greenhouse gases due to human activities (carbon dioxide, methane, nitrous oxide, and halogenated carbons) has disturbed this balance resulting in warming of the Earth (DECC, 2012e). As a result, changes in the hydrological and terrestrial, marine and freshwater biological systems have been observed. An increase in average air and ocean temperatures resulting in melting of snow and ice and rising global average sea levels have also been observed (IPCC, 2007). Global increases in CO_2 concentrations are primarily due to use of fossil fuels and changes in land use. Increased CH_4 emissions have been attributed to both agriculture and fossil fuel use while nitrous oxide emissions are particularly related to agriculture (IPCC, 2007). In the UK, agriculture is responsible for 9% all GHG, 44% of all methane and 80% of all nitrous oxide emissions (DECC, 2010a). The potential climate change impact of the greenhouse gases can be compared using GWP. Table 4 summarises the GWP of greenhouse gases over varied time periods. The United Nations Framework Convention on Climate Change (UNFCCC) has recommended values of GWP reported in the Second Assessment Report (SAR) of IPCC for use in company reporting of GHG emissions (IPCC, 1996).

Table 4 Global Warming Potentials relative to carbon dioxide (IPCC, 2006)

	Global Warming Potentials for given time horizon (years)			
Gas	**100 (SAR)**	**20**	**100**	**500**
Carbon dioxide	1	1	1	1
Methane	21	72	25	7.6
Nitrous oxide	310	289	298	153

2.2.2 Emissions models available

A number of models are available for the estimation of GHG emissions from a dairy farm. Some of these models and their applicability to the project are discussed below.

IPCC (2006) The Intergovernmental Panel on Climate Change (IPCC) has developed a series of equations based on the source of emission to calculate the national GHG inventories whose reporting was made mandatory under the United Nations Convention of Climate Change (UNFCCC) via the Kyoto Protocol (UNFCCC, 1997). The IPCC methodology may be implemented at three levels, or Tiers, of detail and complexity based on the data available and applicability to the current farm setting. Tier 1 is a simplified methodology based on default emission factors specified for the region when no country specific data is available. Tier 2 is a more complex approach that requires detailed country specific data and is recommended if the source of emission is a key source category that represents a large portion of the country's total emission while Tier 3 is a detailed approach that may employ development of sophisticated models and direct experimental measurements. This methodology is subject to extensive international peer review to ensure accuracy of estimates.

Thus, the methodology outlined in IPCC (2006) which is authoritative and globally accepted methodology has been used as the basis of the emissions model as it is generic enough to be applied globally while being able to incorporate country/farm specific variations.

Salter and Banks (2009) developed a UK based AD tool which calculates the energy balance on a whole farm basis in the presence or absence of a digester. The model is capable of evaluating both arable and dairy farms that may accept other feedstock like food-waste as additional input. The calculations are based on basic farm parameters like the size of the farm, the areas of various crops cultivated, the number of livestock kept on farm, etc. These may be altered in order to evaluate specific farming conditions. The energy balance takes into account both direct and indirect uses of energy and its production on farm. The direct and indirect uses include diesel fuel use by farm machinery, energy required in production of mineral fertilisers used on farm, parasitic loads, embodied energy in digester, etc. The model is spread-sheet based and is available to the public and is supported by a manual. The advantage of this model is that it is comprehensive, transparent and flexible.

Holos software (Little *et al.*, 2008) developed at Agriculture and Agri-food Canada, is a whole farm modelling software program that estimates GHG emissions (CO_2, CH_4 and N_2O) based on inputs entered or scenario chosen by the user. The model is based on IPCC methodology that has been modified for Canadian conditions. Besides the GHG emissions from enteric fermentation, manure management, cropping systems and energy use, carbon storage and loss from lineal tree plantings and changes in land use and management have been included. It may be used for evaluating methods of GHG emission reduction since it is not directly applicable to the UK farming sector.

Canadian Economic and Emissions Model for Agriculture (Agriculture and Agri-Food Canada, 1999) has been developed to calculate the GHG emissions from agriculture in Canada and to estimate impacts of agricultural policies on GHG emissions. The earlier version CEEMA1.0 integrated the already existing Canadian Regional Agricultural Model (CRAM) with a Greenhouse Gas Emissions Sub-Model (GHGEM). CEEEA 2.0 is based on IPCC data or Canadian data where it is available. The model uses data that is specific to Canada and is used to estimate impact of Canadian policies on emissions from various regions in Canada and hence is not suitable for use in this study.

CLA CALM Carbon Accounting for Land Managers by Country Land and Business Association in association with Savill (CALM, 2009). It is an activity based model calculating emissions from energy and fuel use, livestock, cultivation and land-use change, and application of nitrogen fertilisers and lime for the UK. The emissions are balanced against carbon sequestration in soil and trees. IPCC methodology and UK GHG inventory is used for calculations. It is available as a web based application. The calculator requires farming data like energy used, mineral fertilisers bought as input. Hence, it can evaluate an already existing farm but not estimate values for a planned one. It does not include the option for digestion of slurry and hence cannot be used in our study.

FarmGHG developed by Olesen *et al.* (2006) at the Danish Institute of Agricultural Sciences is one of the models available for estimating GHG emissions from a whole farm. It is designed to quantify the flow of carbon and nitrogen on a dairy farm. It has been developed in Delphi and is not very user-friendly. It is a useful tool for a user who knows and understands Delphi.

The Dairy Greenhouse Gas Model, version 1.2 (DairyGHG) (Rotz and Montes, 2009) has been developed to provide a simple tool for predicting the integrated net greenhouse warming potential of all GHG emissions from dairy production systems. Secondary emissions from the production of farm inputs such as machinery, fertiliser, fuel, electricity, and chemicals are also included to determine an overall carbon footprint for the production system. This model has been developed for dairies in the USA. The weather information is specific to the states in the USA. It does not allow for a fully grazed dairy or for an on-farm anaerobic digester. Therefore, it is unsuitable for use in our study even though it is based on IPCC Tier 1 and 2 methodologies.

None of the models currently available conduct a transparent whole farm analysis of a dairy farm in the UK and evaluates the full emissions benefits of introduction of an anaerobic digester. A new model was built in this study to fill this gap in knowledge.

2.2.3 Sources of GHG emissions on a farm

GHG emissions originate from a number of sources on dairy farms including livestock, livestock manure, crop production and energy use in dairying etc.

The introduction of an AD unit to the farm leads to potential changes in GHGs emitted. The following sections present the findings from the review of literature for these different potential sources of emissions.

2.2.3.1 Enteric emissions

Methane is produced in herbivores as a by-product of enteric fermentation, a digestive process by which carbohydrates are broken down by micro-organisms into simpler molecules for absorption into the bloodstream (IPCC, 2006). Methane produced in the rumen is exhaled or belched out by the cow and accounts for a majority of the methane emissions from ruminants. Methane is also produced in the large intestine of ruminants and is expelled. The amount of methane that is released depends on the type of digestive tract, age and weight of the animal, and the quality and quantity of the feed consumed (IPCC, 2006).

Enteric emissions account for about 60% of the total GHG emissions from dairy farming. The IPCC 2006 guidelines for National Greenhouse Gas Inventories (IPCC, 2006) recommend Tier 1 emission factors of 109 kg CH_4 head^{-1} year^{-1} for dairy cows and 57 kg CH_4 head^{-1} year^{-1} for other cattle for Western Europe based on the compiled data and opinion of the IPCC expert group. For a more accurate estimate of enteric emissions in the UK from the data available in the literature, Tier 2 IPCC 2006 methodology has been used (IPCC, 2006).

The emissions model developed in this study assumes grazed and grass fed dairy cows, hence literature was searched for emission factors for these particular conditions. The findings are summarised in Table 5.

Lassey *et al.* (1997) measured daily methane emission rates from 10 lactating Friesian dairy cows using the ERUCT technique (Emissions from Ruminants Using a Calibrated Tracer). This was done by placing a known amount of tracer (sulphur hexafluoride, SF_6) in the rumen of the cow, sampling the breath of the cow while grazing on ryegrass and clover and analysing it for CH_4 and SF_6. The average of the methane emissions was 262.8 ± 9.6 g CH_4 head^{-1} day^{-1} which is equivalent to 96 ± 4 kg CH_4 head^{-1} year^{-1}. This figure is slightly lower than the 109 kg CH_4 head^{-1} year^{-1} recommended by the IPCC. The methane emissions accounted for 6.19 ± 0.15% of the gross energy intake calculated based on

IPCC Tier 2 methodology which is in line with the IPCC default methane conversion factor of 6.5 ± 1%.

Ngwabie *et al.* (2009) measured methane emissions from a naturally ventilated barn that housed dairy cows that were lactating or pregnant using photo-acoustic multi-gas analyser and a multiplexer. During the winter months when they were fully housed, the average methane emission rate per head was 11.95 g CH_4 hour^{-1} which is equivalent to 104.6 kg CH_4 head^{-1} year^{-1}. This is slightly higher than the IPCC recommended value; which is reasonable as there will be some emissions from manure dropped by the cows in the barn and from the deep litter on which the pregnant cows were housed. They also measured emissions from the barn during summer when the cows were grazed during the day and were housed in the night. The average for the week of May when the measurements were taken was 79 kg CH_4 head^{-1} year^{-1}.

Laubach and Kelliher (2004) and Laubach and Kelliher (2005) reported the results of a series of experiments measuring methane emissions from herds of cows using different methods. The methane measurements were taken by open path laser measurements and vertical profile mast. Flux gradient technique (FG), integrated horizontal flux technique (IHF) and backward-Langragian stochastic models (BLS) were used to calculate the emission factors. The emission factor obtained by open-path lasers in conjunction with BLS was 402+/- 52 g head^{-1} day^{-1} which is 146 kg head^{-1} year^{-1}. The average from IHF and profile mast measurements was 343 +/- 38 g head^{-1} year^{-1} and from BLS and vertical profile 390 +/- 38 g head^{-1} year^{-1}. These are high numbers as they include emissions from the manure deposited and were taken on a commercial dairy farm, where due to the poor quality of grass available, large amounts of grass silage was fed to the cows to maintain the high rate of milk production. The focus of these experiments was development of the techniques of measurement rather than estimating an emission factor.

Snell *et al.* (2003) measured methane from four naturally ventilated barns housing dairy cows and followers. The rate of emission from dairy cows ranged between 97 and 285 kg head^{-1} year^{-1}, depending on the manure removal system. The highest emission rate came from the building in which manure was being deposited inside the building. Diet related information was not provided.

Bruinenberg *et al.* (2002) synthesised enteric emissions data collected in 3 different laboratories in the Netherlands in the late 1970s and 1990s. 96 data points were collected from dairy cows fed on grass with less than 10% of the feed as concentrate. The average percentage of energy lost in methane was about 6% of the gross energy. Based on Tier 2 calculations of energy requirement and gross energy consumed by a dairy cow, this comes to about 105 kg CH_4 cow^{-1} year^{-1}, which is in line with the recommended IPCC value.

Pinares-Patino *et al.* (2007) studied the effect of stocking rate on the enteric emission rate of grazing heifers using the ERUCT technique. They found that the absolute methane emissions did not vary significantly with the stocking rate and year to year. Grainger *et al.* (2007) carried out methane emission experiments using SF_6 as tracer and chamber techniques on cows grazed all year on ryegrass sward pasture. The average methane emission measured by the chamber technique was 322 ± 57.5 g day^{-1} (117 kg cow^{-1} year^{-1}) and that from SF_6 tracer technique was quite close at 331 ± 74.6 g day^{-1} (120 kg cow^{-1} year^{-1}). These numbers are slightly higher than those recommended by IPCC for Western Europe but are within the Tier 2 calculated range.

Table 5 Enteric emission factors

Reference	Animal category	CH_4 emission rate (kg CH_4 head^{-1} year^{-1})	Notes
IPCC (2006)	Dairy Cows	109	Based on EPA (1994)
IPCC (2006)	Other cattle	57	Based on EPA (1994)
Lassey *et al.* (1997)	Grazing Dairy Cows	96 ± 4	ERUCT, SF_6 tracer, New Zealand, ryegrass and clover, 6.19 +/- 0.15%
Ngwabie *et al.* (2009)	Slatted, Scraped Cow Barn	99 - 114	Sweden, grass and corn silage, Protein pre-mix, naturally ventilated, winter, scraped
Ngwabie *et al.* (2009)	Slatted, Scraped Cow Barn	79	Sweden, grazed during the day, naturally ventilated, summer
Laubach and Kelliher (2004)	Grazing dairy cows	120	Integrated horizontal flux technique, enteric emissions and emissions from deposited excreta combined, NZ
Laubach and Kelliher (2005)	Grazing dairy cows	146	Open path laser method, enteric emissions and emissions from deposited excreta combined, NZ
Snell *et al.* (2003)	Dairy Cows	97-285	no diet information, manure was deposited inside the building with 285 value
Bruinenberg *et al.* (2002)	Grass fed dairy cows	105	96 respiration experiments, grass fed cows, 6% of gross energy
Pinares-Patino *et al.* (2007)	Grazing heifers	73-88	SF_6 tracer, 6-7% of gross energy intake, heifers, France, starting spring
Grainger *et al.* (2007)	Grass fed dairy cows	117	Chamber test, ryegrass fed
Grainger *et al.* (2007)	Grass fed dairy cows	120	SF_6 test, ryegrass fed

2.2.3.2 Emissions from manure management

The term *manure* has been used collectively for dung and urine produced by the livestock while *slurry* is defined as the liquid form of manure produced by addition of waste water to agricultural manure. Emission of methane and nitrous oxide from management of manure has been studied under this section.

2.2.3.2.1 Methane

Methane emissions from manure management tend to be smaller than enteric emissions, with the most substantial emissions associated with confined animal management operations where manure is handled in liquid-based systems (IPCC 2006). The main factors affecting methane emissions include, the volatile solids (VS) content of the manure excreted, the portion of the manure that decomposes anaerobically, temperature, the methane potential of the manure (B_0), and a system specific methane conversion factor (MCF) that reflects the portion of B_0 that is achieved (IPCC, 2006). The volatile solids content of the manure is affected by the species, breed and growth stage of the animals, the feed, the amount and type of bedding material and the degradation processes during pre-storage (IPCC, 2006, Møller *et al.*, 2004a). Lignin in the manure reduces the methane yield while crude proteins increase it (Amon *et al.*, 2007). The volatile solids typically constitute 80% of the total solids which are typically in the range of 7-9% of the fresh weight (Nijaguna, 2002).The MCF will depend on the type of system. Methane will be emitted from slurry and digestate storage tanks, piles of farm yard manure (slurry and straw cleared from cattle bedding), from the application of slurry and digestate to fields and where manure is naturally excreted in the fields, each of these having a different MCF value.

Grazing: Methane production takes place under strictly anaerobic conditions. Holter (1996) measured methane emissions from 1 kg dung pats deposited by grazing cows in Denmark during the summer and found that the emissions were highly variable depending on the temperature and precipitation. Drier conditions led to lesser emissions than wet.

Jarvis *et al.* (1995) performed experiments in the field and laboratories to measure methane emissions from dung pats from cows fed on various diets and under various housing conditions. Jarvis *et al.* (1995) found that high temperatures, whilst stimulating microbial activity and CH_4 production, also stimulate crust formation on the pat. This helps to maintain the anaerobic status of the pat but at the same time changes the CH_4 exchange characteristics between the pat and the atmosphere. Rainfall promotes anaerobic conditions and hence production of methane. Dietary quality of the dairy cow influences the nature of the materials being excreted, especially those volatile solids likely to form potential substrates for CH_4. Jarvis *et al.* (1995) noted that across all dung types, and despite the probable interactions between moisture, temperature and CH_4 generation, there is a strong relationship between C-to-N in the dung and total amounts of CH_4 emitted, i.e. increasing CH_4 with lower C-to-N. Interaction of manure with soil was found to be a minor factor in regulating emissions. Laboratory experiments conducted by Jarvis *et al.* (1995) showed that the dung itself is the primary source of methane emissions. The soil underneath may help in maintaining the degree of anaerobic conditions within the deposited manure.

The emissions from dung pats, at about 1 kg head^{-1} year^{-1}, are a small fraction of the enteric emissions from an animal (approximately 109 kg head^{-1} year^{-1}). IPCC (2006) recommends a methane conversion factor of 1% of the methane producing capacity based on the judgement of the IPCC expert group and Hashimoto and Steed (1994) which is in line with the more recent values presented in literature.

Liquid Slurry Management: Sneath *et al.* (2006) observed that methane is released from the slurry when the concentration of methane in the slurry exceeds its solubility. Bubbles are formed which are then released intermittently either because of a perturbation (feeding, rain, wind) occurring or because the bubbles reach a large size which lead to episodic CH_4 emissions to the atmosphere. The emission factor for stored slurry calculated by extrapolation was 83 kg carbon LU^{-1} year^{-1}. This value is higher than any reported in the literature or calculated using IPCC Tier 2 methodology. Sneath *et al.* (2006) noted that the length of the experiment was too short for the results to be developed into annual emission factors. Covering the slurry resulted in smaller emissions than from uncovered slurry tanks. The emissions from covered tanks were measured using an air injection method while those from uncovered ones used a tracer ratio method. Duration of experiments varied significantly from 2 months for covered and 12 months for uncovered. Hence, the measurement values obtained are not comparable.

Dinuccio *et al.* (2008) conducted storage experiments in the laboratory on slurry and found that about 1 – 1.5% of the volatile solids were lost as methane from an open storage tank over a period of 30 days. Methane emissions were observed to be lower from slurry stored at 5°C than that stored at 25°C for the first 3 days, and vice versa thereafter. This can be explained by the higher moisture loss from the slurry stored at 25°C and the depth of the slurry tank being only 0.2 metres (m). Volumetric reductions of up to 45% were observed. Since the moisture loss was high, the slurry dried out faster and anaerobic conditions were not maintained. Hence, methanogenesis was inhibited by the presence of oxygen in the dried crust and the emission of methane was reduced significantly. This will not be true on field scale where the slurry storage tank will be significantly deeper.

Rodhe *et al.* (2009) conducted a one-year pilot study with conditions similar to full-scale storage with regards to temperature, climate, filling and emptying routines. They measured methane emission rates of 3.6 g kg VS^{-1} in winter and 6.5 g kg VS^{-1} during summer. The average annual methane emission rate was measured as 4.8 g kg VS^{-1}. The annual methane conversion factor, defined as the percentage of methane potential achieved in the system, calculated to be 2.7% which is significantly lower than the 10% suggested by IPCC based on the judgment of the IPCC expert group in combination with Mangino *et al.* (2001) and Sommer *et al.* (2000). The mean annual temperature was 8.1°C which is quite similar to the conditions prevailing in England in winter.

2.2.3.2.2 Nitrous Oxide

Nitrous oxide emissions from grazed cattle are studied under managed soils in the IPCC methodology as it is assumed that no system is in place to manage the manure excreted by the grazing dairy cows and it is directly applied to the soils as organic fertiliser (IPCC, 2006).

The production and direct emission of nitrous oxide from managed manures requires the presence of either nitrites or nitrates in an anaerobic environment, preceded by aerobic conditions necessary for the formation of these oxidised forms of nitrogen. For nitrous oxide emission from manure to occur, nitrification (the oxidation of ammonia nitrogen to nitrate nitrogen) is a necessary prerequisite. The conditions in liquid manure are strictly anaerobic, and hence nitrous oxide is not formed and released. This was noted by Dinuccio *et al.* (2008), Rodhe *et al.* (2009) and Sneath *et al.* (2006) as well as the IPCC Expert Group in combination with Harper *et al.* (2000) and Monteny *et al.* (2001). N_2O production from stored slurries is possible when a dry crust forms on the surface. These emissions occur since the surface crust may contain a mosaic of anaerobic and aerobic micro-sites, which are favourable for N_2O production.

Nitrification is likely to occur in stored animal manures provided there is a sufficient supply of oxygen. Simple forms of organic nitrogen like urea rapidly mineralise to ammonia nitrogen which is highly volatile. Hence, nitrogen is also lost indirectly through volatilisation and run off or leaching of ammonia and nitrous oxide during manure management (IPCC, 2006).

2.2.3.3 Emissions from managed soils

Nitrous oxide is produced naturally in soils through the processes of nitrification and denitrification. Nitrification is the aerobic microbial oxidation of ammonium to nitrate, and denitrification is the anaerobic microbial reduction of nitrate to nitrogen gas (N_2). Nitrous oxide is a gaseous intermediate in the reaction sequence of denitrification and a by-product of nitrification that leaks from microbial cells into the soil and ultimately into the atmosphere. One of the main controlling factors in this reaction is the availability of inorganic N in the soils (IPCC, 2006). The emissions of N_2O that result from anthropogenic N inputs or N mineralisation occur directly from the soils to which the N is added or released. Nitrous oxide is emitted indirectly following volatilisation of NH_3 and NO_x (from managed soils, from fossil fuel or biomass combustion) and the subsequent redeposition of these gases and their products (NH_4^+ and NO_3^-) to soils and waters. Indirect emissions also originate from leaching and runoff of N (mainly as NO_3^-) from managed soils

(IPCC, 2006). These emissions are accounted for under the indirect emissions category by the IPCC methodology.

Direct emissions

The IPCC (2006) methodology estimates nitrous oxide emissions based on human induced net N addition to soils in the form of deposition of manure, spreading of slurry, application of mineral fertilisers, mineralisation of nitrogen in crop residues, or on mineralisation of N in soil organic matter following drainage/management of organic soils, or cultivation/land use change on mineral soils. In the present study organic soils and land use change have not been considered. The emission factor recommended by IPCC (2006) for direct emissions from nitrogen additions from mineral fertilisers, organic amendments and crop residues is 0.01 kg N_2O-N kg^{-1} N based on Bouwman *et al.* (2002b), Bouwman *et al.* (2002a), Stehfest and Bouwman (2006) and Novoa and Tejeda (2006) while that from deposition of excreta by dairy cattle is 0.02 kg N_2O-N kg^{-1} N. Emission of nitrous oxide depends on the rate of excretion of N by the animals and the type of manure management system used.

Soil nitrification is an aerobic process which is dependent on the availability of ammonium and oxygen. Denitrification is an anaerobic process which is controlled by the availability of carbon, oxides of nitrogen and the oxygen supply (Bouwman *et al.*, 2002b).

Nitrous oxide emissions from managed soils are much higher in autumn and winter than during summer (Allen *et al.*, 1996). Increased emissions induced by freezing and thawing events account for a substantial part of the annual emissions in colder countries. Emissions from poorly drained soils are higher than well drained soils because of better maintenance of anaerobic conditions (Senbayram *et al.*, 2009). Flynn *et al.* (2005) analysed the variation of nitrous oxide emission factors with change in rainfall and temperature and suggested incorporation of a climate variable to the emission factors using annual rainfall and temperature data as derived by Dobbie *et al.* (1999).

Flessa *et al.* (2002) noted that highly significant linear relationship existed between the annual N_2O emission and total N input. These results agree with those summarised by Bouwman *et al.* (2002b) who found that annual N_2O

emission from cultivated soils was decisively influenced by N supply. It varied from 1.6 kg N_2O-N ha^{-1} for 1-50 kg N ha^{-1} applied to 6.8 kg N_2O-N ha^{-1} for > 250 kg N applied ha^{-1}. This was, however, for a subset of the data. Bouwman *et al.* (2002b) synthesised 846 measurements of nitrous oxide emissions from 126 different sites and came up with an emission factor of 1.25 ± 1% of applied N. Kaiser and Ruser (2000) observed that 0.7 – 2.86 % of applied N as slurry was emitted as N_2O. Ellis *et al.* (1998) compared nitrous oxide emissions from fertiliser application, surface slurry application and slurry injection and measured 2.1 %, 3.8% and 3.4% emissions respectively. Flessa *et al.* (2002) measured the emissions from grazing at 3.2% per kg N excreted. The default IPCC emission factor is 2%.

Indirect emissions

The indirect emissions of nitrous oxide from volatilisation and subsequent deposition are calculated by estimating the total amount of nitrogen applied to the soil in the form of slurry, manure deposited by grazing animals and synthetic fertilisers, the fraction of it that volatilises and the emission factor. While calculating the emissions from leaching, the nitrogen that is applied to the soil as well as that which is mineralised is taken into consideration, along with the fraction that leaches and a corresponding emission factor (IPCC, 2006).

The main input parameters for nitrous oxide emissions are the amount of nitrogen excreted and the emission factors for direct and indirect emissions. The IPCC (2006) has recommended an emission factor of 0.01 kg N_2O-N kg^{-1} NH_3-N + NO_x-N volatilised for indirect nitrous oxide emissions from nitrogen volatilisation and redeposition and 0.0075 kg N_2O-N kg^{-1} nitrogen from leaching and runoff. The emission factors recommended by (IPCC, 2006) have largely been accepted in the literature as very little data is available.

2.2.3.4 Emissions from manufacture of mineral fertilisers

Swaminathan and Sukalac (2004) have reported that the production of fertilisers accounts for 1.2% of the total energy consumed in the world and is responsible for about the same proportion of GHG emissions. Ammonia (NH_3), potash (elemental potassium K) and phosphorus (phosphate, P_2O_5) are used in the production of crops including grass and winter wheat. While the use of a blend is most common, assumption of use of straights (which supply only one major plant nutrient) results in a more accurate nutrient, energy, emissions and cost calculations. In the UK, the manufacture of mineral fertiliser releases 7.05 kg CO_2 eq., 1.72 kg CO_2 eq. and 1.68 kg CO_2 eq. per kg of N, potash and triple superphosphate fertiliser manufactured, respectively (Mortimer *et al.*, 2007).

2.2.3.5 Emissions from usage of fuel

Farm machinery, like harvesters and tractors, runs on diesel fuel. The usage of fuel depends on the size of farm, the type of farm equipment used, the number of cuts of grass, the number of fertiliser applications, etc. The emission factor for the manufacture and use of diesel in the UK is 0.3 kg CO_2 eq. kWh^{-1} (DECC, 2012a).

2.2.3.6 Emissions from electricity and heat consumption

The energy supply industry, which primarily based on natural gas (47%) and coal (28%), is responsible for 35% of all GHG emissions in the UK (DECC, 2010a). The GHG emission factor for use of electricity produced by major power stations in the UK after accounting for the losses incurred during its transmission and distribution has been used. The Digest of UK Energy Statistics (DUKES) reports emission factors of 0.58982 kg CO_2 eq. kWh^{-1} of electricity consumed in the UK and 0.25892 kg CO_2 eq. kWh^{-1} of heat from liquefied petroleum gas (LPG) (DECC, 2012a).

2.3 Anaerobic digestion

2.3.1 Overview

Anaerobic Digestion (AD) is a process where biomass is broken down by micro-organisms in the absence of air. In controlled anaerobic digestion, biomass is put inside a sealed tank (anaerobic digester) and naturally occurring micro-organisms digest it, releasing biogas. The breakdown of organic compounds is achieved by a combination of many types of bacteria and archaea. Anaerobic digestion takes place at two optimum temperature ranges, 35-40˚C (mesophilic) and 55-60˚C (thermophilic) (Defra, 2011b). The biomass added to the digester is broken down into sugars, amino acids and fatty acids, then fermented to produce volatile fatty acids and finally methanogens produce biogas, comprising of CH_4 (53-70%), CO_2 (30-47%) and other trace gases including nitrogen (N_2), hydrogen sulphide (H_2S), ammonia (NH_3) and chlorine (Cl_2) (Persson *et al.*, 2006). Figure 2 shows an example of implementation of an anaerobic digester on a dairy farm.

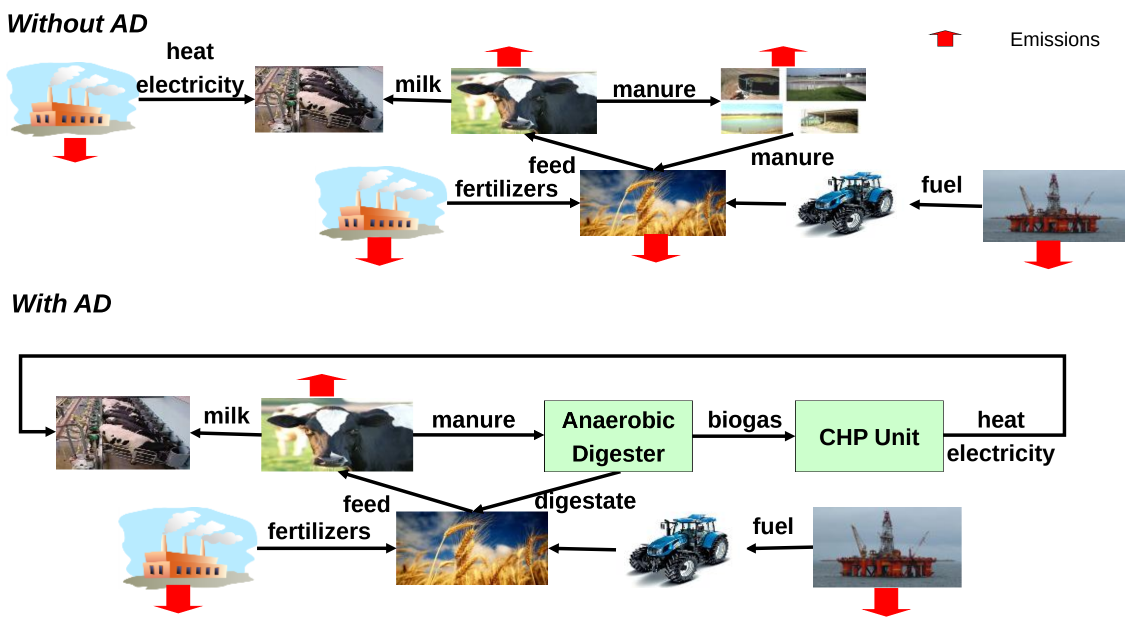


Figure 2 Example of implementation of a digester on a dairy farm

2.3.2 Benefits of anaerobic digestion

The products and benefits of anaerobic digestion of slurry have been discussed below.

2.3.2.1 Production of biogas

The biogas that is produced from anaerobic digestion of biomass can be combusted to provide heat and electricity via a CHP or burned for heat. Alternatively, it can be upgraded to bio-methane to be used as a transport fuel or supplied through the gas grid as a replacement for natural gas, reducing dependence on fossil fuels.

The amount of biogas produced can be determined by calculation based on the amount of volatile solids available for digestion. Organic materials contain a number of organically digestible volatile solids (VS) which can be anaerobically broken down to produce biogas. These materials have an 'ultimate' methane value which can be achieved under optimal conditions of temperature, nutrients, digestion time, etc. The potential methane yield is therefore, often expressed as a specific methane yield (B_0) relating to a particular type of material under a fixed set of conditions. Examples of specific methane yields of cattle slurry and the conditions under which they were achieved are shown in Table 6 (for ease of comparison, values presented have been converted to a standard m^3 CH_4 kg^{-1} VS).

Table 6 Examples of specific methane yield of cattle slurry

References	Specific methane yield (m^3 CH_4 kg^{-1} VS)	Notes
IPCC (2006)	0.24	Generalised value for ultimate methane yield.
Rodhe *et al.* (2009)	0.294	100 days, 37 °C, inoculum from energy crop and municipal solid waste biogas plant, no cow diet information made available, use of blank not clear.
Kaparaju (2003)	0.13-0.16	122 days, 35 ± 1 °C with inoculum
	0.07-0.08	122 days, 35 ± 1 °C without inoculum
Amon *et al.* (2007)		60 days, 38 °C, varied ratios of concentrates, hay, grass silage and maize silage as listed below
	0.1365	0: 5.2: 10.4: 0
	0.1318	0: 5.4: 6.4: 5.8
	0.1663	4.6: 4.0: 4.8: 5.2
	0.1431	5.8: 5.0: 10.0: 0
	0.1255	11.0: 3.2: 3.8: 3.6
	0.1592	10.0: 3.0: 6.2: 0
Frost and Gilkinson (2011)	0.16	Field data.
Møller *et al.* (2004b)	0.148 ± 0.041 m^3 kg^{-1} VS added	Average from varied feed, 100 days, 35 ± 0.5 °C

IPCC (2006) recommends a value of 0.24 m^3 CH_4 kg^{-1} VS excreted for maximum methane yield which is based on the volatile solids in the manure rather than a measured one. Rodhe *et al.* (2009) measured an average methane yield of 0.294 m^3 CH_4 kg^{-1} VS added in 100 days of incubation at 37°C in laboratory experiments. These experiments, however, used inoculum from a production-scale digester digesting energy crops and municipal solid waste and it is not clear if a blank was used or not during the experiments. The impact of using inoculum to assist digestion was studied by Kaparaju (2003) who found that the methane yields doubled from 0.07-0.08 m^3 kg^{-1} VS added to 0.13-0.16 m^3 kg^{-1} VS added under similar operating conditions.

Amon *et al.* (2007) studied the impact of diet on the methane producing potential of the manure. By changing the proportion of concentrates, hay, grass silage and maize silage, a variation of 0.125–0.166 ml CH_4 kg^{-1} VS added was observed in the methane yield. Inoculum was used and the methane produced from the inoculum was subtracted to obtain the reported results. The operating temperature was 38 ˚C, the total solids content 9% and retention time 60 days. The maximum specific methane yield was measured from cows that were fed a balanced diet. This variation was also reported by Møller *et al.* (2004b). The cows were fed different combinations of roughage (maize and clover-grass silage), hay, concentrates, barley and minerals. Møller *et al.* (2004b) observed that the methane yield varied between 0.1 and 0.207 m^3 CH_4 kg^{-1} VS added. The experiments were conducted at 35 ± 0.5 ºC for 100 days

Thus, the values reported by Amon *et al.* (2007) and Møller *et al.* (2004b) are actual measurements and exhibit a similar range while those reported by Rodhe *et al.* (2009) are much higher.

2.3.2.2 Production of digestate

Digestate is the left over indigestible material and micro-organisms. It contains valuable plant nutrients like nitrogen, phosphate and potassium. It can be used as a fertiliser and soil conditioner (SAC, 2007). By providing low carbon fertilisers for agriculture, AD can help deliver a sustainable farming sector, where resources are reused on-farm to reduce GHGs and provide secure and sustainable inputs, particularly phosphate (Defra, 2011d).

2.3.2.3 Odour reduction

Processing of livestock manures in an anaerobic digester can significantly reduce the odour (Powers *et al.*, 1999, Zhang *et al.*, 2000). Odour reduction of up to 50% can be achieved (Powers *et al.*, 1999). In an experimental set of anaerobic digesters, H_2S and mercapatans were reduced to negligible a concentration with little residual odour. This verifies the beneficial effect of anaerobic digestion on odour reduction from animal manure (Zhang *et al.*, 2000).

2.3.2.3.1 Pathogen removal

Presence of pathogens in untreated slurry applied to land poses a biosecurity risk. The concentration of pathogens may be reduced by anaerobic digestion of slurry. Gadre *et al.* (1986) found that incubation of 10 days at 37 ºC resulted in inactivation of all *Salmonella* in cattle slurry. The decay rate of bacteria during digestion depends on many factors including temperature, retention time, pH, volatile fatty acids, type of digestion, bacterial species and available nutrients (Sahlström, 2003).

2.3.2.3.2 Nutrient recycling

The energy intensive production of nitrogen and the mining of phosphate from non-renewable sources can be reduced by the use of digestate as fertiliser (SAC, 2007), thus replacing manufactured and mined fertiliser. A large part of the mineral and trace elements that are fed to the cow is excreted out with the manure and while only a small proportion is absorbed. These, however, become more available to the plants if the slurry is digested, and then to the cows which are fed these plants (Bywater, 2011). Hence, the absorption of mineral and trace elements is increased indirectly.

2.3.2.4 Other on-farm benefits

Additional benefits of digesting slurry based on the experience of farmers who have deployed digesters at their farms, have been reported by Bywater (2011) and are listed below.

- Ease of spreading due to lower viscosity. Addition of water is not required for mixing, pumping or spreading.
- Faster re-grazing. Cows can be grazed after 2-3 weeks of application, thus increasing the spreading window for land application.
- Quick integration into the soil. Digestate being less viscous does not taint the following crop or interfere with crop production equipment.

- Quality of produce. Farmers have noticed better quality grass and garden crops after the application of digestate.
- Reduced Biological Oxygen Demand (BOD). Digestate has a lower BOD than undigested slurry, making it less damaging to watercourses.
- Encourages nitrogen fixing clover in the leys.
- Kills weed seed to decrease herbicide application.
- Farm income diversification (Bywater, 2011).

2.3.3 Benefits of AD over other technologies

The advantages of AD over other renewable energy technologies:

- Depending on the infrastructure available, the biogas produced may be used in the form of electricity, heat or bio-methane.
- Energy is generated continuously and can be stored in the grid in the form of gas.
- Bio-methane is suitable for heavy goods vehicles (HGVs) and has the potential to reduce reliance on imported gas.
- AD facilities can be swiftly constructed.
- Relatively inexpensive when compared to other renewable energy technologies.
- Inputs/outputs and scale are flexible i.e. plants can be designed according to the feedstock available locally and maybe modified while being connected to the grid.
- Low carbon fertilisers are provided for agriculture.
- Helps in making the farming sector more sustainable by reusing resources within the farm to reduce GHGs, provide renewable energy and sustainable agricultural inputs, particularly phosphate (Defra, 2011b).

2.3.4 Current status of AD in the UK

As of Feb 2013, there were a total of 104 anaerobic digesters operating in the UK with an additional 146 in the water industry. Agricultural products or by-products (slurries, manures, crop or crop residues) were used by 40 as feedstock while 46 were community digesters digesting predominantly food waste collected from multiple sources. There were 18 industrial digesters treating on-site waste like brewery effluent or food processing residues. Only 3

AD plants (Didcot Sewage Works, the Adnams Brewery and Rainbarrow farms) were upgrading biogas to bio-methane and injecting it into the gas grid (Defra, 2013). The low uptake of anaerobic digestion may be attributed to the fact that the farmers perceive the establishment costs to be too high and the returns too low (Tranter *et al.*, 2011). There is also a perceived difficulty in obtaining planning permission and a lack of information available on AD. The same barriers were also seen in the survey conducted by Mbzibain *et al.* (2013).

2.3.5 Typical impurities in biogas

The typical impurities present in biogas that need to be removed in order to use it as vehicular fuel or inject it into the gas grid to be used in place of natural gas have been listed below. The harmful effects of these impurities and the technologies available for their removal have also been discussed.

2.3.5.1 Carbon dioxide

Carbon dioxide is a major proportion of biogas (30-47% by volume) (Persson *et al.*, 2006). Removal of CO_2 is necessary for consistent gas quality and higher heat value required for vehicles or grid injection. The technologies available for carbon dioxide removal are:

1. Absorption
    a. Water scrubbing
    b. Organic solvents
2. Pressure Swing Adsorption (PSA) which removes carbon dioxide by its adsorption on activated carbon or molecular sieves.
3. Membrane separation
4. Cryogenic separation
5. In-situ methane enrichment

2.3.5.2 Sulphur gases

Hydrogen sulphide is the primary form of sulphur present in biogas along with other sulphur gases (disulphides, thiols). It is highly reactive in the presence of water and elevated temperatures and hence has to be removed in order to avoid corrosion of compressors, gas storage tanks and engines (Persson *et al.*, 2006).

Hydrogen sulphide can be removed from the biogas using any of the following methods:

1. Biological desulphurisation
2. Iron chloride dosing to digester slurry
3. Impregnated activated carbon
4. Iron hydroxide or oxide
5. Sodium hydroxide scrubbing

2.3.5.3 Water

Biogas is saturated with water when it leaves the digestion chamber. This may condense in the pipelines and, along with oxides of sulphur, cause corrosion. It is necessary to remove it before the biogas is burnt which may be done by:

- Refrigeration
- Adsorption of water on the surface of a drying agent like silica gel or aluminium oxide
- Regeneration at elevated or atmospheric pressure
- Absorption of water in glycol or hygroscopic salts (Persson *et al.*, 2006).

Of the 137 upgrading facilities operating in the Europe, water scrubbing (35%), PSA (30%) and use of chemical absorption (23%) are most commonly used (Persson *et al.*, 2006). The remaining 12% comprise of physical absorption, membrane and cryogenic separation (Persson *et al.*, 2006).

2.3.6 Use of upgraded gas

Upgraded biogas may be injected into the gas grid or used as vehicular fuel.

As per the UK Gas Safety (Management) Regulations (1996), bio-methane must meet the requirements listed in Table 7 for gas grid injection.

Table 7 Gas quality requirements for gas grid injection

Content or characteristic	Value
Hydrogen sulphide (H_2S)	≤ 5 mg m^{-3}
Total sulphur (including H_2S)	≤ 50 mg m^{-3}
Hydrogen (H_2)	≤ 0.1% (molar)
Oxygen (O_2)	≤ 0.2% (molar)
Wobbe Number (calorific value divided by the square root of the relative density	47.2 – 51.41 MJ m^{-3}
Odour	Gas below 7 bar will have a stenching agent added to give a distinctive odour
Impurities and water and hydrocarbon dew points	The gas shall not contain solids or liquids that may interfere with the integrity or operation of the network or appliances.
Incomplete combustion Factor (ICF)	≤0.48
Soot Index (SI)	≤ 0.60

Upgraded biogas offers the flexibility of immediate use, storage or pipeline transport. The overall efficiency of energy capture is higher as energy loss related to transmission is reduced when compared to electricity via CHP unit. Upgraded biogas can be produced in remote locations without the worry of utilisation of heat. On the other hand, there are additional chemical, water or other waste streams that require additional treatment. Also, upgrading technology is currently more expensive that CHPs. Hence, in the UK, most AD plants employ CHP units to use the biogas produced rather than upgrading it.

2.3.7 Emissions from AD

There are sources of emissions associated with the employment of AD on a dairy farm. These may be from the storage of digestate, application of digestate to the field or as fugitive emissions from the digester and other equipment. These sources of emissions have been discussed in detail in the following sections.

2.3.7.1 Storage of digestate

After being digested in an anaerobic digester, the slurry is held in storage tanks until it can be applied to the field. In general, and especially in nitrogen vulnerable zones (NVZ), the storage tanks are emptied out in early spring, i.e. March or April as application of organic fertilisers is best during these months and also because its application is prohibited during the winter months. Hence, the digestate is accumulated in storage tanks for a few months over the winter. Storage tanks may be uncovered, covered with straw or wooden lids or may be gas tight containers connected to the anaerobic digester. Depending on the microbial activity, residual potential of the digestate, the type of storage tank, the climate and the duration of storage, greenhouse gases maybe produced and emitted or collected during this period.

The extent of digestion of slurry varies from digester to digester depending on the hydraulic retention time, the temperature and the composition and age of slurry. If the hydraulic retention time of the slurry in the digester is insufficient, digestion will continue during storage and emissions of methane will be observed. On the other hand, if most of the volatile solids have been converted into methane and captured during digestion, the methane produced during storage will be minimal.

Temperature also has an effect on the time required to complete digestion of the slurry. Based on OFMSW (Organic Fraction of Municipal Solid Waste), Hansen *et al.* (2006) derived an exponential relationship between the production rate for methane and the storage temperature of slurry.

$$E_{CH4} = 0.0004 * e^{0.159t}$$

Where E_{CH4} is the rate of methane production, $Nm^3\ CH_4\ Mg^{-1}\ VS\ h^{-1}$

t is the temperature of digested waste within the interval 5 to 35 ºC

A clear linear relationship between the log-transformed estimated methane production and the inverse of temperature in Kelvin has also been reported by Khan *et al.* (1997) as shown in Figure 3. A linear relationship between the slurry temperature and the air temperature was also reported.

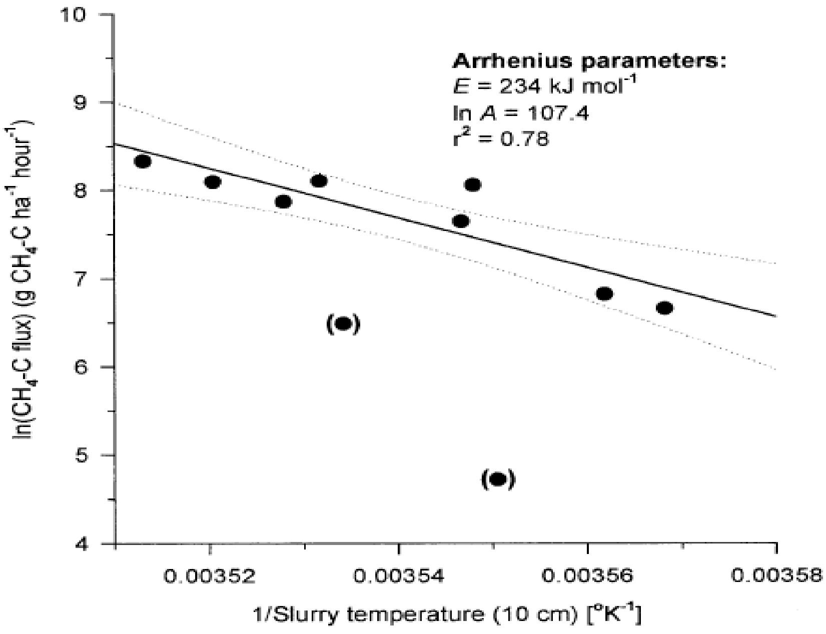


Figure 3 Relationship between methane production and temperature (Khan *et al.*, 1997)

According to Davidsson *et al.* (2007), during anaerobic digestion of municipal solid waste in biogas plants, 70-80% of the organic matter is typically degraded, leaving 20-30% that may be degraded in the storage tank where the digestate is kept for months before it can be applied to the land as fertiliser. Ploechl *et al.* (2009) presented the emissions from a storage tank as a function of the percentage of methane potential achieved in the digester and the methane potential achieved in the storage tank.

Clemens *et al.* (2006) observed that digestion of slurry reduced methane emissions and that increasing retention time from 29 to 56 days further reduced the storage methane emissions. Clemens *et al.* (2006) also observed a reduction in methane emission by covering the storage tank in both winter and summer. This observation is counter-intuitive and has not been adequately explained in the paper. The cover will help maintain the anaerobic conditions in the storage tank and prevent the formation of crust. This should lead to an increase in the methane production as in a digester rather than reducing it.

Börjesson and Berglund (2006) and Paavola and Rintala (2008) measured methane emissions from stored digested slurry and found these in the range of 5-15% of the total biogas production. Umetsu *et al.* (2005) observed higher methane emissions from stored undigested slurry than digested slurry. This relationship was also observed by Amon *et al.* (2006) and Clemens *et al.* (2006). This can be attributed to the fact that there is more biodegradable carbon available in undigested slurry than digested. The biodegradable carbon is captured as methane during the process of digestion.

Nitrous oxide is produced as an intermediate during nitrification and denitrification and the presence of oxygen is essential for its production. Due to the anaerobic conditions in a digestate storage tank, the nitrous oxide emissions are very small and occur only if a crust forms on the surface. Clemens *et al.* (2006) observed no nitrous oxide emissions from stored slurry during laboratory experiments. During the field study, however, nitrous oxide emissions were observed in all experiments. The origin of these emissions is not clearly explained. Nitrous oxide emissions may be completely eliminated using gas tight storage tank for digestate, however, as there is no headspace oxygen available for conversion of ammoniacal nitrogen to nitrous oxide.

Similarly, even though methane may be produced in a gas tight storage tank the overall environmental impact can be reduced by capturing and using it to produce heat and electricity via CHP.

2.3.7.2 Emissions from field application of digestate

The emission of greenhouse gases from application of digestate to land depends on various factors such as the condition of the digestate, the time and method of application, the soil and the climate. Some of these factors and their effect on emissions are discussed below.

Methane:

There are very short methane emission events immediately after application of slurry and digestate (Dittert *et al.*, 2009). Two sources of methane emissions from field application identified by Wulf *et al.* (2002) are:

- Emissions immediately following application attributed to the release of dissolved methane produced during storage.
- Anaerobic conditions promoted by the injection of digestate.

The method of application of digestate plays an important role in determining the extent of emissions. Injection of digestate leads to higher methane emissions compared to splash plate, trail hose and trail shoe methods of application (Wulf *et al.*, 2002). A splash plate spreader forces the slurry through a nozzle under pressure onto a splash plate for surface application of slurry to the land. In trail hose application of slurry, the boom of the spreader has a number of hoses connected to it and the slurry is distributed close to the surface of the land in bands. The flow of slurry is under pressure differential. Both these methods are used for surficial application of slurry and do not encourage anaerobic conditions. The trail shoe spreader is similar to the trail hose except for a shoe added to the end of each hose that allows the slurry to be deposited under the crop canopy. Injectors place the slurry under the surface of the soil. They may be open slot shallow injectors, injecting the slurry at depth of 50 millimetre (mm) or deep injectors placing it over 150 mm deep. Injection of slurry promotes anaerobic conditions as air contact is reduced and oxygen is depleted due to mineralisation of organic matter. This leads to emissions of methane as long as carbon is available and the anaerobic conditions are maintained (Wulf *et al.*, 2002).

Assuming the use of a trail hose method of digestate application, digested slurry emits less methane than raw slurry. This observation was made by both Clemens *et al.* (2006) and Wulf *et al.* (2002). The band of raw slurry does not disperse as fast as the less viscous fermented substrate, preserves its moisture and thus retains dissolved CH_4, and conserves anaerobic conditions over a longer period which leads to further emission of methane as long as carbon is available. Also, fermented slurry has less available organic carbon for the production of methane as most of the organic carbon is converted into biogas during digestion.

Nitrous Oxide:

- Soil Moisture: The effect of fertiliser type and N application rates on N_2O emission have been found to be significant when the soil moisture content was high (85% of water holding capacity) by Senbayram *et al.* (2009). Nitrous oxide emissions are closely related to the soil moisture content as the main factors driving the denitrification process are redox potential, substrate and oxygen diffusion which strongly depend on water availability and the water-air filled pore-space in soil. When the soil moisture is high, denitrification dominates and N_2O emissions are higher as well (Senbayram *et al.*, 2009). This relationship was also observed by Akiyama *et al.* (2004).

- Method of application: Injection of digestate leads to higher nitrous oxide emissions than from splash plate, trail hose or trail shoe (or harrowed) (Wulf *et al.*, 2002). Injection of slurry promotes anaerobic conditions as air contact is reduced and oxygen is depleted due to mineralisation of organic matter. This encourages the process of denitrification.
- Nitrogen application rate: The total N_2O emissions were similar when comparing unfertilised control soil and soil that received 45 kg N per hectare as digestate. The total emitted N_2O increased sharply when the digestate N rates were raised from 45 to 90 kg N per hectare, and emissions increased linearly with N rates from 90 to 360 kg N per hectare (Senbayram *et al.*, 2009).
- Soil type: Direct nitrous oxide emissions from slurry applied to loamy soil were observed to be at least 3 times higher than from sandy soil. Some of this difference can be attributed to the higher leaching loss from sandy soil and may be compensated by indirect emissions (Dittert *et al.*, 2009). Velthof *et al.* (1998) observed that nitrous oxide emissions were higher from clayey soils than sandy and explained it by a combination of factors like availability of carbon, which controls the potential for denitrification, and the aeration status of the soil.
- Rainfall pattern: The rainfall pattern (continuous irrigation, partial drying and heavy rainfall and periodic heavy rainfall) affects the temporal production of CO_2 and N_2O, but not the cumulative emissions as long as the soil does not completely dry out (Sanger *et al.*, 2010). Independently of the rainfall pattern, all digestate amended soils showed a nitrate leaching peak approximately 5 weeks after its application (Sanger *et al.*, 2010).
- Crop yield: Digestion of slurry has minimal effects on the overall dry matter yields and the nitrogen utilisation efficiency of the crop rotation in comparison with undigested slurry for most crops (Moller *et al.*, 2008). This may be because of the higher ammonia losses after spreading digested slurry, as the increased ammonia concentration and higher pH of the digested slurry promote gaseous nitrogen losses. Also,

if the undigested slurry is incorporated immediately after addition to soil, the organically bound nitrogen of the undigested slurry seems to have enough time in long cycle crops e.g. maize, to become partially mineralised and available to crops. Crops with a short and intensive period of nitrogen uptake like spring wheat, however, may achieve significantly higher grain yields with the help of the more available nitrogen in the digested slurry (Moller *et al.*, 2008).

- Type of vegetation: Wulf *et al.* (2002) observed that the N_2O emissions from trail hose application of co-fermented slurry (digested slurry produced through combined fermentation of 70% dairy cow slurry and 30% organic household waste) to grass land were much higher than from undigested slurry. The exact opposite was observed from application to arable land. Higher emissions from unfermented slurry applied to arable land may be due to the reduction of carbon pools during fermentation. In grasslands, the soil dissolved organic carbon is higher and hence, carbon availability becomes less limiting for denitrification. The ruling factor on grasslands is the contact of slurries with the soil. The co-fermented slurry being less viscous may pass through the soil layers and hence induce soil microbial processes faster than unfermented slurry if applied by trail hose application (Wulf *et al.*, 2002). For spreading co-fermented slurry on grassland, trail shoe application seemed to be the best way of minimising trace gas emissions. On arable land, trail hose application with immediate harrowing is recommended (Wulf *et al.*, 2002).

No significant difference between digested and undigested slurry nitrous oxide emissions over a period of one year has been found (Clemens *et al.*, 2006). Detailed results from this study have been presented in Table 8.

Table 8 GHG emissions after field application of digested slurry, adapted from Clemens *et al.* (2006)

	N_2O emission (1 year) (mg N_2O-N m^{-2})	NH_3 emission (4 days) (mg NH_3-N m^{-2})	CH_4 emission (4 days) (mg CH_4-C m^{-2})	CO_2 equivalents (g CO_2 m^{-2} $year^{-1}$)	CO_2 equivalents (kg CO_2 m^{-3} $year^{-1}$)
Control	28.0 (12.38)		0.8 (0.83)	13.7	
CAN	58.6 (28.16)	298 (344)	1.9 (1.37)	30.0	
CS-0	40.7 (11.30)	711 (475)	27.1 (6.97)	24.0	4.2
CS-29	42.7 (16.32)	797 (889)	16.1 (3.28)	25.1	5.9
MIX-29	41.6 (10.52)	1385 (761)	15.2 (4.16)	27.4	8.1
MIX-56	29.5 (12.33)	768 (334)	20.6 (2.79)	18.6	5.1

2.3.7.3 Fugitive Emissions

The release of the biogas produced by the anaerobic digester may be controlled or uncontrolled (termed fugitive emissions). Flaring of the gas or use in a CHP unit can be considered as controlled, while fugitive emissions are defined here as uncontrolled emissions of biogas due to leaks and various other unintended or irregular releases from digestion or equipment or CHP unit. The extent of fugitive emissions from a digester depends on the quality of construction and its management and operation. Liebetrau *et al.* (2010) identified CHP unit and digestate storage units, when constructed without a sealed cover, as the two main sources of fugitive emissions. A portable flame ionisation detector was used to detect methane sources at 10 agricultural biogas plants in Germany. Emissions from CHP averaged at 1.73% of the methane converted. 0.27% of converted methane was emitted while mixing feed with digestate. Fugitive emissions measured by Flesch *et al.* (2011) averaged at 3.1% of the methane produced under normal operating conditions. An inverse-dispersion technique was used to measure and calculate the total emissions over 4 seasons to account for seasonal variability. Silsoe Research Institute (2000) conducted experiments on fugitive emissions from two anaerobic digesters and estimated these at 3.5% and 2.4% of the biogas produced.

2.3.7.4 Embodied Carbon in AD

Embodied carbon of a building material can be defined as the total carbon released over its life cycle. This includes extraction of raw materials, manufacturing and transport (Hammond and Jones, 2008). Embodied carbon may be calculated over cradle-to-grave (production till final disposal), cradle-to-gate (production only) or cradle-to-site (production and transportation to site where the material is used) depending on the data available. Cradle-to-gate has been found to be the most common boundary condition and has been used in the Inventory of Carbon and Energy (Version 1.6) (Hammond and Jones, 2008). Farm buildings including anaerobic digesters are constructed using concrete, steel, insulation and wood. It has been estimated that concrete has 0.13 kg CO_2 embodied kg^{-1}, steel 1.77 kg embodied CO_2 kg^{-1} and insulation has 1.86 kg embodied CO_2 kg^{-1} of material (typical consumption mix of insulation materials in the UK).

2.3.8 Economics of Anaerobic Digestion

Economics related to the introduction of anaerobic digestion to a dairy farm has been reviewed in the following sections. The literature has been reviewed for the capital cost of the digester, CHP, upgrading unit and the associated operating costs.

2.3.8.1 Capital Cost of Anaerobic Digester

An anaerobic digestion plant digesting slurry consists of

- Feeding technology including mixing pit, pumps and feeder
- Digester equipment which includes concrete/steel digester, mixer, heating circuit, sensors, cover, gas storage
- Post digestion storage tank for digestate, gas storage
- Technology plant that houses the electrical, gas equipment
- CHP unit complete with the engine, measuring and controlling technology and a container module (Kottner *et al.*, 2008).

Besides the cost of equipment required for digestion, the capital cost includes the cost of plant design, grid connection, planning and approval cost and earth works. Unforeseen costs or a contingency of 5% of the capital cost is also factored in (Kottner *et al.*, 2008). Macerators, pre- or post- pasteuriser and additional permits and earthworks may be required if crop residues or food waste are accepted as feedstock for the digester (Kottner *et al.*, 2008). The configuration of the AD plant may vary based on the feedstock, level of automation and the infrastructure available.

2.3.8.1.1 Estimates available for the UK

Estimates of the cost of anaerobic digestion plants have been presented in Table 9.

Table 9 Estimated range of capital cost of anaerobic digesters in the UK

Reference	Cost range
Environmental Resolve (1997)	£3,000 - £7,000 kW^{-1}
Redman (2010)	£2,500 - £6,000 kW^{-1} £400 - £750 m^{-3}

Environmental Resolve (1997) brought together the industry, environmentalists, planners and government agencies to establish capital cost range and Good Practice Guidelines for AD. The Redman (2010) estimates for initial capital investment exclude connection to the grid, earthworks, pasteurisation, etc. More recent estimates of digestion cost in the UK are presented in Table 10.

Table 10 Recent AD capital cost estimates (£'000 MW^{-1}) for the UK (DECC, 2011)

Plant capacity	<1MW	1 to 6 MW
High	6,985	6,260
Median	4,463	4,000
Low	2,396	2,147
Feedstock	**Food Waste**	**Farm Waste**
High	6,915	6,711
Median	5,241	3,906
Low	3,740	1,673

Literature was reviewed for information available on existing slurry based digesters in the UK and their capital cost since only broad guidelines are available from the industry. All inclusive capital costs of digesters in the UK that are primarily digesting animal slurries have been reported by Bywater (2011). These, along with the quotes invited from various vendors for setting up anaerobic digesters on existing farms in Cornwall (Kottner *et al.*, 2008), have been compiled and presented in Section 5.1.

2.3.8.1.2 Methods of calculation of capital costs

In order to establish a way to quantify the upfront investment required for setting up anaerobic digesters in the UK, literature was reviewed for various methods of calculation currently in use.

Capital cost per dairy cow

The AgStar program of the United States Environmental Protection Agency (US EPA) deals with the anaerobic digestion of agricultural wastes. The guidelines published by US EPA have been presented in Table 11.

Table 11Capital cost estimates used in the USA (AgSTAR, 2009)

Digester type	Capital Cost ($)	Capital cost per dairy cow ($ dairy cow^{-1})
Complete mix	563 * number of dairy cows + 320864	7881 * (dairy cows)$^{-0.3152}$
Plug Flow	617 * number of dairy cows + 566006	13308 * (dairy cows)$^{-0.3493}$
Covered Lagoon	400 * number of cows + 599556	68516 * (dairy cows)$^{-0.6074}$

It may be noted that these are valid only for herd sizes greater than 500 cows which are not common in the UK where the average size is 145 cows as reported in Section 2.1.2. Also, capital costs based on number of cows are not flexible enough to incorporate changes in digester feedstock, for example addition of crop residues.

Scale up factor

Karellas *et al.* (2010) and Zglobisz *et al.* (2010) used a scale up factor approach for estimating the capital cost. A model digester with a known capacity and a known capital cost was chosen. The capital cost of larger digesters was then estimated based on a scaled up treatment capacity using the formula:

CAPEX of Actual Biogas Plant/ CAPEX of Model Biogas Plant = (Treatment capacity of Actual Biogas Plant/treatment capacity of Model Biogas Plant)^(Scale up factor)

Where CAPEX stands for capital expenditure

The scale up factor chosen for digester costing was 0.6 and that of CHP was 0.8 by Karellas *et al.* (2010) while Zglobisz *et al.* (2010) assumed 0.7 for the digester. Scale up factor is a standard method used for estimating the cost of equipment in the chemical engineering industry. The choice of scale up factor for anaerobic digestion equipment used, however, is based on a different industry. The lack of standardisation of equipment and various levels of automation in AD plants makes this generic assumption invalid. This approach is useful when the data available is scarce or if the digester planned is similar to one already existing.

Per kW of installed capacity

Redman (2010), DECC (2011) and Environmental Resolve (1997) have presented estimates of the range of capital cost of anaerobic digester plants in the UK based on the installed energy capacity as discussed in Section 2.3.8.1.1.

Capital cost estimates based on installed capacity, however, do not account for an alternative use of biogas such as upgrading of biogas for use as vehicular fuel and would require additional information for the derivation of digester capital cost.

Per m^3 of digester capacity

Murphy and Power (2009) developed a power relationship between the quantity of waste treated and the capital cost of digester, biogas upgrading and compression facilities based on literature. The relationship is, however, based on dry digestion technology (DRANCO) applied in Ireland and hence not directly applicable to this study.

2.3.8.2 Capital Cost of CHP

A Combined Heat and Power (CHP) unit burns biogas in a combustion chamber producing a flow of hot air. This hot air is used by the generator to produce electricity while the exhaust heat can be pumped through insulated pipes to provide space and water heating for local buildings (Kottner *et al.*, 2008). CHP is a relatively developed technology and its capital costs depend on the type of engine and size. Kottner *et al.* (2008) invited quotes from multiple suppliers for the identified farms in the Cornwall region. The quotes included capital cost of engine, generator, measuring and control technologies and have been presented in Section 5.2. The installed cost of a CHP plant varies between £550 to £1,050 kW^{-1} of electrical output with economies of scale working in favour of larger units (DECC, 2012c).

2.3.8.3 Capital cost of upgrading biogas

The capital cost of upgrading biogas depends on the technology used, the extent to which it needs to be cleaned and the size of the upgrading unit. There is a lack of UK based costing data available for upgrading as there are only 3 operating units (Defra, 2013). Hence, literature was searched for capital cost information available from other countries. Data was compiled from the actual plants in the EU and quotes presented by Persson and Hogskola (2003). Small scale upgrading plants have been successfully implemented for the purpose of grid injection or use as vehicular fuel in EU (Finland, Austria, Sweden, Germany and Hungary) and India (Kaparaju *et al.*, 2012). Figures for capital cost obtained from suppliers have been presented in Section 5.3.

2.3.8.4 Operating cost

The operating cost of a digester includes its maintenance and repair, the labour required to run it, insurance payments, expenditure on buying feedstock if necessary, etc. Kottner *et al.* (2008) estimated the insurance cost at 1.5% of the total capital cost and the costs of maintenance and repair of 'construction' and 'technology' at 2% and 3% of their capital costs respectively. Actual data on the running cost of a digester are sparse. The estimates compiled by Bywater (2011) show that these may vary between 2 and 11% as presented in Table 12. The operating cost varies with the type and level of automation of the digester, the skill set of the farmer and the input feedstock. The expenditure on fulfilling the electricity and heating requirement of the digester is in addition to these operating expenditures.

Table 12 Running cost of digesters operating in the UK (Bywater, 2011)

Farm	Capital cost (£ year^{-1})	Operating cost (£ year^{-1})	Percentage of capital cost (%)	Notes
Hill Farm	£50,000	£1,000	2	Maintained by the farmer
Shropshire Farm	£45,000	£1,900	4.2	Maintenance cost
Walford and North Shropshire college digester	£135,000	£3,600	2.6	
Bank Farm	£105,000	£3,000	2.8	Includes CHP maintenance
Copys green farm	£750,000	£83,000	11	Feedstock expenditure included
Kemble Farm	£1,200,000	£33,000	2.75	

2.3.9 Existing economic models

A number of economic models have been developed for both farms and anaerobic digestion. Some of these have been listed below and their relevance to the project has been discussed.

FarmWare 3.6 (AgSTAR, 2010) software has been developed by AgSTAR, a part of the United States Environmental Protection Agency (USEPA), to assess whether or not an anaerobic digester can be integrated into an existing farm or planned manure management system. FarmWare estimates the cost and the financial benefits that may be gained by producing energy on farm for use and/or for sale. The incentives provided by the government and, the cost and selling prices of electricity are different in the USA. Hence, the results from this model are not applicable in the UK and may be used for reference purposes only. Additionally, it is a purely economic model.

The **NNFCCC biogas calculator** (Redman, 2010) is a part of a biogas toolbox designed to assist an AD developer in assessing the costs and revenues related to AD. The model takes into account the capital and operating costs, capital depreciation, FIT/ROCs, gate fees, fertiliser value of slurry and digestate to calculate the profitability of the enterprise, the internal rate of return and the return on capital. The model focuses on the economics of anaerobic digestion and the sourcing of feedstock is not taken into account. The model recommends a range for expected capital cost of the digester and the parasitic loads. All the costing information is an input to the model rather than it being information given by the model. This model may be used for the purpose of evaluating the digester on its own but not as a part of a bigger farming system.

Jones (2010) developed an economic model as a linear programming simulation model run on the GAMS modelling platform. It is an activity based model which maximises the net economic margin for specific farms. It evaluates arable and dairy farms for higher food and commodity prices, import of feedstock and compares it to reference runs. The model assumes a loan for 10 years at 4% rate of interest. It assumes capital cost of £1.5 million for an AD plant of 500 kilowatt (kW) installed capacity. It also assumes that electricity generated is exported while that required to run the digester plant comes from the mains supply. The model is set up for very specific scenarios. Also GAMS not being freely available to the public makes the model difficult to evaluate.

Moorepark dairy system model developed by Shalloo *et al.* (2004) is a stochastic budgetary simulation model of an Irish dairy farm. It studies the effect of varying biological, technical and physical processes on farm profitability. It does not study the GHG emissions from farms and is hence not applicable.

The existing models are focussed on one aspect of farming or anaerobic digestion, be it emissions, economic, energy or chemical/biological analysis of farms. Few of those available are relevant for dairy farms based in the UK. None of the existing models, discussed above, are designed to assess the different dairy management practices, GHG emissions from dairy farming, anaerobic digestion and economics related to the same which are adaptable to the conditions prevailing in the UK. There is also an absence of work that brings together the economic and environmental aspects of AD that may be used to evaluate the impact of policy on the farming industry. This research will fill this gap by combining these various aspects and develop a tool that may be used in policy evaluation.

2.3.10 Current financial incentives

The UK government has recognised anaerobic digestion as a well proven renewable energy and waste management technology. It has committed itself to making the most of the potential of anaerobic digestion to contribute to the climate change, waste management and wider environmental objectives of the government (Defra, 2011b). This occurs through a number of incentives.

2.3.10.1 Feed in Tariffs

Feed-in Tariffs (FITs) were introduced in April 2010, as a part of the clean energy cash back scheme in which payments are made to ordinary energy users for the renewable energy they generate (DECC, 2012d). Feed-in Tariffs provide a guaranteed price for a fixed period to small scale renewable electricity producers. They are intended to support all renewable electricity generation sources of less than 5 MW to various degrees depending on the technology used (DECC, 2012d). The feed-in tariffs are index linked, i.e. they are adjusted pro-rata to the retail price index. Reduction of support is planned as renewable energy technologies become cheaper. Feed-in tariffs have been designed by the government to give a 5-8% rate of return to the investor (DECC, 2012d).

Eligibility: Renewable energy technologies with less than 5 Megawatt (MW) installed capacity are eligible for a feed-in tariff which varies with the technology. Systems installed before 15th July 2009 and registered under renewables obligation before 31st March 2010 are eligible for a base tariff of 9 pence kWh^{-1}. AD facilities of less than 5 MW completed after 15 July 2009 are eligible for the FIT (DECC, 2012d).

Generation Tariff: Generation tariff is paid for every kilowatt hour (kWh) of electricity produced. For the year to 31st March 2012, anaerobic digestion facilities of less than or equal to 250 kW are entitled to 14.7 pence kWh^{-1} of electricity generated and facilities of greater than 250 kW and less than 500 kW to 13.6 pence kWh^{-1}. Facilities of greater than 500 kW installed capacity are entitled to 9.9 pence kWh^{-1}. These tariffs are valid for a period of 20 years. Payments are made for the electricity generated irrespective of whether it is used on-site or exported to the grid. These payments are made by the energy supplier of the generating property.

Export Tariff: Export tariff is a bonus payment made for every kWh of surplus energy generated that is exported to the electricity grid. This tariff is the same for all renewable energy generation technologies. The floor price for the year from 1st April 2011 has been set at 3.2 p per kWh. Like the generation tariff, this price is index linked to the retail price index. Generators have the freedom to choose this floor price or negotiate their own selling price with the electricity supplier for the year.

Digression: As the volume of renewable technologies builds up, digression of tariffs has been planned. The digression may be triggered by any of the 3 mechanisms listed below:

- Pre-planned digression – this is simple regular percentage reduction in tariff offered to new facilities. It stands at 10% every 6 months for solar PV and 5% every year for all other technologies.
- Contingent digression – this is a deployment based digression. For every technology overall installed capacities have been decided. When these are reached, the tariff offered to new facilities is reduced after a 2-3

month notice period. In order to guarantee the tariff that will be offered to a particular facility, a system of preliminary accreditation has been introduced.
- Annual tariff reviews – the government may review tariff on an annual basis to ensure that desired outcomes are being achieved.

Advantages:

- The feed in tariffs are independent of the market, offer guaranteed payments for the lifetime of the facility and hence offer security to the renewable energy producer and encourage investment.
- FITs encourage non-traditional investors like small scale investors and community groups.
- FITs encourage different scales of energy producers to try out new technologies.

Disadvantages:

- The funds are limited and the subsidy is passed onto the taxpayers.
- FITs focus primarily on the production of energy rather than the reduction of carbon footprint.

As on 01/01/2013, 1655.43 MW of capacity from 358,295 installations were claiming FITs. Of these, 1.8% of the energy produced was from anaerobic digestion facilities while photovoltaic accounted for 90% (OFGEM, 2013).

FIT is a commonly implemented renewable energy policy. As of early 2012, 65 countries had enacted feed in policies all over the world including the USA, Germany, India and Australia (REN21, 2012).

2.3.10.2 The Renewable Heat Incentive (RHI)

Heating accounts for 47% of the UK's carbon dioxide emissions and 60% of average domestic energy bills (Parliamentary Office of Science and Technology, 2010). Approximately 69% of heat is produced from gas while oil and electricity account for 11% and 14% respectively. Solid fuel is used to produce 3% of the heat produced in the UK and renewables just 1% (Parliamentary Office of Science and Technology, 2010). The Renewable Heat Incentive (RHI) has been set up under the Energy Act 2008. The RHI provides financial assistance to generators of renewable heat, and producers of renewable biogas and bio-methane. In order to be injected into the grid, the biogas needs to be cleaned of impurities, dried and upgraded to higher methane content (95%) so that it resembles the qualities of natural gas. The RHI went live in November 2011 and unlike the feed in tariff, is funded by the Treasury (DECC, 2012b).

Eligibility: Eligible technologies include biomass boilers, biogas combustion, deep geothermal, ground source heat pumps, energy from biomass proportion of municipal solid waste, solar thermal (up to 200 kW_{th}) and water source heat pumps that have been built after 15th July 2009. RHI can be claimed for eligible uses of heat which may be determined using the following guideline:

- The heat load should be an existing or new heat requirement.
- The heat must be supplied to meet an economically justifiable heating requirement.
- Acceptable heat uses are space, water or process heating where the heat is used in fully enclosed structures.

Tariff: For biogas on-site combustion (up to 200 kW_{th}) and injection of bio-methane (all scales) into the grid, the RHI has been set at 7.1 p per kWh with a tariff lifetime of 20 years (DECC, 2012b).

Heat transmission is accompanied by heat losses ranging from 3.5% to 20% depending on the transmission distance (Poeschl *et al.*, 2010). Seasonal variation in the demand for heat is quite significant. Biogas from a digester is available all through the year and once upgraded to bio-methane and injected into the grid has minimal losses in transmission. Upgrading of biogas and injection of propane is expensive but is already being used in Germany, France, Austria and the USA. In Germany, heat generation by biogas plants corresponds to 3-4% of the heat generated from renewable energies (Poeschl *et al.*, 2010).

2.3.10.3 Renewables Obligation Certificates

As per the EU Renewable Energy Directive, the UK is required to supply 15% of total energy demand from renewables by 2020. The Renewables Obligation places an obligation on suppliers of electricity in the UK to source an increasing proportion of their electricity from renewable sources (DECC, 2012h). The obligation for the year 2011-12 was set at 12.4% of the supply.

This policy is aimed at supporting and encouraging large scale (>5 MW) renewable energy projects in the UK. The government intends that suppliers will be subject to a renewables obligation until 31 March 2037.

Mechanism: The renewables obligation has been implemented using the following mechanism:

Renewables Obligation Certificate: A Renewables Obligation Certificate (ROC) is a 'green' certificate issued to an accredited generator for eligible renewable electricity generated within the United Kingdom and supplied to customers within the United Kingdom by a licensed electricity supplier. Different technologies receive different levels of support or ROCs MWh^{-1} depending on their costs and potential of large-scale deployment (DECC, 2012h). Anaerobic digestion is among the technologies that receive additional support in the form of multiple ROCs. Anaerobic digestion can receive 2 ROCs MWh^{-1}. To get accreditation for the RO by the Office of Gas and Electricity Markets (OFGEM), an AD plant needs to pass OFGEM's test of reasonableness and use an approved electricity meter (DECC, 2012h).

Buy-out Fee: In case of failure to meet this obligation, the supplier is required to pay a buy-out fee. The proceeds of this fee are redistributed amongst suppliers who have produced the required amount of ROCs in a particular period. The buy-out price for the compliance year 2012-2013 has been set at £40.71 per ROC (DECC, 2012h).

ROC market: ROCs are issued to renewable energy generators but sold to energy suppliers who are obliged to meet their renewable energy targets. The ROCs are sold in addition to the electricity, thus creating two income streams for the renewable energy generators. The price of the ROCs is determined by demand and supply and has varied from £39.52 and 51.24 ROC^{-1} over the past 2 years (October 2010-2012) (E-ROC, 2012).

Advantages:

- The price of ROCs is market dependent and hence, ROCs offer the potential of high profits but with the market related risk.
- The ROCs market is more effective for large scale energy producers which have diversity in their investments/sources.

Disadvantages:

- They encourage maximum production of energy and not maximum mitigation of GHG emissions.

- It is perfectly legal for a supplier to source all its electricity from non-renewable sources of energy and buy the required ROCs from the market to fulfil its obligation. Even though the market price of ROC, £41.33 as of 20 December 2012 (E-ROC, 2012), is higher than the buy-out price, having the ROCs entitles the supplier to the buy-out fund for the compliance year.
- The cost of ROCs is passed on to the consumers via higher energy prices.

2.3.10.4 Renewable Transport Fuel Obligation Certificates

The Renewable Transport Fuel Obligation (RTFO) requires suppliers of fossil fuels to ensure that at least 5% of the road fuels they supply in the UK are made up of renewable fuels (DoT, 2012). Renewable Transport Fuel Certificates are awarded per litre of biofuel or per kilogram of bio-methane supplied, provided that it is dutiable and meets the sustainability criteria. Biofuels derived from waste (including bio-methane from anaerobic digestion of cattle slurry) are eligible for double RTFCs. There is no guaranteed price for RTFCs. The value is determined completely by market forces (DoT, 2012).

The biofuel or the feedstock used to produce biofuel may be produced within the UK or imported as long as it meets the sustainability criteria.

Upgrading of biogas to vehicle fuel is still not a very common practice in the UK due to the high costs of upgrading as discussed in Section.

2.3.10.5 Current grants for AD in the UK

A number of schemes have been made available to provide incentives for individuals and organisations to adopt anaerobic digestion. These schemes provide financial and technological support to the interested parties in setting up AD.

2.3.10.5.1 Rural Development Programme for England (RDPE)

Anaerobic digestion, along with a range of other measures, is eligible for support under the Rural Development Programme for England 2007-2013

(RDPE). RDPE is investing in the capacity of England's bio-energy supply chains to ensure that they are able to compete to meet the rising demands for bio-energy products, particularly biomass. RDPE invests to purchase or develop equipment for collaborative use, provide support and training to enhance competitiveness and raise standards across the supply chain (Defra, 2012).

2.3.10.5.2 WRAP Anaerobic Digestion Loan Fund (ADLF)

ADLF is a £10 million fund designed to support the development of new food waste based AD capacity in England. The fund aims to support 300,000 tonnes of annual capacity to divert food waste from landfill by 2015. The fund provides asset backed loans for plant, machinery and/or ground works. The minimum loan is £50,000 and maximum £1,000,000 with a maximum term of 5 years (WRAP, 2012).

2.3.10.5.3 Enhanced Capital Allowance

The Enhanced Capital Allowance Energy Scheme provides businesses with enhanced tax relief for investments in equipment that meets published energy-saving criteria. The equipment must be specified in the Energy Technology List (ETL) which is managed by the Carbon Trust on behalf of the UK Government (Carbon Trust, 2012a). This provides a cash flow boost and an incentive to invest in energy-saving equipment, which normally carries a price premium when compared to less efficient alternatives. The Combined Heat and Power Unit (CHP) is listed on the ETL. However a certification on good working condition of the CHP is required in order to qualify for the allowance (Carbon Trust, 2012a).

2.3.10.5.4 Carbon Trust Loan

The Carbon Trust provides 0% interest loans to businesses investing in energy-saving equipment. The loan amount varies from £3,000 to £100,000. The loan amount is dependent on the size of the overall investment and the CO_2 savings of the project (Carbon Trust, 2012b).

A number of schemes and grants are available to fund anaerobic digestion projects. These offer some support to those willing to use anaerobic digestion but given the current low uptake of the technology. It is clear that they do not incentivise the dairy farmers to build and run anaerobic digesters. The need for a more effective policy, that rewards mitigation and penalises emission, is thus highlighted.

2.3.11 UK and EU policy and regulations

2.3.11.1 Climate Change Act 2008

The Climate Change Act 2008 is a legally binding target to reduce the UK's GHG emissions by 34% by 2020 and at least 80% by 2050 compared to 1990 levels (DECC, 2012g). Carbon budgets were introduced to meet these targets. A carbon budget is a cap on the total quantity of GHG emissions emitted in the UK over a specified time. The first 3 carbon budgets were set in law in May 2009.

- The first carbon budget (2008-2012) requires 23% emissions reduction below 1990 level.
- The second carbon budget (2013-2017) requires a 29% reduction in GHG emissions.
- The third carbon budget (2018-2022) takes this reduction requirement to 35% (DECC, 2012g).

The Carbon Plan published in December 2011 outlines the steps the government will take to achieve these targets, and the contribution of each sector towards it. Sectorial plans include low carbon buildings, improving residential insulation and energy efficiency, energy labelling of appliances, deployment of low carbon heating, more efficient combustion engines in vehicles, use of sustainable biofuels, capping aviation emissions, better design of industrial processes, replacement of fossil fuels with low carbon alternatives, carbon capture and storage, low carbon electricity and diverting waste from landfills (DECC, 2012g).

Current Status: Emissions have fallen by a quarter since 1990 (HM Government, 2011).

2.3.11.2 EU Renewable Energy Directive

Under the EU Renewable Energy Directive (2009/28/EC) UK is required to source:

- 15% of its energy from renewable sources by 2020,

- 10% of energy used in transport from renewable sources by 2020

Current Status: Renewables sources accounted for 8.7% of the electricity and 2.2% of heating and cooling generated in the UK in 2011. Additionally, 2.9% of the transport energy was from renewable sources in 2011, putting the overall renewable consumption as a percentage of capped gross final energy consumption using net calorific values at 3.8% (DECC, 2012f).

2.3.11.3 AD strategy and action plan

The AD strategy and action plan is a joint Government and industry publication and emphasises the government's commitment to achieving a zero waste economy by encouraging waste management and waste to energy technologies including anaerobic digestion (Defra, 2011b).

This plan establishes that digested manure/slurry as not a waste if the digestate is used as fertiliser. This applies to both solid and liquid digestate.

Key features of the Action Plan are:

Knowledge and Understanding

- Establish baseline of AD activity in the UK.
- Training provisions for technical competence and also individual needs.
- Development of knowledge regarding beneficial use of digestate.

Smarter Working Models

- Improve understanding of the economic, environmental and social aspects of all models of AD.
- Technological and best fit solutions to be examined for all types of AD projects.
- Acknowledging the limited use of bio-methane.
- Improve understanding and knowledge of the operation of AD on farm.

Regulation and Finance

- Identify regulatory issues that could pose obstacles to the adoption and operation of AD facilities.

- Improve understanding of the current regulatory process for obtaining permits for AD.
- Simplify the protocols governing injection into the gas grid and connection to the electricity grid for small capacity plants.
- Build investor confidence by reducing the risks and costs associated with providing finance.
- Provide guidance to developers to obtain finance necessary to bring forward projects of all types and scales.

2.3.11.4 Anaerobic Digestate – Quality Protocol

The Quality Protocol for anaerobic digestate (WRAP 2010) specifies the end-of-waste criteria for digestate or when the digestate will normally be regarded as having ceased to be waste and therefore no longer subject to waste management controls. The criteria are listed below.

- It has been produced using non-waste biodegradable materials, source-segregated input materials specified in the protocol or animal by-products transformed under Article 15 of the EU ABPR and UK legislation making provision for the administration and enforcement of ABPR.
- It meets the requirements of an approved standard i.e. BSI PAS 110: 2008.
- It is destined for appropriate use in one of the designated market sectors
    - Agriculture, forestry and soil/field-grown horticulture; and
    - Land restoration.

2.4 Cost of Carbon

With the increasing focus on global warming and climate change, various different methods of putting a cost on carbon have been developed. These have been outlined below and the relative advantages and disadvantages discussed.

2.4.1 Cap and Trade – Market Price

Cap and Trade is a market based policy which imposes a limit on the total allowable emissions from all sources in accordance with the emission targets and the desired environmental effect. Authorisations to emit in the form of emission allowances are then allocated to the affected emission sources. This policy allows the emission source the flexibility to comply with the limits by either adopting lower carbon technologies, or by buying in allowances from the market (US EPA, 2009). In many cap and trade systems, organisations which do not pollute may also participate. Thus environmental groups can purchase and retire allowances and hence drive up the price of the remainder credits in the market by reducing supply.

There are currently six exchanges trading carbon allowances: the Chicago Climate Exchange, European Climate Exchange, NASDAQ OMX Commodities Europe, PowerNext, Commodity Exchange Bratislava and the European Energy Exchange.

European Union Emissions Trading Scheme (EU ETS)

The European Union Emissions Trading Scheme (EU ETS) was set up in 2005 in order to meet the EU's GHG emissions reduction targets established under the Kyoto protocol (European Union, 2012). Member States develop a National Action Plan (NAP), approved by the European Commission, capping the total amount of emissions allowed from all installations covered by the scheme, e.g. iron and steel, electricity generation, mineral processing industries, etc. The installations are required to monitor and report their emissions according to the allowances distributed by the Member State. Surplus or deficit allowances can be sold or bought amongst participating installations to meet their respective targets. Thus the market price of carbon is determined. Agriculture is currently not covered under the EU ETS. The EU ETS makes sure that the allocations of member countries are in line with the Kyoto Protocol (European Union, 2012).

The first phase of EU ETS was considered a failure due to the over allocation of permits. This resulted in a near zero value of carbon credits. The second phase of EU ETS is currently on going (2008 – 2012). The third phase of EU ETS will not have any national allocation plans (NAP). The allocation will be determined at the EU level (European Union, 2012). Carbon was trading at €3 tonne^{-1} in January 2013 which much lower than abatement costs (McGrath, 2013).

Advantages:

- It encourages cheap abatement.
- It provides environmental benefit without affecting economic growth.
- Innovation, efficiency and early action are rewarded.
- Incentives are provided for doing better and consequences for doing worse.

Disadvantages:

- This method favours cheap abatement methodologies across all sectors and there is little incentive for industries whose abatement costs are more than the market price of carbon, to reduce emissions.
- The carbon price varies day to day and it is difficult for any policy decisions to be based on it.
- There is a higher risk on investment due to the volatility in the price especially exposing smaller businesses.
- Transaction costs – these are costs that originate from the exchange rather than the production of goods or services. They may have three potential sources:
    - Finding a buyer or a seller.
    - Bargaining and finalising deals – insurance, legal fee, time and fee for brokerage.
    - Monitoring emissions and enforcing limits (Stavins, 1995).
- Carbon Leakage – The effect that a regulation of emissions in one country has on the emissions in other countries that are not subjected to the same regulations is referred to as carbon leakage. Cap and trade emissions may lead to higher emissions outside of capping area.

2.4.2 Social Cost of Carbon (SCC)

The Social Cost of Carbon is the marginal damage cost associated with an incremental emission of GHG, summed over its lifetime and discounted back to the year of the emission (DECC 2009b). In other words, the social cost of carbon measures the full global cost today of an incremental unit of carbon (or equivalent amount of other greenhouse gases) emitted now, summing the full global cost of the damage it imposes over the whole of its time in the atmosphere. SCC signals what society should be, in theory, willing to pay now to avoid the future damage caused by incremental carbon emissions. The SCC depends on the stabilisation trajectory that a country is following which may or may not be the same as the rest of the world (Defra, 2007). The Stern Review (Stern, 2005) found that the value of the SCC depends on the current atmospheric concentrations when that tonne of GHG is released.

Advantage:

1. The Stern review may be used in ensuring that the targets for emission reduction and atmospheric GHG concentration are set at the right level.
2. This pricing relies on modelling climate damages from integrated assessment modelling.

Disadvantage:

1. Valuation of damage that climate change will create in the long term is highly uncertain.
2. Climate change impacts are non-linear. There is a concave relationship between emissions and increase in temperature i.e. additional emissions produce decreasing impact on the temperature. There is a convex relationship between damages and temperature i.e. damages increase more than proportionately with temperature.

2.4.3 Shadow Price of Carbon (SPC)

While SCC is determined purely by our understanding of the damage caused and the way it is valued, the shadow price of carbon (SPC) can be adjusted to reflect the policy and technological environment. The shadow price of carbon is based on the social cost of carbon for a given stabilisation goal. SPC takes more account of uncertainty and is based on a stabilised trajectory. The SPC is dependent on the year the carbon is abated/emitted (Defra, 2008b).

The Stern Review (Stern, 2005) calculated the social cost of carbon at \$30 tonne^{-1} CO_2 eq. in 2000, equivalent to £19 tonne^{-1}CO_2 eq. This is the number that has been recommended by Defra as the basis of a shadow price of carbon profile for use in policy and investment appraisals across government in the

UK. Using uprating conventions, Defra adopted an SPC in 2007 of £25 tonne^{-1} CO_2 eq. It is based on a stabilisation concentration of 550 ppm CO_2eq.

The SPC has the same advantages and disadvantages as the SCC. Both SCC and SPC give a direction to the global policy based on climate change and its impact. This is, however, a theoretical price based on damages caused by climate change and hence has a lot of uncertainty and assumptions associated with it.

2.4.4 External cost of the human activities

In economics, an externality refers to situations when the effect of production or consumption of goods and services imposes costs or benefits on others which are not reflected in the prices charged for the goods or services being provided (Khemani and Shapiro, 1993). The impacts of greenhouse emissions from various industrial and agricultural activities are borne by society in the form of environmental damage and health costs. The government, society or third parties bear these costs that are not recovered from the emitters of these gases or accounted for in the pricing of their products.

To put the damage in economic terms, in 1996, the annual external environmental and health costs of UK agriculture were estimated to be £2343 million (range for 1990-1996: £1149-£3907 million) (Pretty *et al.*, 2000), equivalent to £208 ha^{-1} of arable and permanent pasture. This accounts for only those externalities that give rise to financial costs and is likely to underestimate the total negative impact of agriculture (Pretty *et al.*, 2000). The total measurable damages due to air emissions in the UK in accounting year 2007 were estimated at about £2000 million (Jacobs and SAC, 2008).

When such externalities are not included in prices, they distort the market by encouraging activities that are costly to the society even if the private benefits are substantial. Internalisation of these costs, through taxation or incentives, can help discourage pollution by making the polluter pay for the negative impact on the environment and at the same time incentivise sustainable behaviour and policy. For example, an agricultural system that uses excess fertiliser not only pollutes the nearby surface and groundwater resources, but also affects plants and soil of neighbouring farms and countries by deposition of ammonia. At the same time one that fixes nitrogen by planting leguminous plants, not only negates the need for fertilisers thus preventing emissions but also improves the soil health and quality on the farm.

This method of accounting, however, has a lot of uncertainty associated with it in terms of the damage caused to air, water and ecosystem especially due to wide range in cost depending on the timeframe considered. Also it tries to put a value on intangibles like the value of fresh air, taking a walk in the park or observing diverse wildlife, etc. Even though a lot of research has been done on the impacts of pollution, the real, long term, all-inclusive impact is not known and hence cannot be valued.

2.4.5 Marginal Abatement Cost (MAC)

Marginal Abatement Cost (MAC) is the cost of mitigating emissions by one tonne CO_{2eq} rather than the damage imposed by emissions and is hence a more proactive approach to incentivising the solution of the problem of global climate change. The advantage of this approach is that it is objective and target consistent. It is a bottom up approach which can help the UK in achieving the mitigation targets set and relies on in-depth abatement cost modelling. MAC can be used to evaluate the relative feasibility of abatement technologies across industries as well as those within the sector.

In a major shift in carbon valuation policy, in July 2009, DECC moved away from the social cost of carbon and the shadow price of carbon based on the Stern review, to the cost of mitigating emissions (DECC, 2009b). For evaluating policies related to emissions not covered by EU ETS (the ‘non-traded sector’), a non-traded price of carbon will be used, based on the marginal abatement cost required to meet a specific emissions reduction target. A short term non-traded price of carbon has been set at £60 per tonne of CO_2 equivalent until 2020 with a range of +/- 50% (DECC, 2009b). Where a policy delivers mitigation at a cost lower than the non-traded price of carbon, it will be considered to be cost effective.

Marginal abatement cost curves are a standard tool to illustrate the economics of abatement initiatives aimed at reducing emissions of pollutants. The costs of abatement measures and their potential contribution towards meeting an abatement target are evaluated on the basis of a base year. The marginal abatement cost curves developed by McKinsey and Company for the UK for CBI concluded that 90-95% of the abatement measures required to reach the 2020 targets will cost less that €60-€90 tonne^{-1} CO_2 eq. (using 2002 as the base year). Anaerobic digestion was not evaluated in the study (Confederation of British Industry, 2007).

Only one study, conducted by Moran *et al.* (2008), has been done on the evaluation of abatement measures available for the livestock industry in the UK. Moran *et al.* (2008) proposed a MAC of £26 tonne^{-1} CO_2 eq. using on-farm anaerobic digestion for medium sized dairy farms in the UK. This MAC is, however, based on capital cost estimates from FEC services (2003) which does not take mortgage payments into consideration and assumes an annual running cost of 2% of capital cost (which is lower than the current estimates). These assumptions have led to underestimation of the marginal abatement cost. Moran *et al.* (2008) categorised livestock mitigation options into animal and manure management and evaluated their abatement potential and cost effectiveness as presented in Table 13.

Table 13 Cost effectiveness of livestock mitigation options

Measure	Cost Effectiveness (£ 2006 tonne^{-1} CO_2 eq.)
Ionophores	-50
Maize silage	-270
Improved productivity	-0.07
Improved fertility	-0.04
On-farm AD – Large dairy farm	14
On-farm AD – Medium dairy farm	26
Bovine Somatotropin (bST)	230
Transgenics	1,740

Marginal abatement cost can also be used to compare the currently available low carbon technologies. Table 14 summarises the MAC estimates available for other renewable energy technologies in the UK.

Table 14 MAC of other renewable energy technologies (Committee on Climate Change, 2008)

Technology	MAC (£ tonne^{-1} CO_2 eq. abated)
On-shore wind	£55-133
Off-shore wind	£85-152, £153 (£71-£243) (Vivid Economics in association with McKinsey & Co., 2011)
Marine	£193

The valuation by carbon market is very volatile, its future is uncertain and hence it is not suitable for use in this study. The social cost of carbon, shadow price of carbon and external cost of carbon are all based on valuation of global damage which is difficult to ascertain and has a number of uncertainties associated with it. Marginal abatement cost being the most objective and bottom up approach has been chosen as the carbon valuation study by the UK government and for this study.

Having established the existing knowledge related to the digestion of dairy cow slurry, the methods used in the project are outlined. The following chapters develop the emissions and economic models that will be used to derive information that can produce MACs for various farming scenarios including the introduction of AD. The issues of accurately determining the MAC required for incentivising farmers to take up AD on dairy farms, reducing energy requirements and emissions from both energy generation and dairy production will be addressed. Elements that are most critical to the financial and environmental feasibility of anaerobic digestion will be identified.

3. Farm model

The farm model lays the foundation for the emission and economic models and analysis of the results obtained. This model calculates the intermediate variables required for obtaining the emissions from a farm and the profit made. The variables include herd size for a given size of farm, the allocation of land within the farm for various farming activities, the amount of manure collected and available for digestion, and the sizing of digester and CHP unit required.

3.1 Herd size

A dairy herd comprises of dairy cows and followers (heifers, dry cows, breeding bulls). For a given size of farm, the herd size is calculated based on the livestock density and ratio of dairy cows to followers. In this study, the ratio of dairy cows to followers is assumed to be 0.9 based on McHoul *et al.* (2012). For a given size of farm, the livestock density, including dairy cows and followers, is limited by the organic nitrogen application regulations for NVZs (Defra, 2009). Based on McHoul *et al.* (2012) and Defra (2009), the livestock density is assumed to be 1.6 livestock units (LU) ha^{-1} assuming a cow to be 1 LU while a follower is 0.6 LU.

3.2 Land Allocation

Given the total size of the farm and the calculated herd size, the farm land is allocated to different uses in order to meet the requirements of the cattle. Assuming that no feed is imported, grass silage is grown to be fed to the cattle when they are housed and pasture maintained for when they are outdoors, grazing. In order to maintain their milk yield, the dairy cows are fed winter wheat grown on farm as concentrate. The farm land is allocated for each one of these requirements based on the nutritional requirements of the cattle as follows:

1. The total concentrate requirement of the dairy cows is determined using the average milk yield of a dairy cow in the UK based on Defra (2011a),

7,406 litres year^{-1} and the concentrates required to achieve that milk production, 1.9 tonnes year^{-1} (Nix, 2007).

2. The total area of land required to fulfil the winter wheat requirement is calculated based on an average yield of 8.5 tonnes ha^{-1}.
3. The total metabolisable energy in grass silage and pasture is calculated based on the dry matter content of the crops and the metabolisable energy per hectare using values given in Table 15.

Table 15 Yield and metabolisable energy in crops

	Yield (t FM ha^{-1})	Dry matter content (g kg^{-1})	Metabolisable energy (MJ kg DM^{-1})
Grass silage	45	250	11
Pasture	35	180	12

4. The net energy requirement of the cattle (MJ day^{-1}) is calculated based on the methodology outlined in Section 4.1.
5. The area of grass silage and pasture (ha) required is calculated based on the metabolisable energy available and the net energy requirement of the cattle.
6. The total land required is calculated by summing up the areas required for winter wheat, grass silage and pasture.
7. If the total required is less than the total farmland available, then the ratio of the winter wheat, grass silage and pastures are taken and the farmland available is divided in the same ratio.
8. If the total required is more than the farmland available, then this can be corrected by reducing the livestock density of the farm.
   A screenshot of the module is presented in Figure 4.

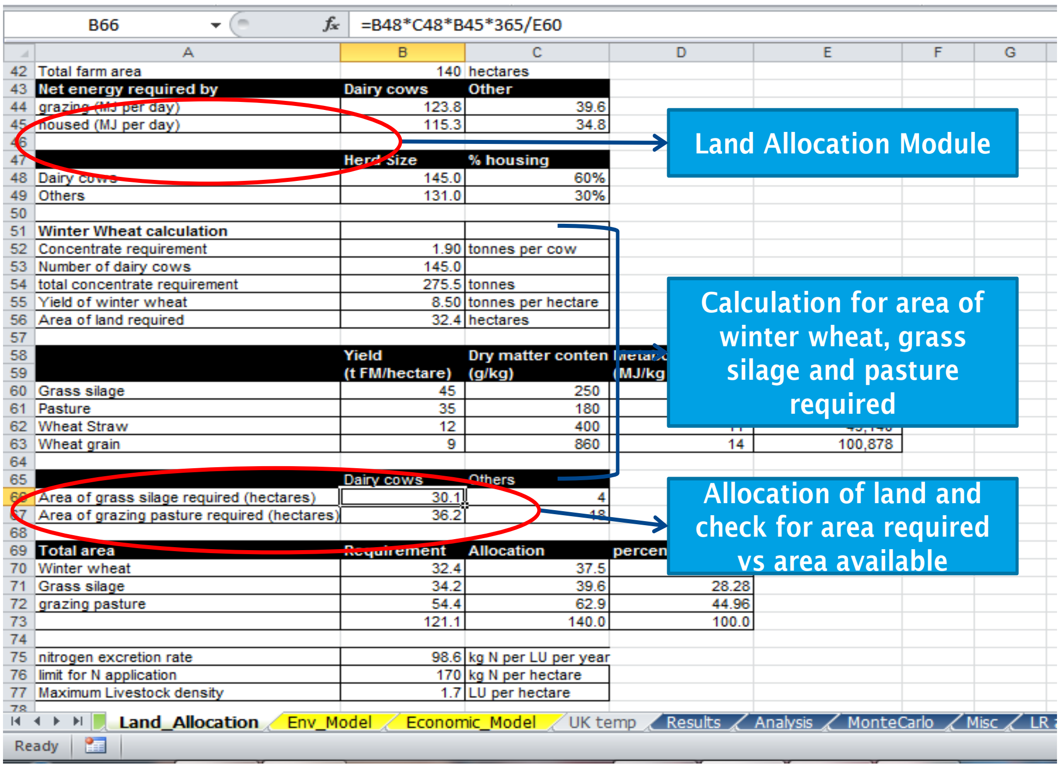


Figure 4 Land allocation module

3.3 Manure management

Based on the herd size, the total amount of manure excreted by the cows and followers is calculated using a manure excretion rate of 19.3 and 14.6 tonnes year^{-1} for dairy cows ($Excretion_{dairycow}$) and other cattle ($Excretion_{follower}$), respectively (Defra, 2010b).

It is assumed that on a farm without a digester, the manure excreted by the cattle during housing is collected and stored in a slurry tank and then spread on the field. The manure excreted by the cattle during grazing is allowed to lie as is assuming a uniform distribution across the grazed fields.

When a digester is operating on the farm, the manure collected from the housed dairy cows and followers is collected and fed to the digester. The digestate is stored in a post-digestion storage tank and then applied to the fields using the same machinery as that used for spreading raw slurry. The manure excreted during grazing is allowed to lie as is.

Based on these assumptions, the amount of manure collected is calculated:

Farm model

$$Manure_{collected} = Excretion_{dairycow} * Housing_{dairycow} + Excretion_{follower} * Housing_{follower} \quad [1]$$

Where $Manure_{collected}$ is the amount of manure collected when the dairy cows and followers are housed, tonnes year^{-1}

$Housing_{dairycow}$ is the percentage of housing of dairy cows, expressed as decimal

$Housing_{follower}$ is the percentage of housing of followers, expressed as decimal

Manure deposited ($Manure_{deposited}$) while grazing is calculated:

$$Manure_{deposited} = Excretion_{dairycow} * (1\text{-}Housing_{dairycow}) + Excretion_{follower} * (1\text{-}Housing_{follower}) \quad [2]$$

3.4 Mineral fertiliser requirement

Slurry is applied to the land as organic fertiliser. The nutrient requirement of the crop that is not met by slurry is met by the import of mineral fertilisers. The amount of mineral fertiliser that needs to be imported is calculated as follows:

1) It is assumed that when the cows are housed, the manure is collected and stored in slurry tanks and subsequently applied to the crops, first to grass silage and then to winter wheat. When the cows are grazing, it is assumed that the manure is evenly spread on the pasture.
2) The amount of nutrients (N, P_2O_5 and K_2O) available in the slurry is calculated by multiplying the manure collected, as calculated in Section 3.3, and the nutrient content of slurry (Defra, 2010b) presented in Table 16.

Table 16 Nutrient content of slurry from dairy cows and followers

Nutrients Available (kg m^{-3})	N	P_2O_5	K_2O
Slurry - Dairy Cow	5.1	2.2	3.9
Slurry - Other cattle	4.1	1.7	3.9

3) The nutrient requirements of the pasture, grass silage and winter wheat are identified based on Defra (2010b) and presented in Table 17.

Table 17 Fertiliser requirement of crops

Crop requirement (kg/ha)	N	P_2O_5	K_2O
Pasture	240	50	30
Grass Silage	250	110	260
Winter wheat	220	95	115

4) The slurry is applied till the N requirement of the grass silage is met and the remainder is applied to the winter wheat.
5) The amount of mineral fertiliser required is calculated based on the balance of N, P_2O_5 and K_2O requirements of the pasture, grass silage and winter wheat that has not been met by the slurry.

Organic nitrogen application is assumed to be limited by the guidelines set for NVZ (Defra, 2009). In order to utilise the nutrients present, it has been assumed that the cattle slurry is applied as a priority and any remaining nutrient requirement of the crop/grass is met by use of mineral fertilisers. These are assumed to be applied as 'straights' (single nutrients) in order to meet the exact requirements of the crops. Figure 5 shows the module for the calculation of mineral fertilisers required with varying herd sizes and housing percentages.

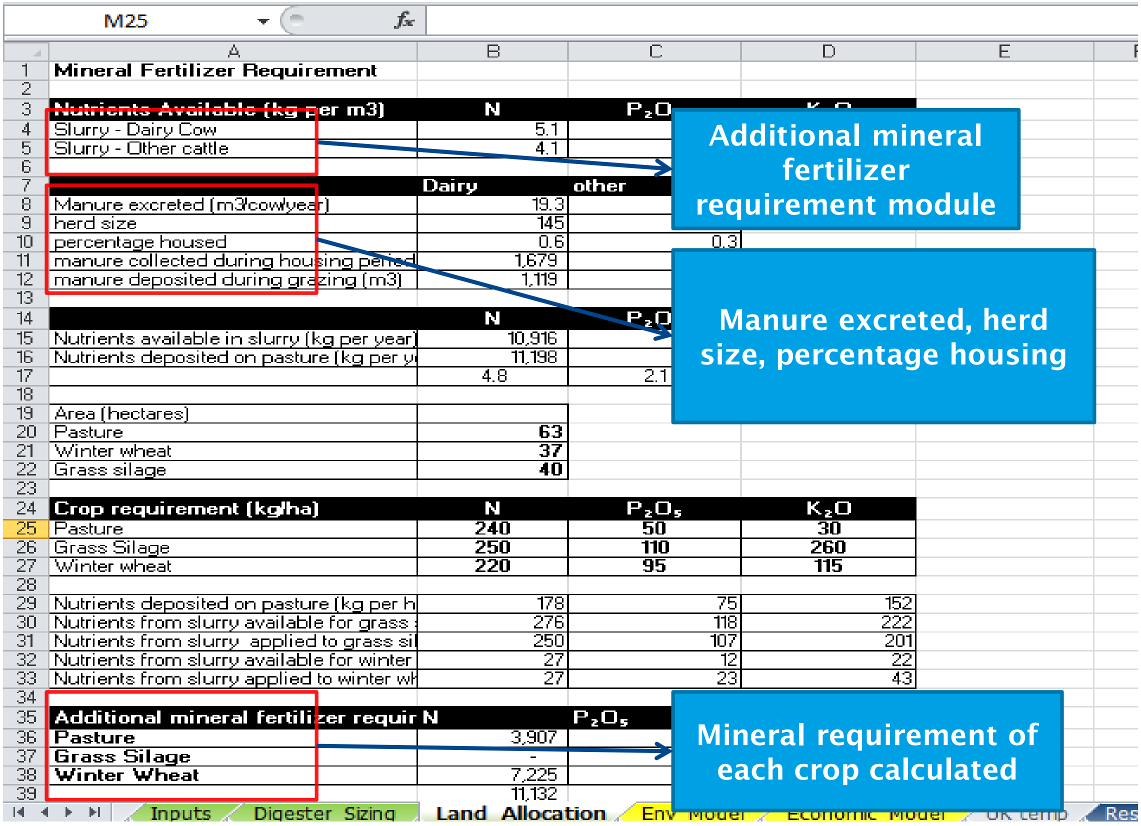


Figure 5 Mineral fertiliser calculation module

3.5 Digester and CHP Size

The digester is assumed to be a continuously stirred tank reactor (CSTR) in steady state. The slurry collected is fed to the digester on a daily basis and a portion of the digestate is removed at the same time. The amount of volatile solids present in the slurry has been calculated based on estimates of percentage of total solids and volatile solids in the slurry:

$$VS = Manure_{collected} * \%TS * \%VS * 1000 / 365 \qquad [3]$$

Where VS is daily volatile solid excreted, kg dry matter animal^{-1} day^{-1}

$Manure_{collected}$ is the manure collected while housing of cattle, tonnes animal^{-1} year^{-1} (calculated as per Section 3.3)

%TS is the proportion of total solids in the excreted manure, % (8% (Nijaguna, 2002))

%VS is the proportion of total solids excreted that are volatile, % (80% (Nijaguna, 2002))

The working volume of the digester required to digest the slurry is based on an organic loading rate (OLR) of 3 kg VS m^{-3} day^{-1} (Nijaguna, 2002).

$DV_m = VS/OLR$ [4]

Where DV_m is the digester volume required for manure, m^3

VS is the daily load of volatile solids to be added, kg VS day^{-1}

It is assumed that the slurry is evenly available throughout the year. Seasonal variation in slurry collection is not considered as the budgeting of both emissions and revenue is done on a yearly basis.

Allowing for 10% of the digester volume for gas collection, the final volume of the digester (DV) is calculated.

$DV = DV_m + 10\%\ of\ DV_m$ [5]

Assuming a cylindrical shape for the digester with a diameter to height ratio (R) of 4 (Samer, 2012), the radius (r) and height (H) of the digester are calculated.

$r^3 = (DV * R)/ (2 * \pi)$ [6]

$H = 2r/R$ [7]

Retention time (RT, days):

$RT = DV * 365/ Manure_{collected}$ [8]

3.6 Methane captured

The volume of biogas that is produced from anaerobically digesting the volatile solids in the manure is dependent on the retention time of the system, as reported in Section 2.3.2.1. The longer the retention time, the closer the methane produced is to the specific methane yield of the manure. However, the retention time of the manure in the digester is much shorter than that reported for the specific methane yield so only a proportion of the biogas is produced here, the rest of the biogas is potentially released while the digestate is in storage. The extended period of storage (up to 150 days as required by the NVZ regulations) allows for the breakdown of most of the remaining volatile solids. In the system modelled here it is assumed that the digestate storage containers are fully enclosed, allowing the capture of any biogas produced. The methane yield for the manure digested is therefore assumed to be close to the specific methane yield. The methane captured is initially stored in the digester (the digester and storage spaces being connected) and then used in either a CHP unit or boiler.

Any remaining methane not captured (e.g. from digestate stored for less than the extended period) is accounted for in the field based methane emissions from the applied digestate. For the initial modelling runs it is assumed that the cattle are being fed on a grass and concentrate based diet leading to a specific methane yield (B_o) average of 0.141 m^3 CH_4 kg^{-1} VS added (Amon *et al.*, 2007, Cornell, 2011, Møller *et al.*, 2004b). The methane produced ($CH_{4produced}$, m^3 $year^{-1}$) is calculated as below.

$$CH_{4produced} = VS * B_o * 365 \quad [9]$$

The module built for the calculation of digester size required and the methane produced is presented in Figure 6.

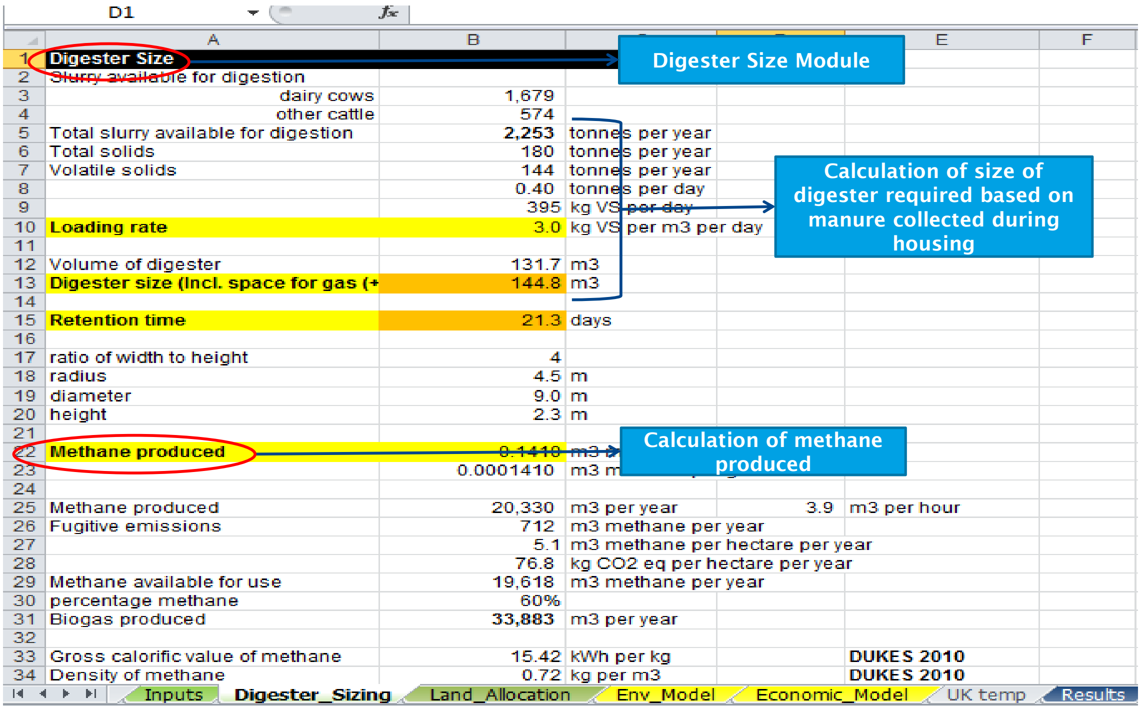


Figure 6 Digester size calculation module

The energy value of biogas is estimated assuming 60% of biogas produced is CH_4 with a gross calorific value of methane (CV_{CH4}) of 15.4166 kWh kg^{-1} and density of methane (D_{CH4}) of 0.717 kg m^{-3} (DECC, 2010b). The installed capacity of CHP ($CHP_{installed}$, kW) required is calculated as below:

$$CHP_{installed} = CH_{4produced} * CV_{CH4} * DC_{H4} / (365*24) \quad [10]$$

The heat and electricity generated by the CHP have been calculated as explained in Sections 4.8 and 4.9.

4. Emissions model

The emissions model is based on emissions factors for the three main greenhouse gases, CO_2, CH_4 and N_2O. The methods used to determine these are based on IPCC methodology (IPCC, 2006) explained in Section 2.3.9. Since no direct measurements have been taken or planned during the course of this study, Tier 2 methodology has been used wherever possible. Tier 1 estimates have been made only in case of absence of reliable data. All emissions have been calculated in carbon dioxide equivalents (CO_2 eq.) using global warming potentials (GWP) of 21 and 310 for methane (CH_4) and nitrous oxide (N_2O), respectively (IPCC, 1996). Since farm land can be utilised with varying intensity and variable number of cows can be stocked on the land, the emissions from the farm have been averaged over the farming area. It may be noted that emissions stemming from land use change (for example grassland or forestland to cropland) have been assumed to be negligible, as most of the land in the UK is already managed and deforestation for land use change is minimal. This assumption is based on the fact that the total grassland area has increased in England since the year 2000, mainly due to increases in permanent grassland, although it is still lower than in 1990 (Fowell, 2010). Also, the forest area in the UK has increased at the rate of 0.31% annually from 2000 till 2010 (Forestry Commission, 2012).

This chapter presents the methods and equations used in determining the emissions.

4.1 Enteric emissions

A proportion of gross energy intake of the dairy cow is emitted in the form of enteric emissions. The gross energy intake can be back calculated based on quantity and digestibility of the feed and the net energy requirements of the dairy cow, which in turn can be calculated based on its maintenance and growth needs, level of activity, lactation and pregnancy status. The enteric emissions are then calculated based on an annual emission factor and the total energy intake from the feed.

Emissions model

The calculations are applied separately for dairy cows and other cattle under housed and grazed conditions and then summed to get the total annual enteric emissions for the farm. The detailed steps and calculations used are as follows.

1) The net energy for maintenance is the energy required by the cow to maintain body weight

$NE_m = C_f * (weight)^{0.75}$ [11]

Where NE_m is the net energy required by the animal for maintenance, MJ day^{-1}

C_f is an empirically derived coefficient (0.386 for lactating cows), MJ day^{-1} kg^{-1}

Weight is the average live weight of a UK cow, kg (Dairy Cow – 650kg (Defra, 2009)), other cattle- 400kg (Defra, 2009)).

2) From the maintenance energy the net energy required by the animal for its daily activities can be calculated based on the activity levels of the cow (higher for grazing animals as they have to walk to and from grazing areas

$NE_a = C_a * NE_m$ [12]

Where NE_a is net energy for animal activity, MJ day^{-1}

C_a is the coefficient corresponding to animal's feeding situation (0.00 for confined animals, 0.17 for animals grazing a pasture (IPCC, 2006)).

3) Net energy required by the animal for growth

$NE_g = 22.02 * (BW/(0.8*MW))^{0.75} * WG^{1.097}$ [13]

Where NEg is net energy needed for growth, MJ day^{-1}

BW is the average live body weight of the animals in the population, kg (400 kg for other cattle (Defra, 2009)

MW is the mature live weight of an adult female in moderate body condition, kg (650 kg for dairy cow ((Defra, 2009))

WG is the average daily weight gain of the animals in the population (assumed 0.00 for dairy cows as they have assumed to have reached maturity, 0.4 for followers based on (EPA, 1994), kg day^{-1}.

4) Net energy required by lactating dairy cows for the production of milk

$NE_l = Milk * (1.47 + 0.40 * Fat)$ [14]

Where NE_l is net energy for lactation, MJ day^{-1}

Milk is the average amount of milk produced by a dairy cow in the UK, kg milk day^{-1} (7406 litres year^{-1} (Defra, 2011a))

Fat is the fat content of milk, % by weight (3.8% (Nix, 2012)).

5) Additional net energy required by pregnant dairy cows for maintenance

$$NE_p = C_{pregnancy} * NE_m \qquad [15]$$

Where NE_p is net energy required for pregnancy, MJ day^{-1}

$C_{pregnancy}$ is pregnancy coefficient (0.10 for cows (IPCC, 2006))

NE_m is net energy required by the animal for maintenance, MJ day^{-1}.

6) Ratio of net energy available in diet for maintenance to digestible energy consumed (REM)

$$REM = [1.123 - (4.092*10^{-3} *DE\%) + [1.126*10^{-5}*(DE\%)^2] - (25.4/DE\%)] \qquad [16]$$

Where DE% is digestible energy expressed as a percentage of gross energy (75% for grass and winter wheat (IPCC, 2006)).

7) Ratio of net energy available for growth in a diet to digestible energy consumed (REG)

$$REG = [1.164 - (5.160*10^{-3}*DE\%)+[1.308*10^{-5}*(DE\%)^2]-(37.4/DE\%)] \qquad [17]$$

Where DE% is digestible energy expressed as a percentage of gross energy (75% for grass and winter wheat (IPCC, 2006))

8) Gross energy intake (GE) is then calculated using Equation 11 to Equation 17

$$GE = [((NE_m + NE_a + NE_l + NE_p)/REM) + (NE_g/REG)]/(DE\%/100) \qquad [18]$$

The annual enteric emission factor is calculated.

$$EF_{enteric} = (GE * (Y_m/100)*365)/55.65 \qquad [19]$$

Where $EF_{enteric}$ is the emission factor, kg CH_4 head^{-1} year^{-1}

GE is gross energy intake, MJ head^{-1} day^{-1} (calculated from Equation 18)

Y_m is methane conversion factor, per cent of gross energy in feed converted to methane (6.5% of GE (IPCC, 2006))

55.65 (MJ kg^{-1} CH_4) energy content of methane (DECC, 2012f).

The total enteric emissions (kg CO_2 eq. ha^{-1} year^{-1})

$$E_{enteric} = EF_{enteric} * \text{Number of cows} * GWP_{CH4} /FarmSize \qquad [20]$$

Where FarmSize is the size of the farm, hectares.

Figure 7 shows the calculation of enteric emissions from grazed dairy cows. Similar calculations are made for housed dairy cows and other cattle.

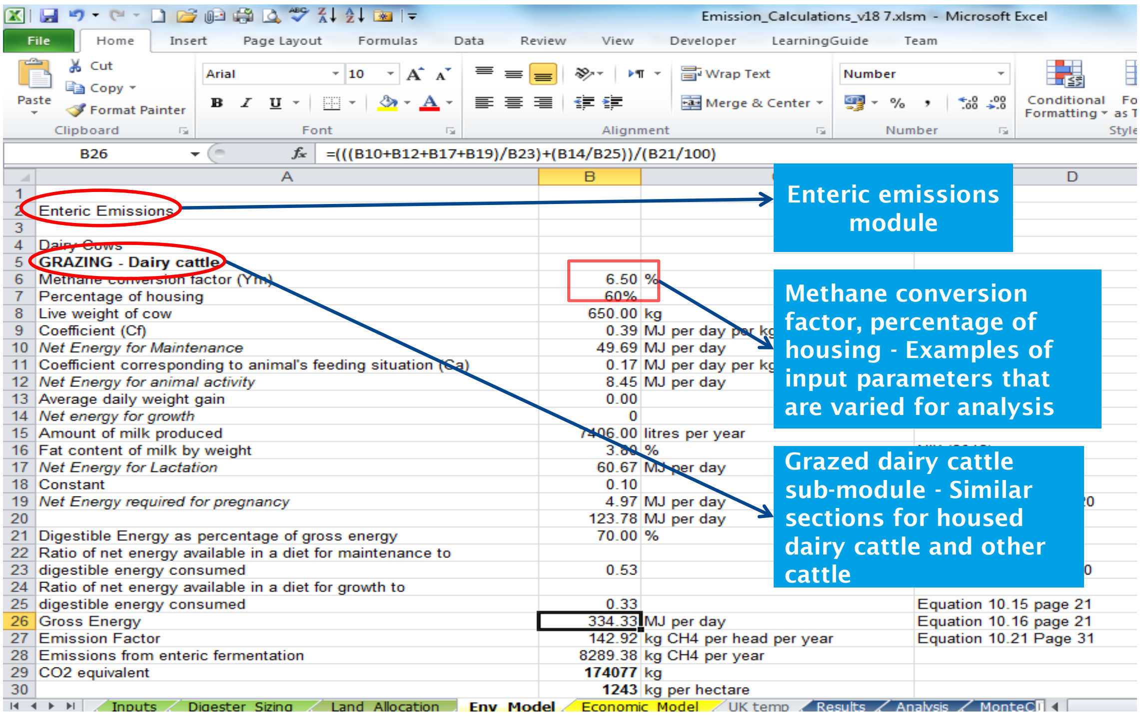


Figure 7 Enteric emissions calculation module

4.2 Manure Management

Manure management results in the emissions of both CH_4 and N_2O. These are calculated using the methodology presented in the following sections, where slurry is defined as dairy cow manure with minimal amount of water addition.

4.2.1 Methane

Manure management is classified under two broad headings in this research:

Housed - It is assumed that manure is managed in a slurry based system and is stored in a slurry tank for up to 6 months before being applied to the field as an organic fertiliser. This is typical for UK dairy farms when the cows are housed.

Grazed - Excreta from grazed dairy cows and followers are assumed to be spread evenly on the pasture.

Emission factor for methane emissions from manure:

$$EF_{manure} = (VS * HousingFactor * 365) * [B_o * 0.716 * \Sigma((MCF/100)*MS)] \quad [21]$$

Where EF_{manure} is annual methane emission factor, kg CH_4 animal^{-1} year^{-1}

365 is conversion factor for days in a year, days year^{-1}

0.716 is conversion factor of m^3 CH_4 to kg CH_4.

MCF represents the methane conversion factors for each manure management system by climate regions, % (Grazing 1%, Slurry 10% for average annual temperature <10°C (IPCC, 2006)).

MS is fraction of livestock whose manure is handled using each manure management system, dimensionless.

HousingFactor is HousingPercentage for housed cows and (1-HousingPercentage) for grazed cows.

HousingPercentage is the proportion of time in an average year that the cows spend indoors.

Total emissions of methane from manure management

$$E_{manure,CH4} = EF_{manure,CH4} * \text{number of cows} * GWP_{CH4} / FarmSize \quad [22]$$

Figure 8 shows a screen shot of the spread sheet calculation for methane emissions from manure management of housed and grazed dairy cows. Similar calculations are made for other cattle.

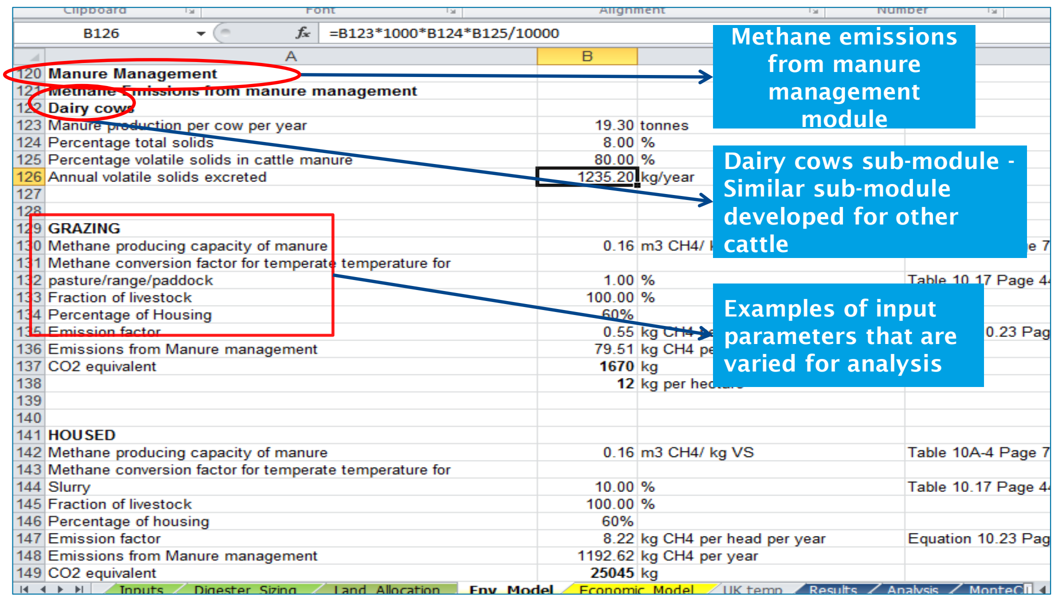


Figure 8 Manure management calculations module

4.2.2 Nitrous Oxide

The manure deposited by the cattle while grazing on pastures is allowed to lie as is. Direct and indirect emissions associated with the deposited manure are, therefore, treated as emissions from managed soils (IPCC, 2006) using the methods below.

4.2.2.1 Direct Emissions

N_2O emissions from manure management are based on the amount of nitrogen excreted by the dairy cows and followers and an emission factor that varies with the method of managing the manure

$$N_2O_{d\text{-}mm} = N_t * N_{ex} * MS * EF_3 * (44/28) \qquad [23]$$

Where $N_2O_{d\text{-}mm}$ is direct N_2O emissions from manure management, kg N_2O year^{-1}

N_t is the number of head of livestock species

N_{ex} is the annual average N excretion per head, kg N head^{-1} year^{-1} (0.27 kg per animal per day – dairy cows, 0.164 kg per animal per day – other cattle (Defra, 2009))

MS is the fraction of total annual nitrogen excretion that is managed in the manure management system, dimensionless

EF_3 is emission factor for direct N_2O emissions from manure management system, kg N_2O-N kg^{-1} N in manure management system (0.005 kg N_2O-N kg^{-1} N excreted for liquid/slurry with natural crust cover, 0 without crust cover, 0 for grazing (IPCC, 2006))

44/28 conversion of N_2O-N emissions to N_2O emissions.

4.2.2.2 Indirect Emissions

Indirect emissions originating from volatilisation/leaching of N as ammonia or oxides of nitrogen are calculated based on the amount of nitrogen excreted by the cow, the proportion of the N excreted that volatilises/leaches and a respective emission factor. The fraction of excreted N that volatilises/leaches depends on the manure management system. It has been assumed that there are no nitrogen losses from leaching while the manure is being managed in a slurry storage tank.

Volatilisation

$$N_2O_{g\text{-}mm} = N_t * N_{ex} * MS * (Frac_{gasMS}/100) * EF_4 * (44/28) \qquad [24]$$

Where $N_2O_{g\text{-}mm}$ is indirect emissions due to volatilisation of N from manure management, kg N_2O year^{-1}

$Frac_{gasMS}$ is the percentage of managed manure nitrogen that volatilises as NH_3 and NO_x in the manure management system, % (40% for Liquid/Slurry management (IPCC (2006)))

EF_4 is the emission factor for N_2O emissions from atmospheric deposition of nitrogen on soils and water surfaces, kg N_2O-N per kg NH_3-N + NO_x-N volatilised (0.01 kg N_2O-N kg^{-1} NH_3-N + NO_x-N volatilised (IPCC, 2006)).

Leaching

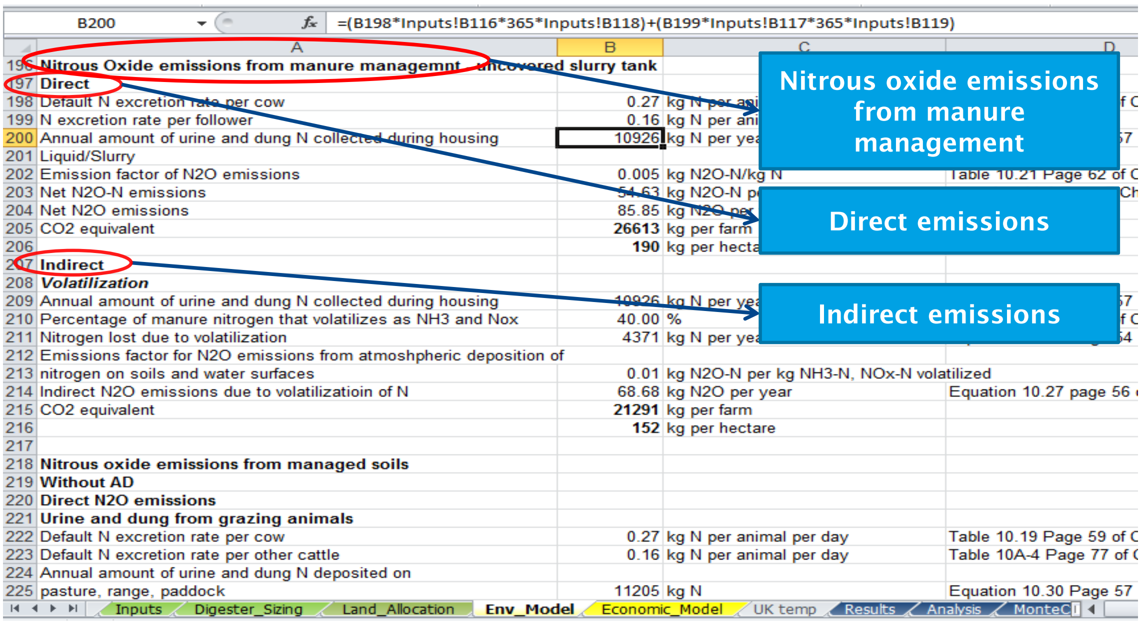
$$N_2O_{l\text{-}mm} = N_t * N_{ex} * MS * (Frac_{leachMS}/100) * EF_5 * (44/28) \qquad [25]$$

Where $N_2O_{l\text{-}mm}$ are the indirect N_2O emissions due to leaching and runoff from manure management, kg N_2O year^{-1}

$Frac_{leachMS}$ is the percentage of managed manure nitrogen losses due to run-off and leaching during solid and liquid storage of manure (0 for Slurry management, 30% for daily spreading and grazing (IPCC, 2006))

EF_5 is the emission factor for N_2O emissions from nitrogen leaching and runoff, kg N_2O-N kg^{-1} N leached and runoff (0.0075 kg N_2O-N kg^{-1} N leached and runoff).

Figure 9 shows the calculation of nitrous oxide emissions from managed manure



	A	B	C	D
196	Nitrous Oxide emissions from manure managemnt - uncovered slurry tank			
197	Direct			
198	Default N excretion rate per cow	0.27	kg N per ani	
199	N excretion rate per follower	0.16	kg N per ani	
200	Annual amount of urine and dung N collected during housing	10926	kg N per yea	
201	Liquid/Slurry			
202	Emission factor of N2O emissions	0.005	kg N2O-N/kg N	Table 10.21 Page 62 of C
203	Net N2O-N emissions	54.63	kg N2O-N p	
204	Net N2O emissions	85.85	kg N2O per	
205	CO2 equivalent	26613	kg per farm	
206		190	kg per hecta	
207	Indirect			
208	*Volatilization*			
209	Annual amount of urine and dung N collected during housing	10926	kg N per yea	
210	Percentage of manure nitrogen that volatilizes as NH3 and Nox	40.00	%	
211	Nitrogen lost due to volatilization	4371	kg N per yea	
212	Emissions factor for N2O emissions from atmoshpheric deposition of			
213	nitrogen on soils and water surfaces	0.01	kg N2O-N per kg NH3-N, NOx-N volatilized	
214	Indirect N2O emissions due to volatilizatioin of N	68.68	kg N2O per year	Equation 10.27 page 56
215	CO2 equivalent	21291	kg per farm	
216		152	kg per hectare	
217				
218	Nitrous oxide emissions from managed soils			
219	Without AD			
220	Direct N2O emissions			
221	Urine and dung from grazing animals			
222	Default N excretion rate per cow	0.27	kg N per animal per day	Table 10.19 Page 59 of C
223	Default N excretion rate per other cattle	0.16	kg N per animal per day	Table 10A-4 Page 77 of (
224	Annual amount of urine and dung N deposited on			
225	pasture, range, paddock	11205	kg N	Equation 10.30 Page 57

Figure 9 Nitrous oxide emission calculations from managed manure

The total methane and nitrous oxide emissions from management of manure are then calculated using Equation 23 to 25.

Emissions model

$E_{manure} = (E_{manure,CH4} * GWP_{CH4}) + ((N_2O_{d\text{-}mm} + N_2O_{g\text{-}mm} + N_2O_{l\text{-}mm}) * GWP_{N2O})$ [26]

4.2.3 Digestate storage emissions

Digestate is assumed to be stored in air tight tanks such that all the CH_4 produced during storage is captured and sent to the CHP unit for energy production. Hence the CH_4 emissions from storage of digestate are negligible. Due to lack of oxygen in the airtight tanks and the anaerobic digestate, it is assumed that no N_2O is produced in digestate storage.

4.3 Managed soils

Direct N_2O emissions from managed soils include emissions from excreta deposited by grazing animals, application of synthetic and organic fertilisers, N from crop residues, and drainage/management of organic soils.

4.3.1 Direct Emissions

Direct N_2O emissions are calculated by summing emissions from various forms of N additions to the soil, excreta deposited by grazing animals and drainage/management of organic soils using Equation 27 to Equation 29:

$N_2O_{direct\text{-}N} = N_2O\text{-}N_{n\ inputs} + N_2O\text{-}N_{prp}$ [27]

Where

$N_2O\text{-}N_{n\ inputs} = [(F_{sn} + F_{on} + F_{cr}) * EF_1]$ [28]

$N_2O\text{-}N_{prp} = F_{prp} * EF_{3prp}$ [29]

Where $N_2O_{direct\text{-}N}$ is annual direct N_2O-N emissions produced from managed soils, kg N_2O-N $year^{-1}$.

$N_2O\text{-}N_{n\ inputs}$ annual direct N_2O-N emissions from N inputs to managed soils, kg N_2O-N $year^{-1}$.

$N_2O\text{-}N_{prp}$ annual direct N_2O-N emissions from urine and dung inputs to grazed soils, kg N_2O-N $year^{-1}$

F_{sn} annual amount of synthetic fertiliser N applied to soils, kg N year^{-1} (calculated as per Section 3.4)

F_{on} annual amount of animal manure, compost, sewage sludge and other organic additions, kg N year^{-1} (calculated)

F_{cr} annual amount of N in crop residues (above-ground and below-ground), including N-fixing crops, and from forage/pasture renewal, returned to soils, kg N year^{-1} (calculated as per Equation 31)

F_{prp} annual amount of urine and dung N deposited by grazing animals on pasture, range and paddock, kg N year^{-1} (calculated)

EF_1 emission factor for N_2O emissions from N inputs, kg N_2O-N kg^{-1} N input (0.01 kg N_2O-N kg^{-1} N input (IPCC, 2006))

EF_{3prp} emission factor for N_2O emissions from urine and dung N deposited on pasture, range and paddock by grazing animals, kg N_2O-N kg^{-1} N input (0.02 kg N_2O-N kg^{-1} N input (IPCC, 2006)).

Nitrogen added by crop residues is derived by estimating the mass of the plant left behind after the crop has been harvested and the nitrogen concentration in the above and below ground organic matter. The total N addition from crop residues is the sum of the above-ground and below-ground N contents.

$$F_{cr} = AG_{dm} * Area * F_{renew} * [N_{ag} * (1 - F_{remove}) + R_{bg\text{-}bio} * N_{bg}] \qquad [30]$$

Where AG_{dm} is above ground residue dry matter, 10^6 grams ha^{-1}

Area total annual area harvested of crop, ha year^{-1}

F_{renew} fraction of total area under crop that is renewed annually

N_{ag} N content of above-ground residues for crop, kg N kg^{-1} dry matter

F_{remove} fraction of above ground residues of crop removed annually for purposes such as feed, bedding and construction, kg N kg^{-1} crop-N

$R_{bg\text{-}bio}$ ratio of belowground residues to aboveground biomass

N_{bg} N content of belowground residues for crop, kg N kg^{-1} dry matter

4.3.2 Indirect Emissions

Indirect emissions occur from the breakdown and conversion of nitrogen applied to the fields. These emissions occur mainly in two forms, volatilisation and leaching.

Emissions from the volatilisation of N

Emissions model

$$N_2O_{ATD} = [(F_{sn} * Frac_{GASF}) + ((F_{on} + F_{prp}) * Frac_{GASM})] * EF_4 *(44/28) \qquad [31]$$

Where N_2O_{ATD} are the indirect N_2O emissions due to volatilisation and subsequent deposition from manure management, kg N_2O year^{-1}

$Frac_{GASF}$ is the fraction of synthetic fertiliser N that volatilises as NH_3 and NO_x, kg N volatilised kg^{-1} N applied (0.10 kg N volatilised per kg N applied (IPCC, 2006))

$Frac_{GASM}$ is the fraction of applied organic N fertiliser materials and of urine and dung deposited by grazing animals that volatilises as NH_3 and NO_x, kg N volatilised kg^{-1} N applied or deposited (0.20 kg N volatilised kg^{-1} N applied or deposited (IPCC, 2006))

EF_4 is the emission factor for N_2O emissions from atmospheric deposition of N on soils and water surfaces, kg N_2O-N (kg NH3-N + NOx-N volatilised)$^{-1}$ (0.01 kg N_2O-N (kg NH_3-N + NO_x-N volatilised)$^{-1}$(IPCC, 2006))

Emissions from leaching of N are calculated

$$N_2O_L = (F_{sn} + F_{on} + F_{prp} + F_{cr}) * Frac_{LEACH-H} * EF_5 *(44/28) \qquad [32]$$

Where N_2O_L are the indirect N_2O emissions due to leaching and runoff from manure management, kg N_2O year^{-1}

$Frac_{LEACH-H}$ is fraction of all N added to/mineralised in managed soils in regions where leaching/runoff occurs that is lost through leaching and runoff, kg N kg^{-1} N additions (0.30 kg N kg^{-1} N additions (IPCC, 2006))

EF_5 is the emission factor for N_2O emissions from leaching and runoff, kg N_2O-N kg^{-1} N leached and runoff (0.0075 kg N_2O-N kg^{-1} N leached and runoff (IPCC (2006))).

The total nitrous oxide emissions from managed soils are then calculated using Equations 26, 30 and 31.

$$E_{soils} = (N_2O_{direct-N} + N_2O_{(ATD)} + N_2O_L) * GWP_{N2O} \qquad [33]$$

Figure 10 presents a part of the module written for the calculation of N_2O emissions from managed soils. The module is linked with various input parameters and modules for example, herd size, percentage housing and the mineral fertiliser calculation module.

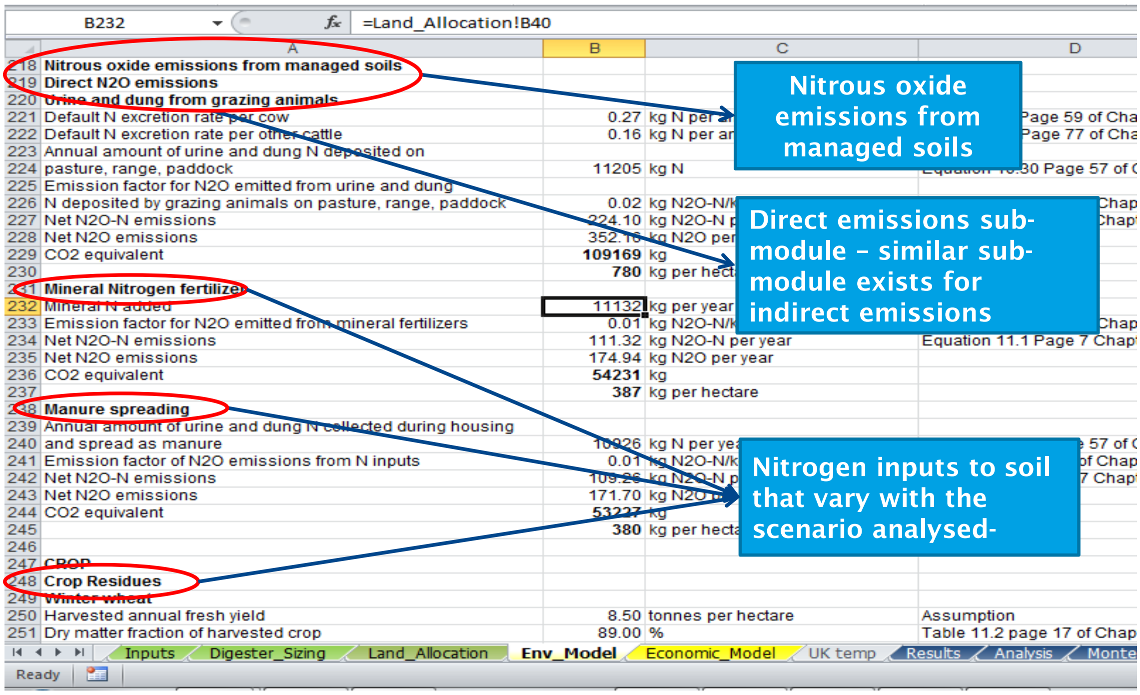


Figure 10 Nitrous oxide emissions from managed soils module

4.3.3 Digestate application emissions

The factors governing emissions from application of digestate are presented in Section 2.3.7.2. IPCC does not specify any emission factors for emissions related to the application of digestate. Due to lack of quantitative data available, the equations and emission factors calculated for slurry application are used for digestate as well.

4.4 Use of fuel in farm machinery

It is assumed that diesel fuel is used in all the farm machinery. The total energy required for farming depends on the crop type, the machinery used, climatic conditions, number of fertiliser and pesticide applications etc. These are calculated using an energy model presented in Salter and Banks (2009) including both direct and indirect energy usage. The amount of energy required multiplied by the emission factor gives us the total emissions from use of machinery.

$$E_{diesel} = (CV_{diesel} / Density_{diesel}) * 277.78 * FU_{diesel} * EF_{diesel} \quad [34]$$

Where E_{diesel} is the emissions from usage of diesel on-farm, kg CO_2 eq. year^{-1}

CV_{diesel} is the calorific value of diesel, GJ $tonne^{-1}$ (42.81 GJ $tonne^{-1}$(DECC, 2010b))

$Density_{diesel}$ is the density of diesel, litres $tonne^{-1}$ (1198 litres $tonne^{-1}$(DECC, 2010b))

277.78 is conversion factor for converting GJ to kWh

FU_{diesel} is the fuel usage on farm, litres $year^{-1}$ (calculated)

EF_{diesel} is the emission factor of GHG emissions from use of diesel, kg CO_2 eq. kWh^{-1} $year^{-1}$ (0.3 kg CO_2 eq. kWh^{-1}(DECC, 2010b)).

4.5 Production of mineral fertilisers

In order to calculate the emissions from the production of mineral fertilisers that are imported and used on the farm, the amount of mineral fertilisers required meet the requirements of the crops is calculated as outlined in Section 3.4. The total emissions from manufacture of the calculated mineral fertilisers are derived as shown below.

$$E_{fertiliser} = \Sigma\,(FU_{i,fertiliser} * EF_{i,fertiliser}) \qquad [35]$$

Where $E_{i,fertiliser}$ is the emissions from production of fertiliser used, kg CO_2 eq. $year^{-1}$

$FU_{i,fertiliser}$ is the fertiliser used, kg $year^{-1}$ (calculated in section 3.4)

$EF_{i,fertiliser}$ is the emission factor from production of fertiliser used, kg CO_2 eq. per kg (7.11 kg eq. CO_2 kg^{-1} nitrogen, 1.85 kg eq. CO_2 kg $P_2O_5^{-1}$, 1.76 kg eq. CO_2 kg^{-1} K_2O (Mortimer *et al.*, 2007))[2]

i is the type of fertiliser.

There may be additional emissions from transport of the fertilisers to the farm, which are not studied here.

4.6 Embodied carbon

Embodied carbon is defined in Section 2.3.7.4. The total embodied carbon is calculated based on the amount of concrete, steel and polyurethane used in the construction of the digester. The digester is assumed to be cylindrical (Section 3.5), with a square reinforced concrete slab as base. The thickness of the concrete walls and the slab are assumed to be 300mm (Samer, 2012) and that of the polyurethane coating 60mm (German Solar Energy Society and Ecofys, 2004). 10mm steel rods are provided at 14m m^{-2} as reinforcement for concrete in walls and slab. The required volume of each construction material is hence calculated. The total embodied carbon for the digester averaged over its lifetime is calculated assuming no recycling. The embodied carbon in the ancillary equipment (CHP unit, additional pumps, pipes, etc.) is low when calculated per hectare and over the life time of the digester (Gazis and Harrison, 2011) and hence has not been included in the model.

$$E_{EC} = \Sigma (V_i * Density_i * EC_i)/Lifetime_{digester} \qquad [36]$$

Where E_{EC} is embodied carbon in the construction materials used, kg CO_2 eq. year^{-1}

V_i is the volume of construction material used (calculated)

$Density_i$ is the density of the construction material (2.24 tonnes m^{-3}, 7.8 tonnes m^{-3} and 0.03 tonnes m^{-3} for concrete, steel and insulation, respectively (Hammond and Jones, 2008))

EC_i is the embodied carbon in 1 kg of construction material (0.13 kg CO_2 eq. kg^{-1} concrete, 1.77 kg CO_2 eq. kg^{-1} steel, 1.86 kg CO_2 eq. kg^{-1} insulation (polyurethane) (Hammond and Jones, 2008))

i is the building material, concrete, steel and insulation.

$Lifetime_{digester}$ is the lifetime of a digester (20 years to be consistent with mortgage payments).

4.7 Fugitive Emissions

The sources of fugitive emissions are discussed in Section2.3.7.3. The fugitive emissions are calculated as shown below.

$$E_{FE} = CH_{4produced} * \%FE * GWP_{methane} \qquad [37]$$

Where E_{FE} is the fugitive emissions, kg CO_2 eq. year^{-1}

%FE is the percentage of methane produced that is released as fugitive emissions, per cent (3.5% (Silsoe Research Institute, 2000)).

4.8 Electricity import/export

The annual electricity consumption on a dairy farm (E_{dairy}) is estimated at 218 kWh cow^{-1} (Dunn *et al.*, 2010). When it is imported from the national grid the GHG emission factor ($EF_{electricity}$) is assumed to be 0.59 kg CO_2 eq. kWh^{-1} (DECC, 2012f).

Emissions model

When an anaerobic digester is operating on the farm, it is assumed that the biogas produced is burnt in a Combined Heat and Power (CHP) unit to produce both heat and electricity.

It may be noted that the emissions from burning of biogas are not considered as they are a part of the natural biological carbon cycle. CO_2 is absorbed from the atmosphere by the plant and is converted into biomass which is consumed by the cattle and is excreted and emitted enterically. A part of the carbon excreted as manure is converted into biogas. The biogas thus produced is burnt in a CHP unit to produce heat and electricity or flared, and the CO_2 produced is released back into the atmosphere, thus completing the carbon cycle.

The electricity produced is used to operate the digester and various other dairy operations like the milking parlour. Any surplus electricity after meeting in house requirements is exported to the national grid.

The electricity produced (E_{CHP}) was calculated

$$E_{CHP} = CH_{4produced} * CV_{CH4} * D_{CH4} * CHP_{electricity} \qquad [38]$$

Where E_{CHP} is the electricity generated, kWh year^{-1}

$CHP_{electricity}$ is the electrical efficiency of the CHP (0.35 (DECC, 2012c)).

The electricity requirement for running the digester equipment e.g. feeders, pumps, mixers, etc. ($E_{parasitic}$) was taken at 7.2 kWh tonne^{-1} of input slurry based on Berglund and Borjesson (2006).

The emissions from electricity usage/production in the dairy farm

$$E_{electricity} = EF_{electricity} * (E_{dairy} + E_{parasitic} - E_{CHP}) \qquad [39]$$

Where $E_{electricity}$ is the emissions from electricity produced and used on the dairy farm, kg CO_2 eq. year^{-1}.

4.9 Heat import/export

The heat requirement for a dairy farm (H_{dairy}) is estimated at 107 kWh cow^{-1} year^{-1} (Dunn *et al.*, 2010). It is assumed that in the case of a farm without a digester, this heat is supplied using liquefied petroleum gas (LPG), with a GHG emission factor (EF_{heat}) of 0.26 kg CO_2 eq. kWh^{-1} (DECC, 2012f). LPG is the 3rd largest primary energy input, with electricity (51%) and oils (primarily diesel at 32%) being the top 2 energy sources used (Warwick HRI, 2007). LPG is also cleaner than other agricultural fuels like fuel oil (DECC, 2012f) and hence assuming use of LPG is a conservative assumption; any other energy source would have a larger carbon footprint thus increasing the emissions abated by AD.

When a digester is operating, calculation of heat produced by the CHP (H_{CHP}) is based on the equation below.

$$H_{CHP} = CH_{4produced} * CV_{CH4} * D_{CH4} * CHP_{heat} \qquad [40]$$

Where H_{CHP} is the heat generated, kWh $year^{-1}$

CHP_{heat} is the thermal efficiency of the CHP (0.49 (DECC, 2012c)).

The heat requirement of a digester is comprised of the heat required to bring the feedstock from ambient temperature to the operating temperature of the digester plus the heat required to maintain it at this temperature ($H_{parasitic}$). The heat required to increase the temperature of the slurry:

$$Heat_{feedstock} = Manure_{collected} * \text{specific heat of water} * (T_{operating} - T_{ambient}) * 277.78 / 1000 \qquad [41]$$

Where $Heat_{feedstock}$ is the amount of heat required to increase the temperature of the feedstock from ambient to operating temperature, kWh $year^{-1}$

Specific heat of water is the amount of heat required to increase the temperature of a tonne of water by 1°C, MJ $tonne^{-1}$ C^{-1} (4.2 MJ $tonne^{-1}$ $°C^{-1}$)

$T_{operating}$ is the operating temperature of the digester, °C (38 °C)

$T_{ambient}$ is the ambient air temperature, °C (8.8 °C, average UK temperature (The Met Office, 2013))

277.78 is conversion factor for converting GJ to kWh.

The amount of heat required to maintain the operating temperature of the digester:

$$Heat_{maintain} = SA_{digester} * R_{effective} * (T_{operating} - T_{ambient}) * 8.760 \qquad [42]$$

Where $Heat_{maintain}$ is the heat required to maintain the operating temperature of the digester, kWh $year^{-1}$

$SA_{digester}$ is the surface are of the digester (= (2*π*r*H) + (2*π*(r^2)))

8.760 is a factor for converting W to kWh $year^{-1}$

$R_{effective}$ is the effective thermal conductivity

$$1/R_{effective} = Thickness_{concrete}/R_{concrete} + Thickness_{polyurethane}/R_{polyurethane} \qquad [43]$$

Where $Thickness_{concrete}$ is the thickness of concrete in digester construction (300mm) (Samer, 2012)

$R_{concrete}$ is the thermal conductivity of concrete (1.31 W m^{-2} C^{-1} (Hammond and Jones, 2008)

$Thickness_{polyurethane}$ is the thickness of polyurethane coating (60mm (German Solar Energy Society and Ecofys, 2004))

$R_{polyurethane}$ is the thermal conductivity of polyurethane (0.03 W m^{-2} C^{-1}(Hammond and Jones, 2008)

Total parasitic heat load of the digester is calculated by combining Equations 41 and 42

$$H_{parasitic} = H_{feedstock} + H_{maintain} \qquad [44]$$

This calculation is illustrated in Figure 11.

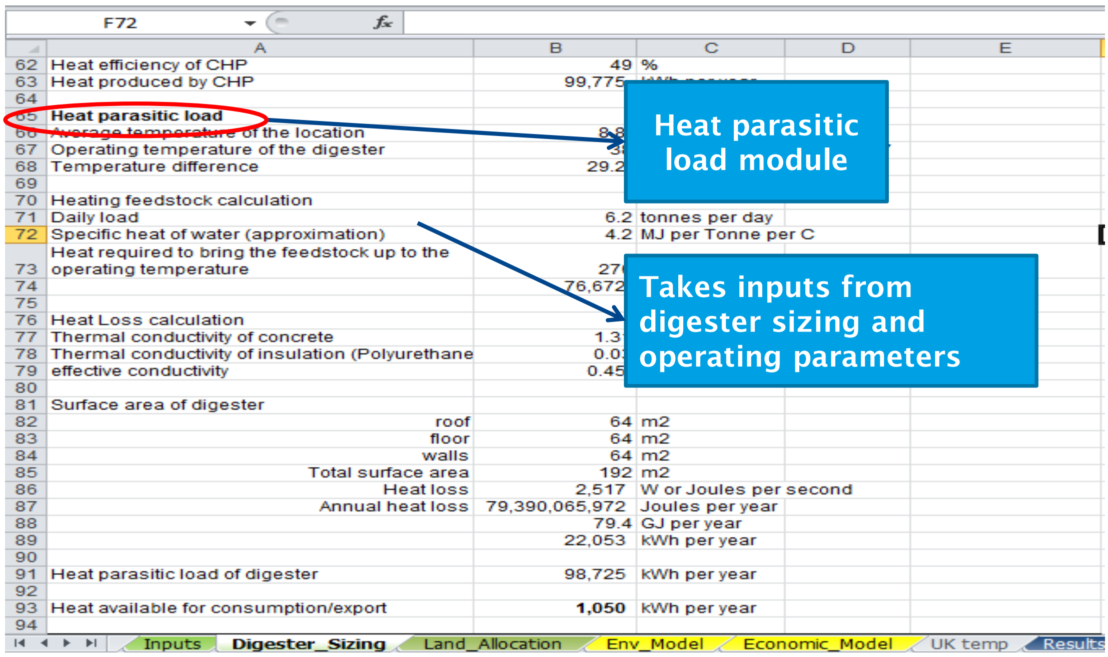


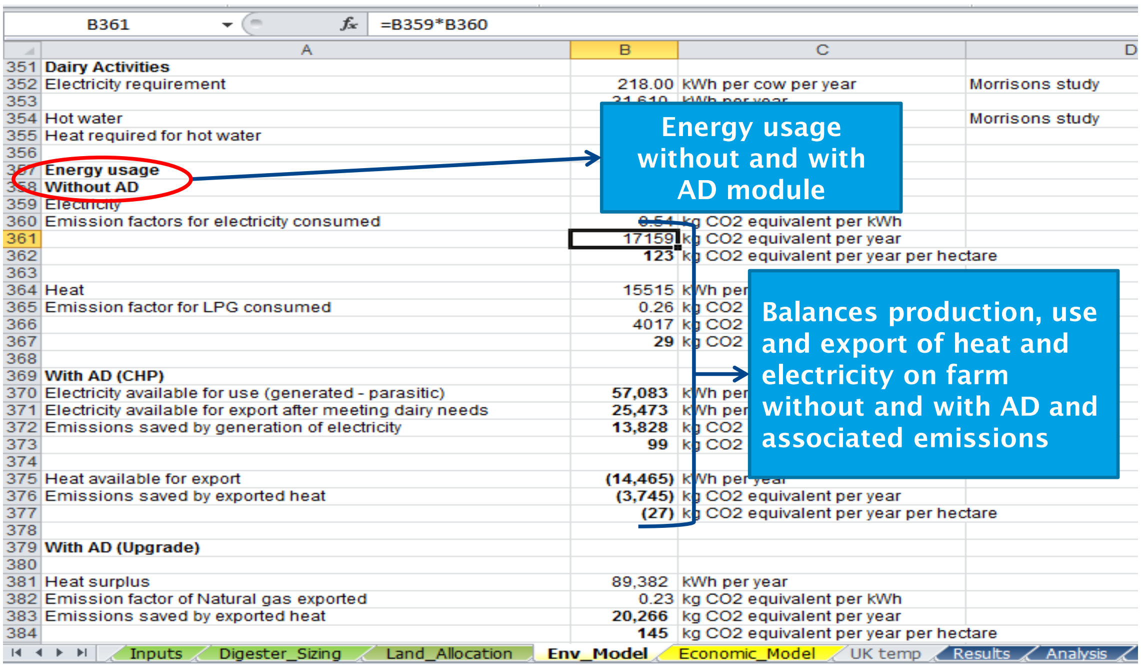
Figure 11 Module for calculating the heat parasitic load of digester

The parasitic load requirement varies through the year with the change in the ambient air temperature. For the purpose of this analysis an average annual air temperature for the UK is used as the ambient air temperature (The Met Office, 2013).

The emissions from production and usage of heat on a dairy farm are calculated based on the dairy heat requirement, Equations 40 and 44. An illustration is presented in Figure 12.

$$E_{heat} = EF_{heat} * (H_{dairy} + H_{parasitic} - H_{CHP}) \qquad [45]$$

Where E_{heat} are the emissions from usage of heat, kg CO_2 eq. year^{-1}



B361 =B359*B360

	A	B	C	D
351	Dairy Activities			
352	Electricity requirement	218.00	kWh per cow per year	Morrisons study
353		21,610	kWh per year	
354	Hot water			Morrisons study
355	Heat required for hot water			
356				
357	Energy usage			
358	Without AD			
359	Electricity			
360	Emission factors for electricity consumed	0.54	kg CO2 equivalent per kWh	
361		17159	kg CO2 equivalent per year	
362		123	kg CO2 equivalent per year per hectare	
363				
364	Heat	15515	kWh per	
365	Emission factor for LPG consumed	0.26	kg CO2	
366		4017	kg CO2	
367		29	kg CO2	
368				
369	With AD (CHP)			
370	Electricity available for use (generated - parasitic)	57,083	kWh per	
371	Electricity available for export after meeting dairy needs	25,473	kWh per	
372	Emissions saved by generation of electricity	13,828	kg CO2	
373		99	kg CO2	
374				
375	Heat available for export	(14,465)	kWh per year	
376	Emissions saved by exported heat	(3,745)	kg CO2 equivalent per year	
377		(27)	kg CO2 equivalent per year per hectare	
378				
379	With AD (Upgrade)			
380				
381	Heat surplus	89,382	kWh per year	
382	Emission factor of Natural gas exported	0.23	kg CO2 equivalent per kWh	
383	Emissions saved by exported heat	20,266	kg CO2 equivalent per year	
384		145	kg CO2 equivalent per year per hectare	

Figure 12 Heat and electricity use and export on the dairy farm without and with AD

4.10 Total Emissions

GHG emissions from the farm livestock, the management of manure and soils along with secondary emissions from burning of diesel fuel, manufacture of mineral fertilisers and heat and electricity production are summed to get total emissions from a farm under the given farming conditions in terms of kg CO_2 eq. ha^{-1} year^{-1}. Addition of emission sources from the introduction of an anaerobic digester in the form of fossil fuel substitution in form of heat and electricity, fugitive emissions and embodied carbon and changes in the existing ones are calculated for the farm set up with an anaerobic digester.

Total emissions are calculated using Equations 20, 26, 33, 34, 35, 36, 37, 39 and 45 and presented in Figure 13:

$$Emissions_{total} = E_{enteric} + E_{manure} + E_{soils} + E_{diesel} + E_{fertiliser} + E_{EC} + E_{FE} + E_{electricity} + E_{heat} \quad [46]$$

This methodology is applied repeatedly by choosing appropriate modules, emission factors and values for input parameters to analyse different scenarios and is further used in the calculation of marginal abatement cost of anaerobic digestion as explained in Section 6.

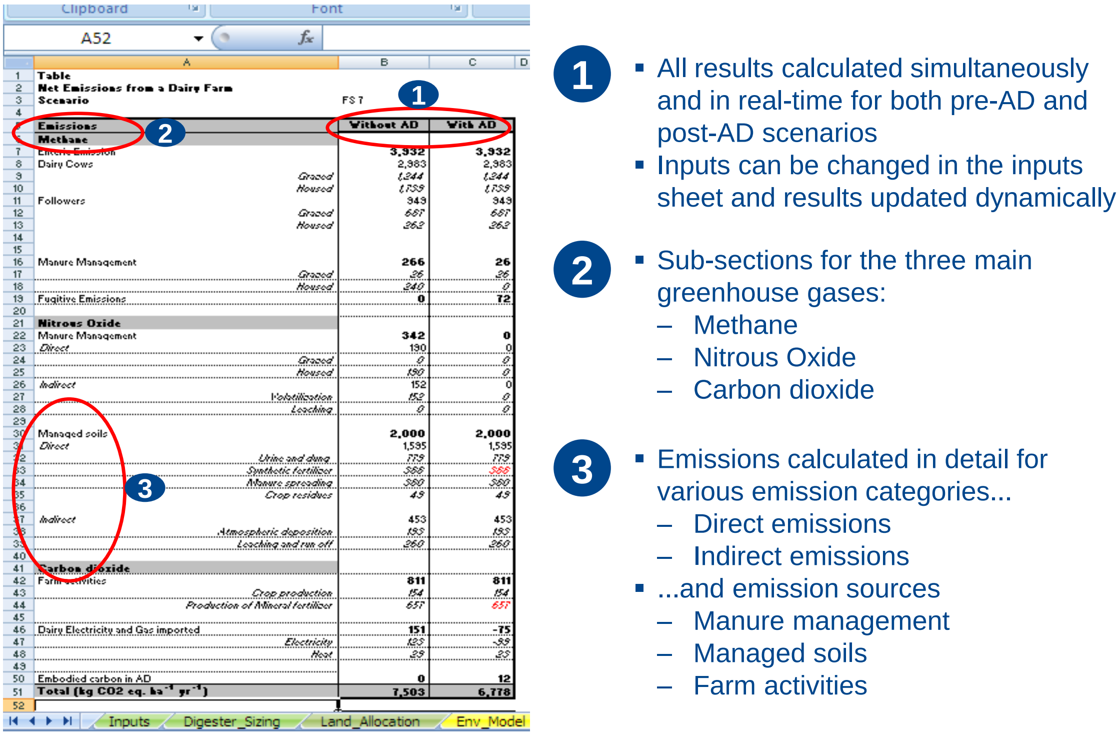


Table			
Net Emissions from a Dairy Farm			
Scenario		FS 7	
Emissions		Without AD	With AD
Methane			
Enteric Emission		3,932	3,932
Dairy Cows		2,983	2,983
	Grazed	1,244	1,244
	Housed	1,739	1,739
Followers		949	949
	Grazed	687	687
	Housed	262	262
Manure Management		266	26
	Grazed	26	26
	Housed	240	0
Fugitive Emissions		0	72
Nitrous Oxide			
Manure Management		342	0
Direct		190	0
	Grazed	0	0
	Housed	190	0
Indirect		152	0
	Volatilization	152	0
	Leaching	0	0
Managed soils		2,000	2,000
Direct		1,595	1,595
	Urine and dung	779	779
	Synthetic fertilizer	386	386
	Manure spreading	380	380
	Crop residues	49	49
Indirect		453	453
	Atmospheric deposition	193	193
	Leaching and run off	260	260
Carbon dioxide			
Farm activities		811	811
	Crop production	154	154
	Production of Mineral fertilizer	657	657
Dairy Electricity and Gas imported		151	-75
	Electricity	125	-99
	Heat	29	23
Embodied carbon in AD		0	12
Total (kg CO2 eq. ha^{-1} yr^{-1})		7,503	6,778

Figure 13 Emissions model results table

This full set of results is produced when any scenario is "run" through the model. The details of sub-sections are also provided in the figure and all results are produced simultaneously for "Pre-AD" and "Post-AD" scenarios for the same set of input parameters.

5. Economic model

The economic model includes analyses of the expenditure and revenue streams of a dairy farm and how these are affected by the introduction of an anaerobic digester. It is assumed that the basic infrastructure required for the functioning of a dairy farm (livestock, land, milking parlour and barn) is owned by the farmer. Detailed descriptions of the methods used for determining farm set up and functioning are presented in Chapter 3. The farm activities are focussed on the production of milk. The expenditure and revenue are calculated on an annual basis. This model primarily considers the revenue streams and expenditures that are affected by the introduction of AD; others like veterinary and medicine costs, water, breeding, etc. are not studied.

5.1 Capital cost of digester

Given the limited uptake of farm waste based digesters in the UK, reliable information on digester costing is scarce. As part of this research a primary task was to develop a methodology to estimate digester capital cost for various different farm sizes and operating conditions. The goal was to have a simple equation that takes in the digester size an input parameter and is able to provide a capital cost estimate. A statistical regression based on available data from literature and quotes from industry participants was used. The capital costs are not adjusted to a base year, assuming inflation compensates for the reduction in technology price due to improvement in technology, increase in number of digesters installed and the lessons learnt from installing and running digesters over a period of time.

Actual Data

Actual UK based capital cost data published by Bywater (2011) and Redman (2010) are used as a primary input. These are supplemented by multiple quotes received from key suppliers in the UK and published by Kottner *et al.* (2008). This aggregate data set is filtered for a set of conditions to arrive at a "cleansed" data set that most closely represents the slurry based digestion on a dairy farm which is the focus of this study. The key filtering criteria and assumptions made in the data cleansing exercise are as follows:

- **Year of construction**: All data points before 1990 are excluded as the inflation and other costs as well as technology have changed significantly and unadjusted prices prior to it are no longer relevant.
- **Farm based digesters**: On analysis of the complete dataset, no clear correlation between size and capital cost of digesters is seen due to the variability in input feedstock and the technology employed. In order to reduce this variability, only data from digesters installed on dairy farms or using slurry as one of the inputs are used, excluding digesters using other feedstock like waste water bio solids plants, organic fraction of municipal solid waste, etc.
- **Farm based digesters accepting food waste**: Based on preliminary analysis, the cost of digesters digesting food waste are found to be very different from that of farm based digesters due to the high cost of pasteurisers, heat requirements of pasteurisation and transport related costs and emissions. Hence, data points that use food waste as a part of the feedstock are excluded.
- **Farm based digesters digesting crop residues**: Due to the scarcity of digesters digesting only dairy slurry, digesters co-digesting slurry with other feed stocks like grass, whey etc. are included in the analysis to have enough empirical cost estimates for statistical analysis.

Quotes

Analysis of the capital cost data made available by Kottner *et al.* (2008) revealed that capital cost of CHP units as well as other site specific costs are included in the quotes.

- **CHP unit**: This research takes CHP and alternative uses of biogas in consideration and deals with these separately from the cost of digesting. Hence, the capital cost of CHP unit is deducted from the total cost of the digester and accounted for separately depending on the scenario.
- **Feedstock storage**: In order to maximise capture of specific methane yield, slurry from dairy cows is not typically stored prior to digesting it. Feedstock storage is more appropriate for crop residues. Hence, the cost of construction of feedstock storage which would not be applicable for slurry based digesters is deducted from the quoted cost of digester.

- **After digestion storage**: Kottner *et al.* (2008) have included capital cost required to construct post-digestion storage capacity. Since January 2012, all dairy farms are required to have a minimum slurry storage capacity of 5 months. Hence, in this analysis it is assumed that slurry storage capacity exists on the farm and the corresponding storage construction capital costs are deducted from the total capital cost data used.
- **Heat distribution systems**: The remote locations of dairy farms in the UK make heat distribution a very unlikely option. The heat produced is likely to be used within the farm and hence, cost of heat distribution system is excluded.

The exclusion of the above costs from the quotes has been possible as the capital cost breakup has been provided in the report, which is not the case with the actual digester costs presented by Bywater (2011). The costs included both those for the digester and the CHP units. In order to derive a cost curve for the digester alone a capital cost curve for CHP units has been developed using the quotes in Kottner *et al.* (2008) as presented in Section 5.2. This curve has been then been used to estimate the cost of CHP unit where installed in the case studies reported by Kottner *et al.* (2008) and the calculated value has been deducted from the total capital cost data to obtain a digester cost estimate. There may be additional site specific costs incurred in the actual digester case studies. These costs have, however, not been broken down and as a result the digester cost curve may suggest a higher cost. The data thus derived is shown in Table 18..

Curve fitting analyses were conducted to estimate the relationship between digester size and capital cost ($CC_{digester}$). These analyses were conducted against quotes only data, actual empirical data and full data set including both quotes and the actual empirical data.

Table 18 Data used for capital cost estimation

Farm	Source	Capital cost (£)	Digester cost (£)	Digester size (m^3)
Tuquoy Farm1	Actual	£80,000	£62,209	75
Corsock Farm	Actual	£160,000	£160,000	80
Hill Farm	Actual	£50,000	£50,000	105
Tuquoy Farm2	Actual	£220,000	£202,209	175
Ryes Farm	Actual	£225,000	£225,000	250
Bank Farm1	Actual	£75,000	£25,522	265
Shropshire Farm	Actual	£45,000	£45,000	300
New Farm	Actual	£250,000	£212,375	320
Walford and North Shropshire college digester	Actual	£135,000	£84,385	330
Castle Farm	Actual	£300,000	£300,000	480
Devon Farm	Actual	£100,000	£42,173	500
Bank Farm2	Actual	£105,000	£55,522	525
Copys green farm	Actual	£750,000	£644,470	870
Lodge Farm	Actual	£650,000	£566,502	1100
Kemble Farm	Actual	£1,200,000	£1,041,948	1480
Site 7 IBBK 1	Quote	£464,489	£383,215	1186
Site 1 IBBK	Quote	£506,921	£337,165	1186
Site 3 IBBK	Quote	£953,176	£477,396	1854
Site 4 IBBK	Quote	£470,054	£372,184	1854
Site 5 IBBK	Quote	£822,122	£642,582	2669
Site 6 IBBK 1	Quote	£876,590	£620,250	2669
Site 2 IBBK	Quote	£1,364,085	£789,930	3707

Best fit curves are provided by power functions (as shown in Figure 14). This agrees with the works of Murphy and Power (2009) and Zglobisz *et al.* (2010) Discussions with Angela Bywater (personal communication 16th August 2012) confirmed that there is substantial scale benefit in the capital cost of an AD installation and a power function would be effective in supporting the hypothesis of lower per unit costs as size of the digester increases.

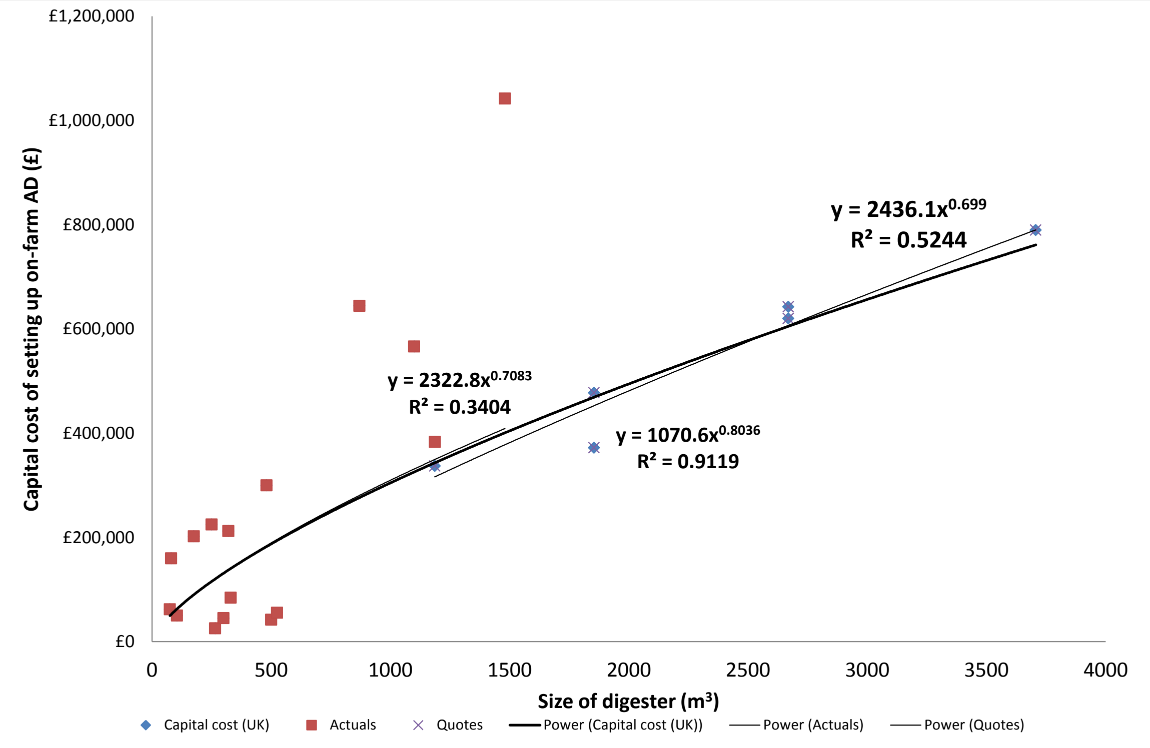


Figure 14 Actual and quoted capital costs of AD

The equation thus obtained has been used for estimation of capital cost.

$CC_{digester}$ = 2436.1 * (digester size)^0.699 [47]

Where $CC_{digester}$ is the capital cost of the digester and additional equipment required, £

Digester size is the size of the digester, m^3

5.2 Capital cost of CHP unit

The capital cost of CHP units is based on the quotes made available by Kottner *et al.* (2008). All the quotes provided for CHP costs in the report are used and these capital cost quotes include the cost of the engine, generator and measuring and control technologies. These data were used to develop a power equation to create a tool for calculating CHP cost for the installed capacity of the CHP unit. The analysis was performed using the same method as for developing the digester capital cost curve. The data used are presented in Table 19.

Table 19 Data used for CHP unit cost estimation

Farm	Source	CHP size (kW)	CHP costs (£)
Site 7 IBBK 1	Quote	75	78,394
Site 1 IBBK	Quote	75	78,394
Site 3 IBBK	Quote	190	121,765
Site 4 IBBK	Quote	104	83,620
Site 5 IBBK	Quote	250	141,140
Site 6 IBBK 1	Quote	250	141,140
Site 2 IBBK	Quote	499	213,008

A power function was fitted to the quotes for CHP units made available in Kottner *et al*. (2008) based on the knowledge of economy of scale and the 'six tenths rule' used widely in the chemical engineering industry. The six tenths rule implies that the cost is proportional to the size/capacity raised to the power 0.6. Power 0.6 (scale up factor) is, however, an approximation. The quotes available for capital cost of CHP have been fitted to a power curve to obtain the value of the scale up factor. The curve obtained is presented in Figure 15.

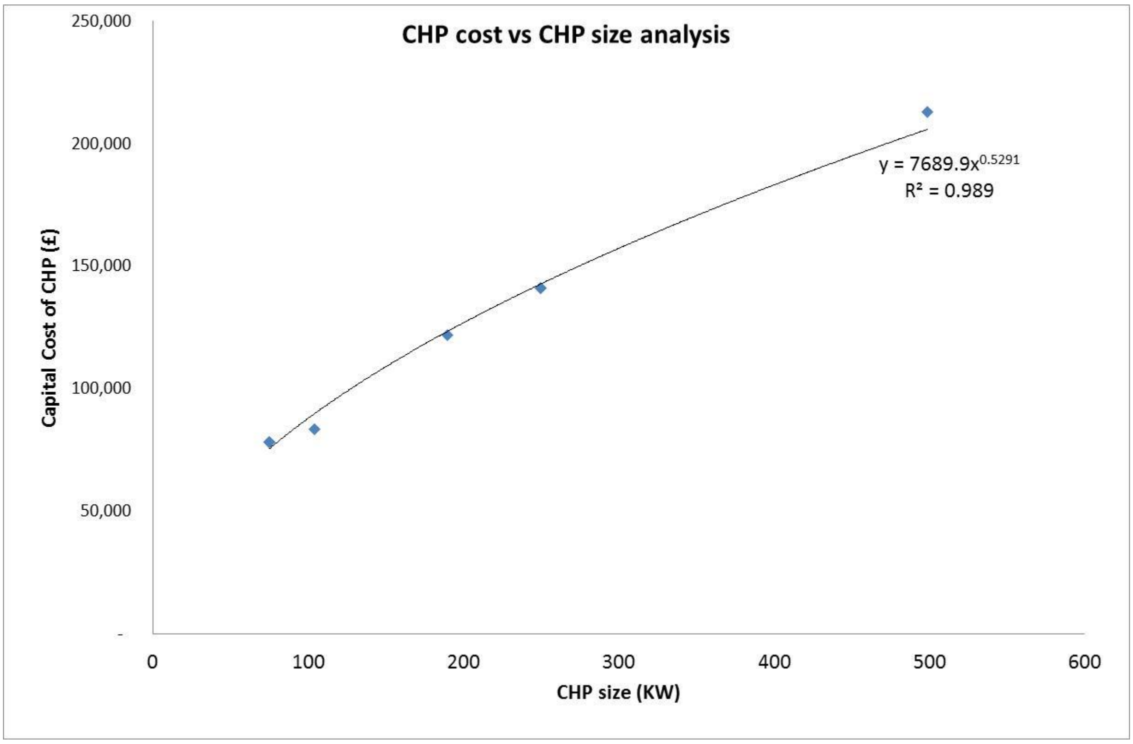


Figure 15 Quotes for capital cost of CHP units

The equation thus obtained was used for estimation of capital cost of CHP.

CC_{CHP} = 7889.9 * (CHP size)^0.5291 [48]

Where CC_{CHP} is the capital cost of CHP unit, £

CHP size is the installed capacity of the CHP installed, kW.

5.3 Capital cost of biogas upgrading equipment

There are currently only three digesters in the UK which upgrade their biogas to bio-methane to be injected into the gas grid (Defra, 2013). Given the limited data available locally, the empirical data set used for statistical regression analysis is based on the estimates available from other European countries and quotes obtained from various vendors employing varied upgrading technologies.

Quotes for capital cost of upgrading equipment reported and actual data from France, Sweden and the Netherlands compiled in Persson and Hogskola (2003) and Kaparaju *et al.* (2012) are used along with quotes received from current suppliers of the technology in Europe, Gastreatment Services BV (GPP2T, GPP4T), HAASE Energietechnik GmbH (BiogasUpgrader BGV 250), Malmberg Ltd. and DMT Environmental Technology.

Data for large scale landfill upgrading plants has not been included in the analysis as the scale is not appropriate.

The data used for estimation is presented in Table 20.

Table 20 Data used for upgrading equipment cost estimation

Capacity (m³ hour⁻¹)	Original Quote	Capital cost (£)	Provider	Data type
120	€ 1,200,000	960,000	Gas Treatment Services	Quote
280	€ 1,400,000	1,120,000	Gas Treatment Services	Quote
250	€ 1,000,000	800,000	Haase	Quote
250	£900,000	900,000	Malmberg	Quote
50	£400,000	400,000	DMT Carborex	Quote
100	£500,000	500,000	DMT Carborex	Quote
150	£600,000	600,000	DMT Carborex	Quote
200	£700,000	700,000	DMT Carborex	Quote
250	9,000,000 kr	828,000	SGC - quote	Quote
150	4,900,000 kr	450,800	SGC - quote	Quote
100	4,300,000 kr	395,600	SGC - quote	Quote
300	7,500,000 kr	690,000	SGC - quote	Quote
300	9,500,000 kr	874,000	SGC - quote	Quote
250	€ 1,952,840	1,562,272	Zeven, Germany	Actual
200	6,700,000 kr	616,400	Actual - Lille, France	Actual
200	3,500,000 kr	322,000	Actual - Linkoping, Sweden	Actual
600	€ 1,925,850	1,540,680	MT Biomethan GmbH	Quote
17	2,500,000 kr	230,000	Biogas Ost - Plonninge biogas plant, Sweden	Quote

The regression analysis on the dataset is similar to the analysis described in the prior two sections. This analysis led to a simple equation that allows estimation of biogas upgrading equipment capital cost based on the unit's size requirement.

The data used is presented in Figure 16.

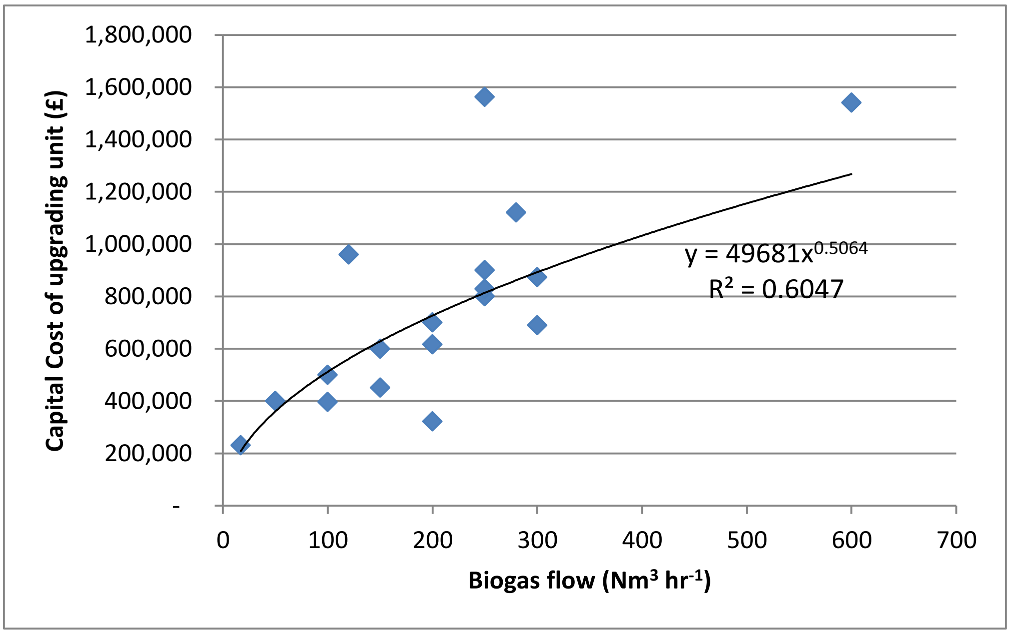


Figure 16 Actual and quotes for capital cost of upgrading plants

A power function was fitted to the data available and the equation hence obtained is used for estimation of biogas upgrading costs.

$CC_{upgrade}$ = 49,681 * (FlowRate)^0.5064 [49]

Where $CC_{upgrade}$ is the capital cost of upgrading unit, £

FlowRate is the installed capacity of the upgrading unit, Nm^3 $hour^{-1}$.

5.4 Mortgage calculation

The capital and installation costs of the digester and the CHP are assumed to be financed via a mortgage. A set-up fee of 1% of the capital cost charged by the bank for the processing of the loan has been added to the mortgage amount. The annual mortgage payment is calculated assuming a fixed rate mortgage. A fixed rate mortgage payment is an industry standard and is assumed for the regularity in monthly/annual budgeting.

M = (r*P)/(1-((1+r)^-N) [50]

Where M is a fixed annual payment, £ $year^{-1}$

r is the annual interest rate, expressed as decimal

N is the number of annual payments

P is the capital borrowed, £ (=1.01 * ($CC_{digester}$ + CC_{CHP} /$CC_{upgrade}$)).

Farm loans are expected to attract an annual percentage rate (APR) of 3-3.5% above the base rate (Nix, 2012). The base rate at the time of writing is exceptionally low at 0.5%. Hence, a 10 year average (30/9/2002 – 28/08/2012) of 3.08% is used (Bank of England, 2012). A mortgage rate on the investment required to set up an AD plant is, therefore, assumed at 6.5% over a period of 20 years.

5.5 Operating costs of AD

The annual operating cost (OC) of a digester is 7% of the capital cost of the digester and includes labour (2% of capital cost), maintenance and repair (3.5% of capital cost), and insurance (1.5% of capital cost) based on Kottner *et al.* (2008) and Redman (2010). The maintenance cost of CHP are estimated at 1 pence (p) for every kWh of electricity produced (Kottner *et al.*, 2008). The operating costs are assumed to remain constant year on year for the lifetime of the digester and are presented in Figure 17.

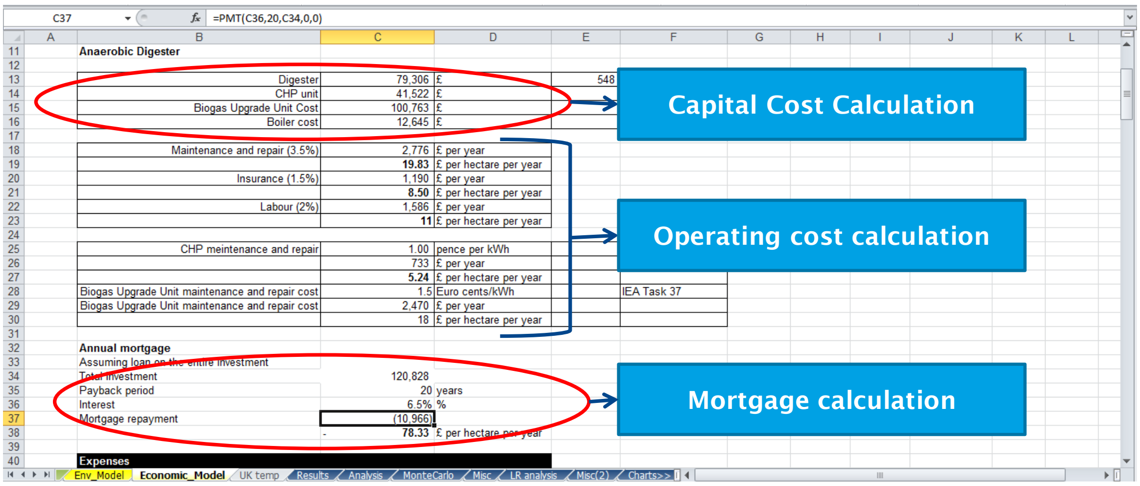


Figure 17 Operating costs and mortgage calculation module

5.6 Labour

There are three main areas where labour is required on a farm, namely, dairy, crop production and digester.

5.6.1 Dairy

The cost of labour for running a dairy ($Labour_{dairy}$) is calculated based on the labour requirement for the herd. It is assumed that 28 hours of labour is required per year for a dairy cow while a follower requires 2.9 hours $month^{-1}$ in summer and 1.2 hours $month^{-1}$ in winter (Nix, 2007). The hourly labour rate is based on the hourly rate of £9.4 $hour^{-1}$ for an Agricultural Grade 6 Worker responsible for the management of the farm (UK Government, 2013).

5.6.2 Crop production

The total cost to the farmer of crop production (grass silage and winter wheat) ($Labour_{crop}$) includes the labour required and is calculated assuming that a contractor is hired for end-to-end production of crops. The costs incurred are estimated based on the cropped area using Nix (2007) and include labour, machinery, fuel and repair costs and depreciation. The £233 ha^{-1} $year^{-1}$ for winter wheat production includes ploughing, cultivation, drilling, spraying, fertiliser application, combining, carting grain, barn work and crop drying. The production of grass silage includes ploughing, seedbed harrowing, loading, carting and application of fertilisers, drilling, rolling, topping, turning, carting and ensiling of grass and costs the farmer £271 ha^{-1} $year^{-1}$.

5.6.3 Digester

The additional labour cost for running a digester ($Labour_{digester}$) is included in the operating costs of AD presented in Section 5.5 and is considered to be 2% of the capital cost of the digester based on Kottner *et al.* (2008).

5.7 Electricity

In the absence of a digester and CHP producing electricity, the farmer would be importing electricity for farming and dairy use. The price for this imported electricity ($Price_{imported}$) is assumed to be 10.22 pence kWh^{-1} based on the average for year 2011(DECC, 2012f).

When there is an on-farm digester producing electricity via a CHP unit, if no subsidy is taken from the government, an export rate is negotiated with the electricity supplier. This price is assumed to be 5.5 pence kWh^{-1}, based on the wholesale price of electricity (Kottner *et al.*, 2008).

The UK government has recently set a feed-in-tariff (FIT) structure that compensates the producers (farmers in this case) for electricity production using renewable energy technologies like hydro, solar and wind. Under the current subsidy structure, a generation FIT is available to the farmer for every unit of renewable electricity generated after accounting for parasitic load. An additional export tariff is given for every unit of electricity that is exported to the grid. The value of FIT (01/06/2012) used for calculation of revenue have been listed in Table 21 (DECC, 2012d). For future years, an increase in FIT in-line with long term average RPI (3% based on the average of last 10 years (Nix, 2012)) is assumed.

Table 21 Current feed in tariff for Anaerobic Digestion

Type and size of plant	Tariff (p kWh^{-1})
Generation tariff ($FIT_{generation}$):	
<= 250 kW	14.7
>250 – 500 kW	13.6
>500 kW	9.9
Export tariff for all levels ($FIT_{exported}$)	3

The potential revenue from the generated electricity, $Profit_{electricity}$ in £ $year^{-1}$ is calculated

$$Profit_{electricity} = ((E_{CHP} - E_{parasitic}) * FIT_{generation}) + (E_{exported} * (FIT_{exported} \text{ or } Price_{exported})) - (E_{imported} * Price_{imported}) \quad [51]$$

Where $E_{exported}$ is the electricity exported from farm, kWh (calculated)

$E_{imported}$ is the electricity imported on farm, kWh (calculated).

5.8 Heat

In the absence of a digester, all heat requirements of the dairy are assumed to be met by importing liquefied petroleum gas (LPG) and using it in a boiler. The choice of LPG as fuel has been discussed in Section 4.9. The current average price of LPG ($Price_{heat,retail}$) in the UK (as of 21/10/12) is 74.71p litre^{-1} (Whatgas, 2012) or 11.53 pence kWh^{-1}.

In the presence of an on-farm digester with a CHP unit, the heat produced is used to meet parasitic load of the digester. In case of surplus heat, it is used by the dairy in the form of hot water for washing. Export of heat in the UK is very site specific and due to the rural location of most digesters not feasible.

In the absence of government subsidy, it is assumed that the price of heat exported is assumed to be zero. The government provides a renewable heat incentive (RHI) of 7.1 pence kWh^{-1} (01/06/2012) that is available for the heat that is generated on farm and is put to an eligible use as outlined in Section 2.3.10.2. The revenue stream generated by the use of heat produced on-farm and the avoided cost of heat import ($Profit_{heat}$) is calculated.

$$Profit_{heat} = (H_{CHP} - H_{parasitic} * RHI) - (H_{imported} * Price_{heat,retail}) \qquad [52]$$

Where $H_{imported}$ is the heat imported, kWh year-1.

The calculation of profit generated from production of heat and electricity is presented in Figure 18.

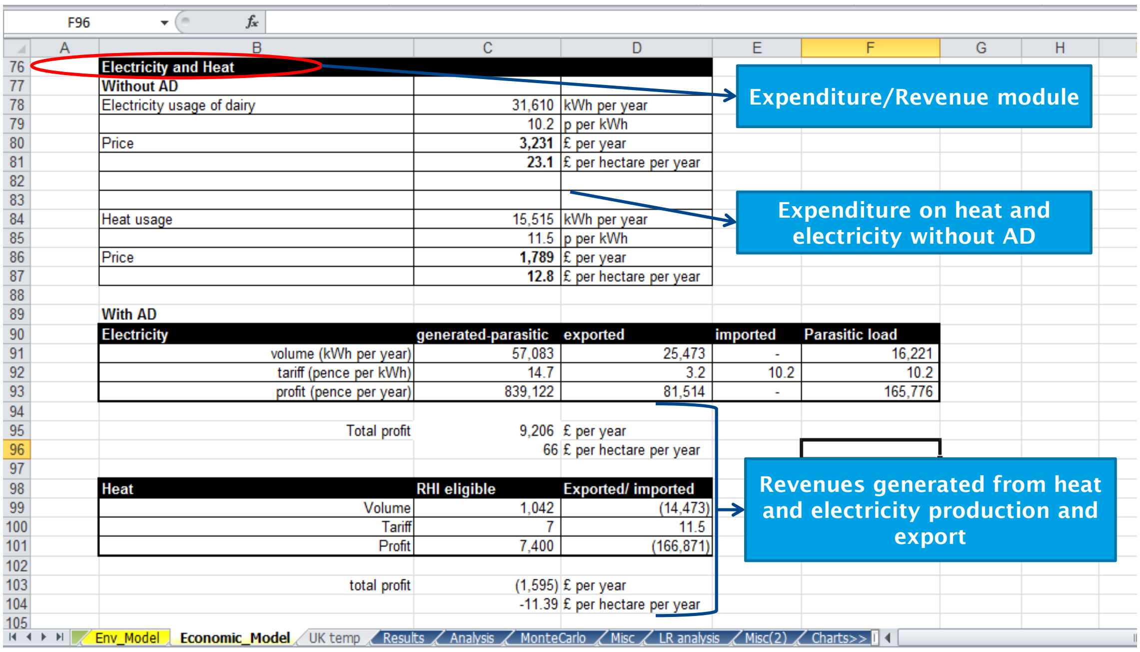


Figure 18 Module calculating revenue generated from heat and electricity production

5.9 Feedstock material for the cattle

Grass: All of the cattle on the farm are assumed to be either in-house, in which case they are fed grass silage produced and stored on the farm, or in the fields in which case they are assumed to be grazing on-grass. The costs of grass silage production have been calculated based on the cost to farmer including labour, tractor, machinery, fuel usage, repairs, and depreciation as per Nix (2007) and have been included under Labour in Section 5.6. Since the grass silage is both produced and used within the farm, it is not considered as a revenue stream.

Winter Wheat: The total concentrate requirement per cow for milk production has been calculated using the estimates available in Nix (2012). Winter wheat is produced on farm to be fed to the dairy cows as concentrate, with the balance being bought from commercial suppliers. This assumption is based on standard practice of UK farmers. The method of calculating the cost of wheat production on farm is similar to that for grass and based on Nix (2007) and is included in Section 3.2.6. The price of imported feed wheat is taken as a 5 year average (for the period March 2006 - March 2011) in order to account for the volatility in the market and the seasonal variation in the price of wheat.

The total cost of imported concentrates:

$$Exp_{Concentrates} = ((CR * head) - (Yield_{ww} * Area_{ww})) * Price_{ww} \qquad [53]$$

Where $Exp_{Concentrates}$ is the annual spending on buying concentrates

CR is the concentrate requirement of a dairy cow for a given milk yield, tonnes year^{-1} (1 tonne cow^{-1} year^{-1} (Nix, 2012))

Head is the number of dairy cows in the herd

$Yield_{ww}$ is the annual yield of winter wheat, tonnes ha^{-1} (8.5 tonnes ha^{-1} (Jackson *et al.*, 2008))

$Area_{ww}$ is the area of winter wheat grown, hectares

$Price_{ww}$ is the price of winter wheat in the UK (£125 tonne^{-1} (March 2006-2011 (Dairyco, 2012c))).

Details of farm area, including the relative proportions of grazed and silage grass and wheat are given in Section 3.2.

5.10 Fertilisers

The total amount of fertilisers required is discussed in Section 3.4. At 98.6 p kg^{-1} N, 94.6 p kg^{-1} P_2O_5 and 58.3 p kg^{-1} K_2O (Nix, 2012), the expenditure on buying fertilisers ($Exp_{fertiliser}$) is calculated based on the quantity of each fertiliser required.

5.11 Milk

It is assumed that the dairy farm sells all the milk collected from the dairy cows as milk and none is processed into other dairy products like butter or cheese. The profit made from selling the milk ($Profit_{milk}$) is based on a 5 year average (January 2007 to December 2011) of the farm-gate price paid to the farmer which is 24.46 p litre^{-1} (Dairyco, 2012d). This is in order to account for the volatility in the price of milk as shown in Figure 19.

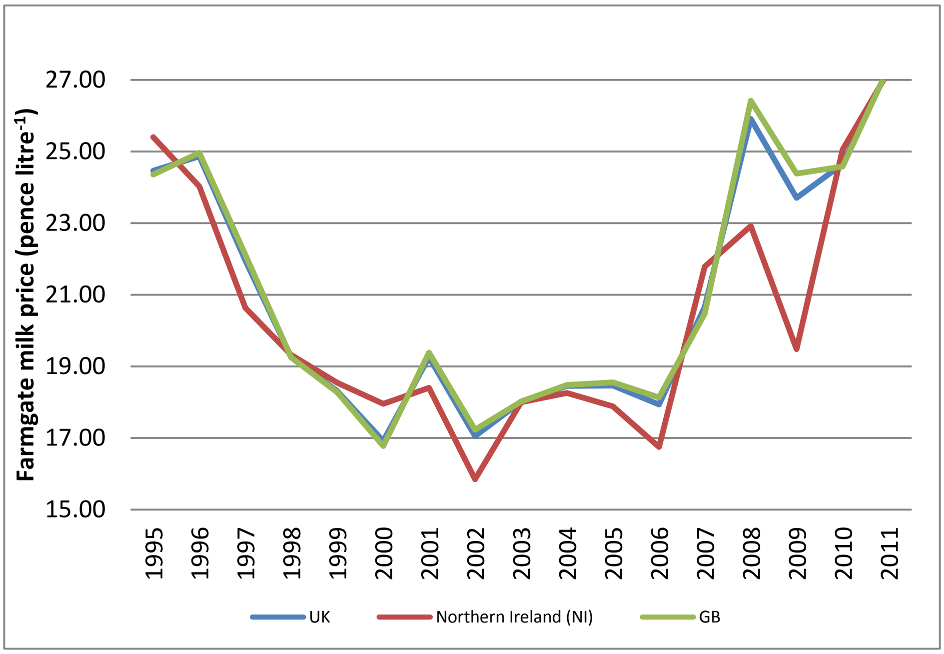


Figure 19 Historical farm-gate price of milk (adapted from (Dairyco, 2012d))

5.12 Total Profit

The total profit from the operation of the dairy farm is calculated by deducting all the expenditures in the form of mortgage payment, operating costs and labour from the revenues generated from sale of electricity, heat and milk, based on the above modules.

$$\text{Total profit} = \text{Profit}_{electricity} + \text{Profit}_{heat} + \text{Profit}_{milk} - M - OC - (\text{Labour}_{dairy} + \text{Labour}_{crop} + \text{Labour}_{digester}) - \text{Exp}_{concentrates} - \text{Exp}_{fertiliser} \quad [54]$$

The full set of results presented in Figure 20 is produced when any scenario is "run" through the model. The details of sub-sections are also provided in the figure and all results are produced simultaneously for "Pre-AD" and "Post-AD" scenarios for the same set of input parameters.

	A	B	C
52			
53	Farm financial data	Pre-AD	Post-AD
54	Running expenses		
55	Mortgage payment	0	71
56	Seeds	23	23
57	Fertilizer	119	119
58	Feed (wheat, grass)	352	352
59	Concentrates bought	-68	-68
60	Bedding	19	19
61	Vet and medicine	48	48
62	Water	29	29
63	Labour		
64	Crops	139	139
65	Dairy	387	387
66	Anaerobic Digester	0	10
67	Maintenance and repair		
68	Digester	0	18
69	CHP	0	5
70	Biogas Upgrade	0	0
71	Digester Insurance	0	8
72	Total	1,047	1,160
73	Value of Produce		
74	Biogas/Boiler	0	0
75	Electricity	-23	66
76	Heat	-13	-9
77	Wheat	242	242
78	Straw	28	28
79	Silage	110	110
80	Milk	1,877	1,877
81	Total	2,222	2,315
82	Profit (£ per year per hectare)	1,175	1,154

1
- All results calculated simultaneously and in real-time for both pre-AD and post-AD scenarios
- Inputs can be changed in the inputs sheet and results updated dynamically

2
- Sub-sections for key components of financials:
  - Running expenses
  - Maintenance & repair of AD related equipment
  - Value of produce (including electricity and heat)

3
- Separate calculation for key components of revenue from AD..
  - Electricity
  - Heat
- ...and farm related activities
  - Milk production
  - Wheat
  - Silage etc

Figure 20 Economic model results table

5.12.1 Net Present Value

The net present value helps in understanding the overall impact of the financial life cycle of a project. NPV calculation takes the future income (accounting for both projected costs as well as revenues) and discounts it into today's value. Discounting the future income into today's value for all years of the project lifecycle provides us with a net position for the lifecycle economics of the project.

Discounting future cash flows to present day terms requires calculation of an appropriate discount rate. The discount rate calculation should account for the inherent risk in achieving future cash flows or in other terms, the project risk. The higher the project risk, the higher the discount rate will be. When the future cash flow is discounted with a higher discount rate, the present value is correspondingly less.

It may be noted that the discount rate used to calculate NPV does not represent the financing costs that an investor or a bank may charge to provide capital for the project. The cost of financing is dependent not only on the collateral, borrower's credit history, investor's alternatives, other incentives from government or interested parties etc.

Economic model

Discount rate for NPV calculation accounts only for the project's operational or execution risks.

5.12.1.1 Discount rate calculation

The discount rate is calculated using the capital asset pricing model (CAPM):

$r = Rf + \beta i(E(Rm) - Rf)$ [55]

Where r is the expected return on capital asset or discount rate

Rf is the risk-free rate of interest, 2.1% (Bank of England – 10 year nominal)

βi is the sensitivity of the asset returns or the beta coefficient, 1.23 (Zglobisz *et al.*, 2010)

E(Rm)-Rf is the market (risk premium, 4.91%, UK specific (Zglobisz *et al.*, 2010).

5.12.1.2 Net Present value (NPV) calculation

The NPV is calculated using the following equation.

$NPV = -C_0 + \Sigma (C_i/((1+r)^{\wedge}i))$ [56]

Where C_0 is the total investment made at time i=0

Ci is the cash flow at time i

i is the time varying from 1 to 20 years

The module developed is presented in Figure 21.

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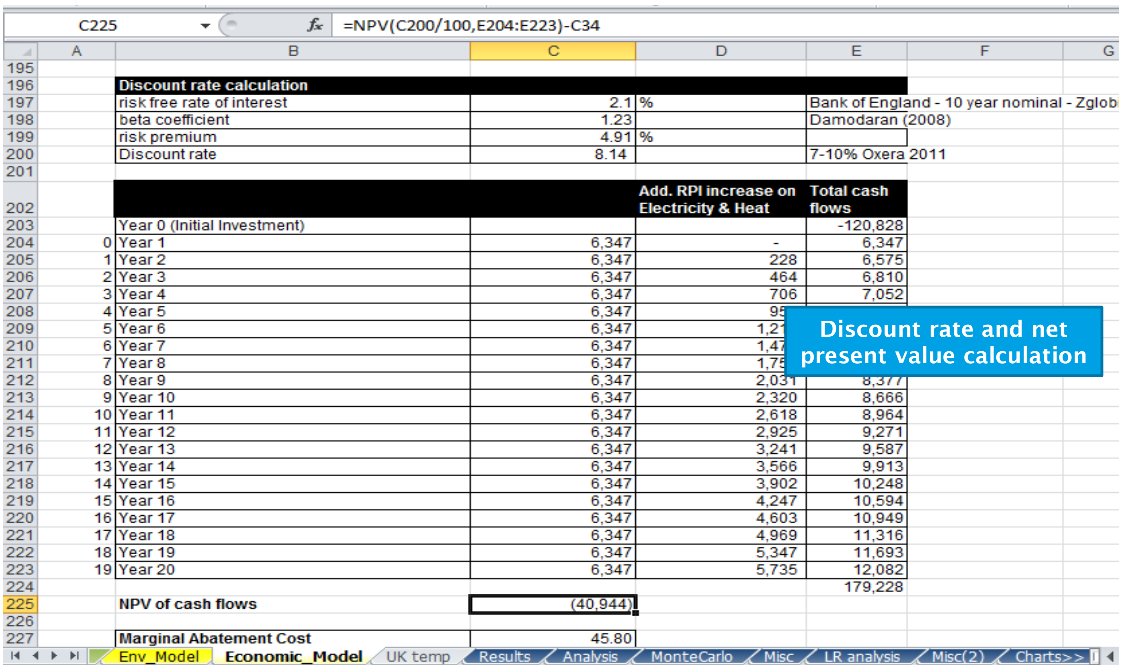

C225 =NPV(C200/100,E204:E223)-C34

	A	B	C	D	E
195					
196		**Discount rate calculation**			
197		risk free rate of interest	2.1	%	Bank of England - 10 year nominal - Zglobi
198		beta coefficient	1.23		Damodaran (2008)
199		risk premium	4.91	%	
200		Discount rate	8.14		7-10% Oxera 2011
201					
202				**Add. RPI increase on Electricity & Heat**	**Total cash flows**
203		Year 0 (Initial Investment)			-120,828
204	0	Year 1	6,347	-	6,347
205	1	Year 2	6,347	228	6,575
206	2	Year 3	6,347	464	6,810
207	3	Year 4	6,347	706	7,052
208	4	Year 5	6,347	[illegible]	[illegible]
209	5	Year 6	6,347	[illegible]	[illegible]
210	6	Year 7	6,347	[illegible]	[illegible]
211	7	Year 8	6,347	[illegible]	[illegible]
212	8	Year 9	6,347	2,031	8,377
213	9	Year 10	6,347	2,320	8,666
214	10	Year 11	6,347	2,618	8,964
215	11	Year 12	6,347	2,925	9,271
216	12	Year 13	6,347	3,241	9,587
217	13	Year 14	6,347	3,566	9,913
218	14	Year 15	6,347	3,902	10,248
219	15	Year 16	6,347	4,247	10,594
220	16	Year 17	6,347	4,603	10,949
221	17	Year 18	6,347	4,969	11,316
222	18	Year 19	6,347	5,347	11,693
223	19	Year 20	6,347	5,735	12,082
224					179,228
225		**NPV of cash flows**	(40,944)		
226					
227		**Marginal Abatement Cost**	45.80		

Figure 21 Module calculating the discount rate and net present value of the digester

NPV > 0 would imply a potentially profitable project while NPV < 0 implies a loss making project. At NPV = 0, the project breaks even financially.

5.12.2 Payback period

Payback period (PP, years) is defined as the first year in which the initial investment is equal to cumulative undiscounted operating cash flows:

$$PP = C_0/(\Delta profit + M) \qquad [57]$$

Where Δprofit is the change in profit from introduction of anaerobic digestion

5.12.3 Internal rate of return

Internal rate of return is defined as the discount rate at which the NPV of the project is 0. In algebraic terms, it is the discount rate r which solves for the following equation:

$$C_0 = \Sigma\, C_i/(1+r)^i \qquad [58]$$

Where Ci is the cash flow at time i

i is the time varying from 1 to 20 years

Economic model

For the purpose of this research, excel functionality that automatically calculates IRR for a series of cash flows was used.

The development of the emission and the economic models lays the foundation for assessing the potential of anaerobic digestion for GHG abatement and the cost at which this abatement is achieved. These models are combined to obtain a MAC for any given run and are used for analyses of MAC under varying farming and operating conditions as presented below.

6. Marginal abatement cost

Marginal abatement cost calculation brings together and links the results from the emissions model to the economic model and the impact of introduction of the digester on each of these. Both the models are run for a particular farm set up with and without a digester using appropriate modules and input values. The marginal abatement cost of GHG emissions using slurry based anaerobic digestion for that farm set-up is calculated from the change in emissions by the introduction of anaerobic digestion and the change in profit.

$$MAC = \Delta profit / \Delta emissions \qquad [59]$$

Where MAC is the marginal abatement cost, £ $tonne^{-1}$ CO_2 eq. abated

Δprofit is the change/loss in profit, £ ha^{-1} $year^{-1}$ (calculated from results obtained from Section 5.12 for any farm with and without a digester)

Δemissions is the change in emissions, tonne CO_2 eq. abated ha^{-1} $year^{-1}$ (calculated from results obtained from Section 4.10 for any farm with and without a digester)

The MAC obtained is used for evaluation of AD as an abatement technology and the formulation of GHG abatement policy.

6.1 Modelled farm

The initial modelling was conducted on an average dairy farm in England based on data published by Defra (2011a). The farm is comprised of 140 hectares (ha) of land. The livestock density has been assumed to be 1.6 LU ha^{-1} based on NVZ regulations. Based on common agricultural practices, it has been assumed that the cows are fed on home grown grass silage when they are housed. Winter wheat is fed to the cows as concentrate and any shortfall is compensated by imported feed wheat. The cows are housed for 60% and the followers for 30% of the year. This assumes that the dairy cows are fully housed for 6 months of the year when the weather is cold (October – March) and spend 5 hours a day indoors during milking during the grazing period. The only produce of the farm that is sold is milk and, upon the introduction of AD, heat, electricity and milk. The model is based on a pre-existing functional farm. Hence no change in land use has been assumed.

6.2 Sensitivity Analysis

Relatively small changes in some variables, such as specific methane yield, can have a large impact on the emissions as well as economic model outputs. The effect of various sample farm setups is studied as part of this research. The interplay of multiple variables, however, makes it challenging to draw meaningful conclusions. To overcome this obstacle as well as in order to better understand the relationship of a particular input parameter with the output variables, a detailed sensitivity analysis has been conducted.

The base case for each of these sensitivity scenarios is the "Modelled farm". On the base case, multiple synthetic farm setups are created by changing the value of only a single input parameter. For major input parameters, typically 10 scenarios are created and the variation in the selected input parameter from scenario to scenario is kept equal. Each interval is kept as $1/9^{th}$ of the expected range of that input parameter.

The range of a parameter is based on general practices and literature values. In particular, the range of values for FIT and RHI analysis are based on the current incentives available from the government for renewable technologies. The farm size analysis is based on the average herd size distribution data made available by Dairyco (2012e). The total solids and organic loading rate ranges are based on Nijaguna (2002). Range of values for specific methane yield was based on the literature review as presented in Section 2.2.3.2.1 while that of livestock density on NVZ regulations. Fugitive emissions were analysed for the entire range of values possible while housing was analysed for most expected range of housing expected in the UK which includes winter housing as well as time spent indoors for milking.

The model allows this range to be changed and the sensitivities to be re-run in "real time". A similar analysis is conducted for other key variables as shown in Table 22. The sensitivity analysis has been conducted by building a sensitivity module. This module allows the range of values for the input parameter to be changed for further research and analysis.

Table 22 Variables for sensitivity analysis

Variable	Minimum value	Maximum value
FIT (pence kWh^{-1})	0	30
Farm size (ha)	50	250
Specific methane yield (m^3 g^{-1} VS added)	0.13	0.15
Housing (% of year)	60	100
Organic loading rate (kg VS m^{-3} day^{-1})	2.5	3.5
Livestock density (LU ha^{-1})	1	1.7
RHI (pence kWh^{-1})	0	30
Fugitive emissions (%)	0	100
Total Solids (%)	7	9

Figure 22 below provides the snapshot of the key input module and

Figure 23 provides a description of the modules highlighted in Figure 22.

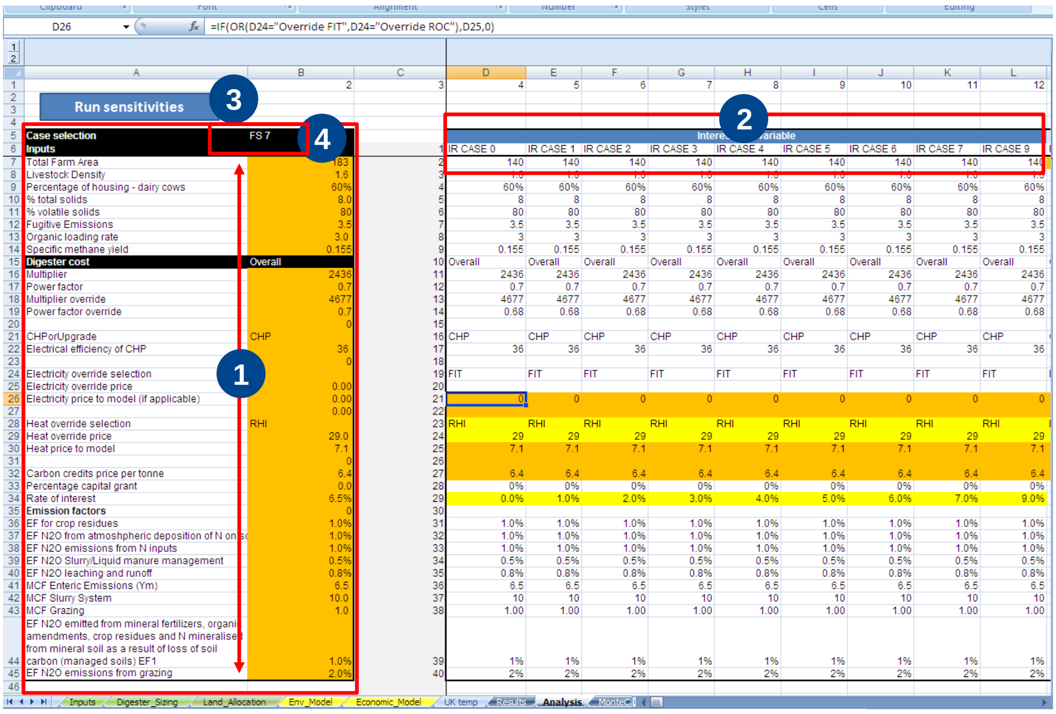

Figure 22 Sensitivity module overall structure (1/2)

	Comments
1 List of input variables	▪ A list of key input variables defined whose values can be changed for each individual scenario. Results are calculated on-the-fly for the scenario selected. ▪ List of input variables can be easily enhanced with limited additional modelling effort
2 Pre-defined scenarios	▪ ~300 pre defined scenarios (farm setups) used in developing sensitivity analysis and for one-off alternative scenarios (Hillsborough case study comparison) ▪ Additional scenarios can be added with minimal effort. No practical limitation on how many scenarios can be tested
3 One-click scenario calculations	▪ One click that will refresh the results for all the 300 scenarios for analysis. ▪ Changes to scenarios can be made once and then a single click will run through each scenario and refresh results
4 Single scenario quick check	▪ Any individual scenario can be selected and tested for variations to input variables ▪ Results will updated in "real time" for checking hypothesis

Figure 23 Sensitivity module overall structure (2/2)

A list of input variables is provided in Figure 24. This is the list that has been used for the purposes of the current research but as described above, this list can be augmented with minimal further "modelling" effort if a new input parameter needs to be introduced.

Inputs	
Total Farm Area	183
Livestock Density	1.6
Percentage of housing - dairy cows	60%
% total solids	8.0
% volatile solids	80
Fugitive Emissions	3.5
Organic loading rate	3.0
Specific methane yield	0.155
Digester cost	Overall
Multiplier	2436
Power factor	0.7
Multiplier override	4677
Power factor override	0.7
	0
CHPorUpgrade	CHP
Electrical efficiency of CHP	36
	0
Electricity override selection	FIT
Electricity override price	0.00
Electricity price to model (if applicable)	0.00
	0.00
Heat override selection	RHI
Heat override price	29.0
Heat price to model	7.1

Figure 24 Sensitivity module input variables full list

6.3 Monte Carlo Analysis

The individual nature of farming practices and farm sizes lead to different MAC values. In order to better understand the profile of MAC values for various UK farms, Monte Carlo analysis was conducted.

Monte Carlo simulations are a particularly useful tool for simulating systems with many degrees of freedom. In this case, key input parameters that have a significant impact on the output variables are considered to be the relevant degrees of freedom and were identified as farm size, maximum methane yield, housing percentage, organic loading rate and livestock density based on the sensitivity analyses.

To generate the values for the Monte Carlo simulation, a Macro was coded in Excel and the code for this is given in Appendix 1. This code generated 5 distinct values in the range of values for the respective input parameter with differences between subsequent values kept identical. The base scenario was assumed to be the “Modelled farm” and all other input parameters were kept identical to the “Modelled farm” setup. The 5 identified input parameters were varied simultaneously. This process led to a generation of 5x5x5x5x5 = 3,125 distinct scenarios and the “Macro” created output for both the environmental and the economic model for each of these scenarios. All of these computations were done for both pre-AD and post-AD setups corresponding to the scenario parameters.

The results of these 3,125 scenarios synthetically represent a large population of potential UK farm setups as the range considered for each of the five input parameters was based on literature and UK related research. The statistical analysis of these 3,125 cases provides insights into the mean and expected behaviour of both the emission and economic related output variables. For the purpose of this research, a detailed analysis of GHG abated, MAC and NPV results was conducted by drawing frequency distributions of results, identifying boundary conditions which provide maximum and minimum values.

The Monte Carlo module has been coded to run through all the combinations of inputs parameters within the expected range as described earlier. The module allows for this analysis to be repeated for a revised set of values.

Marginal abatement cost

Figure 25 provides further details on which input parameters can be changed and where the range of values can be entered while Figure 26 presents the results of the Monte Carlo simulation.

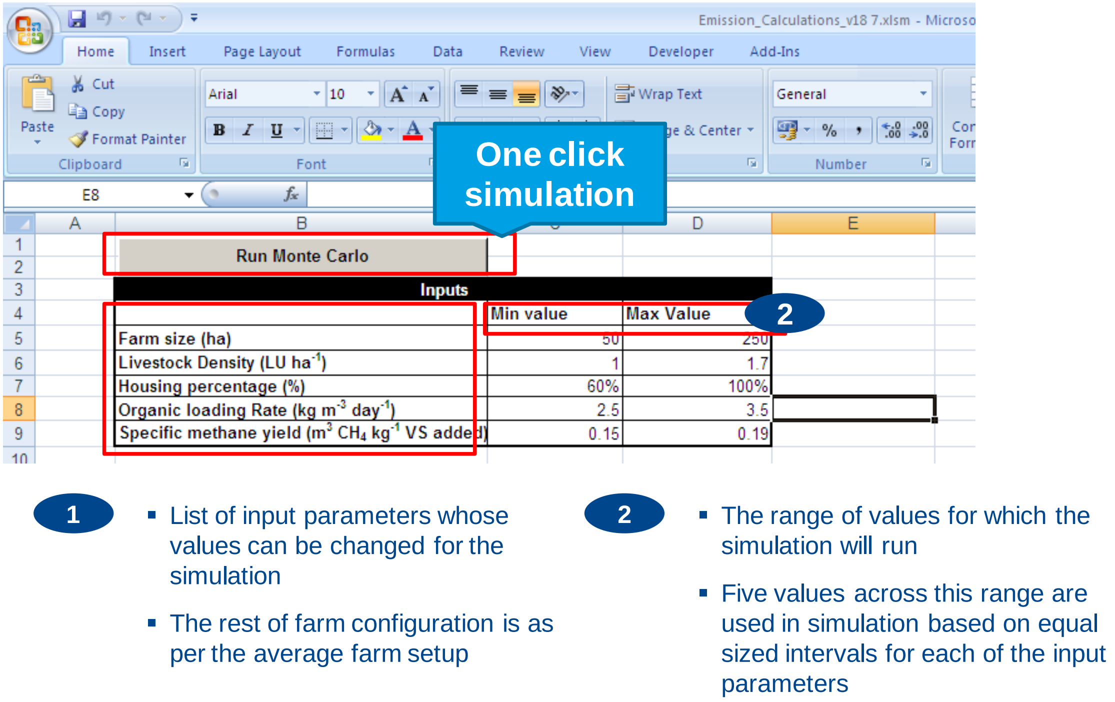

Figure 25 Monte Carlo analyses module (Input section)

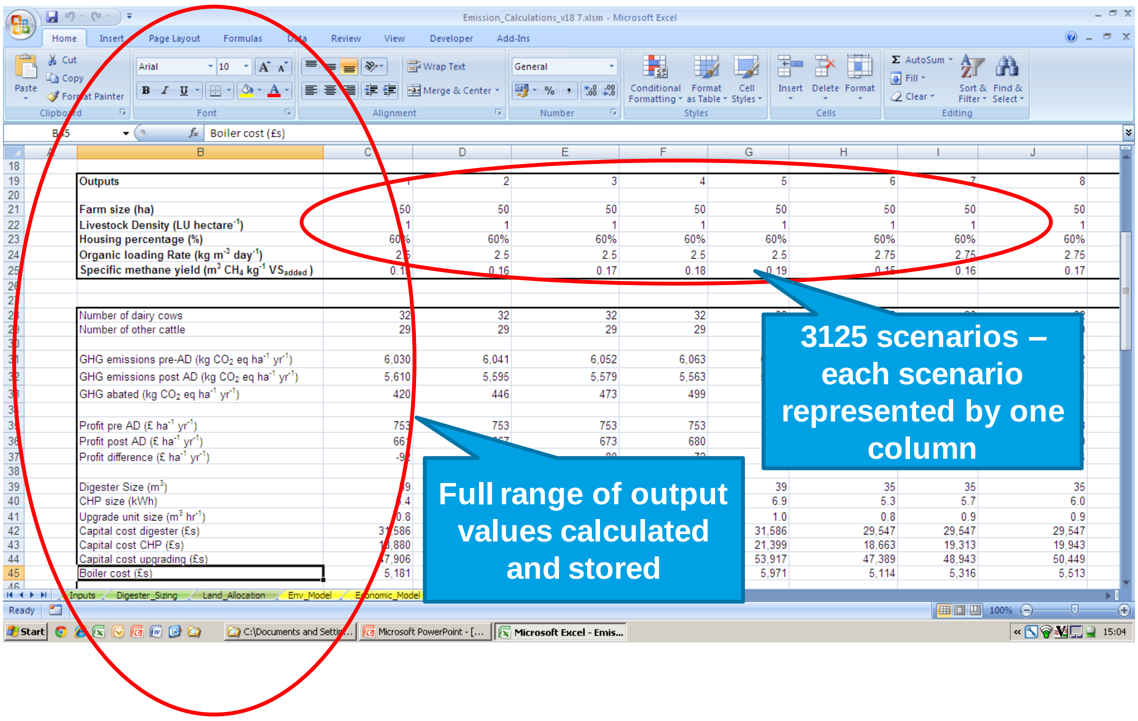

Figure 26 Monte Carlo analyses module (Output section)

7. Farm model results

7.1 Herd size

Based on the area of the "Modelled farm" of 140 hectares and a livestock density of 1.6 LU ha^{-1}, the number of dairy cows is calculated to be 145 and followers, 131. The number of cattle is a primary factor that drives the milk production and hence the economics of the dairy farm. The herd size also effects the feed requirements and hence the land allocation on the farm. In a post-AD scenario, herd size determines the amount of slurry produced which drives the size of digester and CHP units.

7.2 Land allocation

The net energy requirement of grazing dairy cows is 123.8 MJ day^{-1} while that of housed ones is 115.3 MJ day^{-1}. The net energy requirement of grazing followers is 39.6 MJ day^{-1} while those housed need 34.8 MJ day^{-1}. The higher requirement of the grazing cattle as compared to housed cattle is from the additional energy spent in walking to and from the milking parlour and additional activity of grazing.

The cropping area is divided into 3 parts: 37 ha winter wheat; 40 ha for grass silage and 63 ha permanent pasture as per the methods detailed in Section3.2.

7.3 Manure management

The total slurry that is collected and managed in a slurry tank or an anaerobic digester is 2,253 tonnes $year^{-1}$. Additional 2,458 tonnes $year^{-1}$ manure is deposited on the pasture by grazing dairy cows and followers.

7.4 Mineral fertiliser requirement

The manure deposited by the grazing dairy cows and followers is sufficient to meet the phosphorus (P_2O_5) and potassium (K_2O) of the pasture. There is, however, a shortfall of 3,907 kg N $year^{-1}$ (28 kg N ha^{-1} $year^{-1}$) which is made up by mineral fertiliser application. The collected slurry is first applied to grass grown for silage, whose nitrogen needs are completely met. There is, however, an additional requirement for 121 kg P_2O_5 year^{-1} (3 kg P_2O_5 ha^{-1} year^{-1}) and 2,327 kg K_2O year^{-1} (59 kg K_2O ha^{-1} year^{-1}) for optimal growth of grass silage. There is very little slurry left for application on winter wheat. The requirements of winter wheat are primarily met by mineral fertilisers, 7,225 kg N year^{-1} (193 kg N ha^{-1} year^{-1}), 3,124 kg P_2O_5 year^{-1} (83 kg P_2O_5 ha^{-1} year^{-1}) and 3,489 kg K_2O year^{-1} (93 kg K_2O ha^{-1} year^{-1}). The result is the import of 11,132 kg N year^{-1}, 3,245 kg P_2O_5 year^{-1} and 5,816 kg K_2O year^{-1}.

7.5 Digester and CHP size

Determining the volume of the digester is a key step as it is the highest capital cost component and central to both economic and environmental impact calculations. A conservative assumption which allows for no excess capacity in either the digester or the CHP unit has been taken. Some farmers may, however, choose to have some excess capacity available to account for future growth plans or potentially even limit the size owing to financial and other operational constraints. Based on the amount of slurry available, the minimum digester size required would be 145m^3. Assuming a cylindrical shape, a radius of 4.5m and height of 2.3m, is calculated. An organic loading rate of 3 kg VS m^{-3} day^{-1} and total and volatile solids at 8% and 80%, respectively results in a retention time of 21 days.

7.6 Methane produced

20,330 m^3 of methane (contained in 33,883 m^3 biogas) is produced by the digester and requires a 26 kW$_{total}$ CHP unit to generate heat and electricity from it.

The results from the farm model are used to calculate the emissions from the farm as detailed in the following chapter.

8. Emission model results

The total GHG emissions from the “Modelled farm” without an anaerobic digester are 7,193 kg CO_2 eq. ha^{-1} $year^{-1}$. The introduction of a digester reduces the GHG emissions by 725 kg CO_2 eq. ha^{-1} $year^{-1}$, a reduction of 10%. The savings are made up of 20% CH_4, 33% CO_2 and 47% N_2O, primarily from fossil fuel based electricity substitution and captured emissions during manure management. Further details of emissions from the “Modelled farm” as defined in Section 6.1 are presented in Table 23.

Table 23 Emissions model results

	Emissions (kg CO_2 eq. ha^{-1} $year^{-1}$)	
Source	**Pre-AD**	**Post-AD**
Enteric Emission	3,583	3,583
Manure Management		
- CH_4	242	24
- N_2O	342	-
Managed soils	2,049	2,049
Crop production	154	154
Production of Mineral fertiliser	671	671
Electricity	123	-117
Heat	29	15
Fugitive Emissions	-	77
Embodied carbon in AD	-	13
Total	**7,193**	**6,468**

8.1 Enteric emissions

Enteric emissions add up to 50% of the total emissions from the modelled dairy farm without a digester and 55% from the same farm with a digester. The increase in percentage contribution is attributed to the fact that on introduction of a digester, the total GHG emissions from the farm reduce, even though the enteric emissions remain constant. More enteric emissions, 130 kg CO_2 eq. $head^{-1}$ $year^{-1}$ are emitted from grazed dairy cows as compared to housed cows which emit 121 kg CO_2 eq. $head^{-1}$ $year^{-1}$. This is because the animals are more active and consume more energy than those housed; however, this may be compensated for by selective grazing to increase the digestibility of fresh grass. Similarly, grazed followers emit 45 kg CO_2 eq. head^{-1} year^{-1} while those housed emit 40 kg CO_2 eq. head^{-1} year^{-1}.

The enteric emissions are dependent on the livestock and the digestibility of the feed, and hence are not impacted by the introduction of digestion. The emissions may change if the housing of cattle is increased in order to collect more slurry for digestion.

8.2 Manure Management

Manure management accounts for 8% of the overall emissions from the "Modelled farm". Emissions from manure management account for 77% of the total GHG emissions abated by AD.

8.2.1 Methane

Methane emissions are 242 kg CO_2 eq. ha^{-1} year^{-1} accounting for 39 % of the total emissions from manure management. Emissions of CH_4 from manure are significantly higher when manure is stored from housed animals.

The methane conversion factor for a slurry based manure management system reported by Rodhe *et al.* (2009) is 2.7% which is much lower than the IPCC (2006) value of 10-17%. Hence, there may be an overestimation in the CH_4 emissions from slurry management calculated by the model which is based on IPCC methodology.

The emissions from manure deposited in the field from grazed cows do not change with the introduction of AD. The emissions from slurry tank storage are, however, completely eliminated on introduction of AD as the CH_4 in the biogas produced is directly passed on to the CHP. The net impact is that the total CH_4 emissions from manure management are reduced to a tenth of their value on introduction of AD.

8.2.2 Nitrous Oxide

In the pre-AD scenario, N_2O is emitted during the storage of slurry in an open tank and accounts for 190 kg CO_2 eq. ha^{-1} $year^{-1}$ of the emissions from the dairy farm. Indirect emissions from volatilisation and subsequent deposition of nitrogen add another 152 kg CO_2 eq. ha^{-1} $year^{-1}$.

There are no indirect emissions from leaching or run off as it is assumed that the slurry is removed from the housing area regularly and is collected in a slurry tank. This keeps the probability of leaching and run off of slurry to a minimum.

The assumption of crust formation leading to overestimation of N_2O emissions abated by a maximum of 190 kg CO_2 eq. ha^{-1} $year^{-1}$ for the "Modelled farm". As discussed earlier, the model assumes the formation of a crust during slurry storage. This can create aerobic micro-sites and lead to N oxidation and hence, N_2O emissions. The crust formation can happen under high temperature conditions or if the slurry has high total solids content. The former is possible during summer months and dependent on local weather conditions. The latter happens when essentially the slurry has high dry matter content, which can be a result of different farming practices e.g. if the farmer chooses to collect the manure from housed cows by "scraping" rather than flushing, the resultant slurry would have a high total solids content which can potentially lead to crust formation or if the amount of bedding in the slurry is high.

On introduction of AD, the slurry is directly fed into the digester and the digestate is stored in a gas tight storage tank which does not allow any oxidation of the N present and therefore, all direct and indirect emissions are abated.

8.3 Managed soils

Managed soils are responsible for 2,049 kg CO_2 eq. ha^{-1} $year^{-1}$ emitted from the farm. These account for 29% of the total emissions and 82% of all nitrous oxide emissions from the modelled farm without a digester. Most of the nitrous oxide is emitted directly (78%) with a majority of these (73%) arising from deposition and spreading of urine and dung. Crop residues are a minor source responsible for only 2% of the emissions from managed soils. Indirect emissions from atmospheric volatilisation and deposition, and leaching and run-off accounts for 22% of the overall emissions from managed soils.

The emissions from managed soils do not change with the introduction of a digester on the modelled farm but increase in proportion to 32% of all emissions and 100% of all N_2O as the emissions from other sources on the farm reduce. This is based on the assumption that the composition and availability of nutrients in the slurry pre- and post- AD are the same as no conclusive quantitative data was found to establish the difference. This may lead to some under estimation of emissions abated. On the farm level, these should be quantified by conducting field tests. Change in nutrient content of slurry pre- and post- AD will impact not only the emissions from the slurry but also the amount of mineral fertilisers required and the emissions associated with their manufacture and application.

8.4 Use of Fuel in farm machinery

Farm machinery like tractors and harvesters use diesel as fuel. The emissions from the use of diesel account for 154 kg CO_2 eq. ha^{-1} $year^{-1}$ or about 2% of the total emissions from the farm.

These emissions do not change as a result of the introduction of digestion, unless the machinery used for application of slurry is different from that used for application of digestate. For the purposes of the model it is assumed that the farm machinery used and hence the emissions from spreading digestate to land are the same as from manure used in the same way.

8.5 Production of mineral fertilisers

Based on the volume of mineral fertilisers needed to meet the requirement of the crops, as detailed in Section 7.4, the emissions from the production of these mineral fertilisers have been calculated to be 671 kg CO_2 eq. ha^{-1} $year^{-1}$. These account for 9% of total emissions from the modelled farm pre- AD and 10% in the post- AD scenario. The emissions from application of digestate are assumed to be the same those from undigested slurry. There may be some variation in emissions due to the change in nutrient composition of slurry on digestion. This change may be quantified by conducting tests determining the nutrient composition of both raw slurry and digestate. Variation in composition and availability impacts the amount required and hence, the emissions related to their manufacture.

8.6 Embodied Carbon

Emissions from the production of the construction materials for the digester have been accounted for as embodied carbon. At 13 kg CO_2 eq. ha^{-1} $year^{-1}$, these make up for 0.2% of total emissions. These are cradle to gate emissions and there may be further emissions from the transportation of materials to the site and their use, which are not included in the calculation for embodied carbon here.

The embodied carbon content can vary based on the type of digester and is primarily driven by the volume of concrete, steel and insulation material required. Steel digesters tend to have a higher carbon footprint as compared to concrete digesters (approximately 45% higher for the digester used for the “Modelled farm”). The quotes presented in Kottner *et al.* (2008) are, however, based on concrete digesters sourced locally, representing local costs and hence the assumption of this type of digester has been made in the design of “Modelled farm”.

8.7 Fugitive Emissions

Fugitive emissions negate approximately 11% of the GHG abatement benefit from introduction of anaerobic digestion. This variable drives the net environmental impact from the introduction of AD. It may be noted that given the imperfections in operating conditions of a digesters, it is nearly impossible to eliminate fugitive emissions. A farmer can, however, take a number of steps to keep fugitive emissions to a minimum. Such measures may include regular maintenance and monitoring of joints, pipes and valves, covering mixing pits and ensuring that any unused biogas is flared. This is particularly relevant as CH_4 has a GWP of 21.

8.8 Heat and electricity import/export

Overall energy efficiency of the CHP unit caps the thermal and electrical efficiencies. As a result, for a given overall energy efficiency, an increase in electrical efficiency results in a decrease in thermal efficiency. The overall energy efficiency is assumed to be 85% (DECC, 2012).

Figure 27 shows the trade-off between electricity and heat production for various different CHP unit electrical efficiencies based on the "Modelled farm".

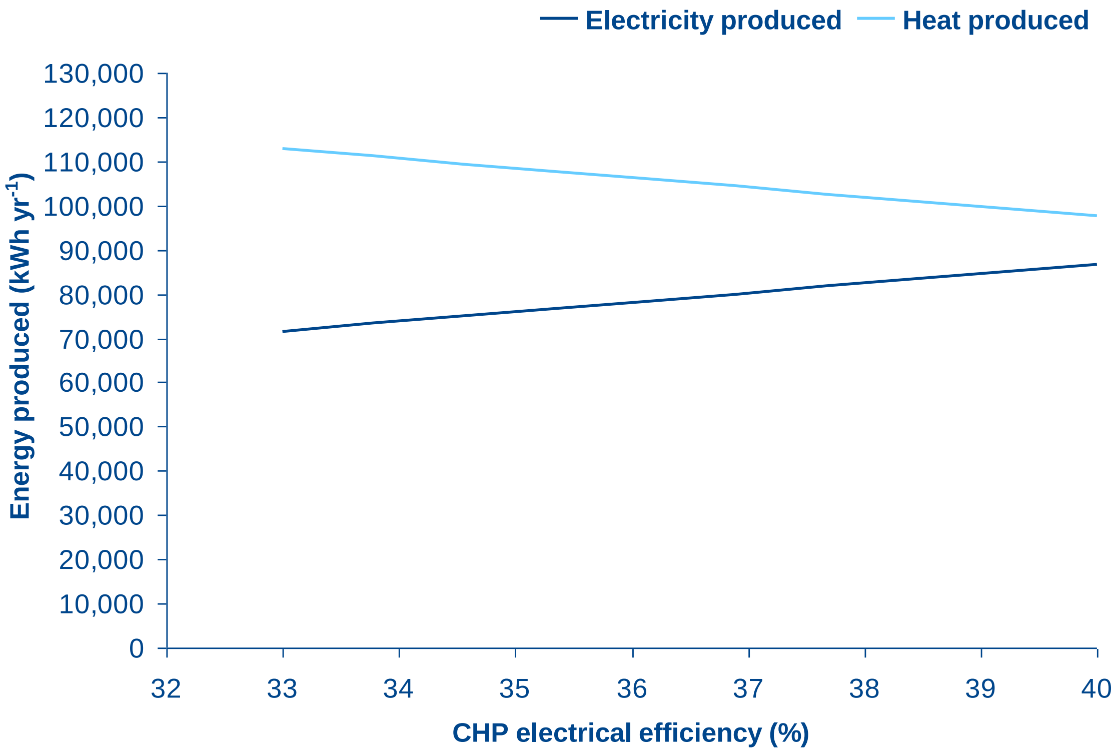

Figure 27 Total heat and electricity produced from a CHP unit with total energy efficiency of 85%

Table 24 provides a summary of the heat and electricity produced/consumed under both "Pre-AD" and "Post-AD" scenarios.

Table 24 Electricity and heat production

Energy production/consumption (kWh year^{-1})	Pre-AD	Post-AD
Electricity		
- Produced	0	78,086
- Parasitic load	0	-16,221
- Dairy use	-31,610	-31,610
- Exported	**-31,610**	**30,255**
Heat		
- Produced	0	106,283
- Parasitic load	0	-98,732
- Dairy use	-15,515	-15,515
- Exported	**-15,515**	**-7,964**

In the case of the "Modelled farm" with AD, there is surplus electricity that can be exported to the grid. The heat produced, however, is limited and fully consumed on-farm.

Electricity: In the pre-AD scenario, all the needs of the dairy farm (31,610 kWh year^{-1}) are met by import of electricity. The introduction of AD allows the farm to meet both the needs of the dairy (31,610 kWh year^{-1}) as well as the parasitic load of the digester (16,221 kWh year^{-1}) from the electricity generated by the CHP (78,086 kWh year^{-1}). There is, additionally, electricity (30,255 kWh year^{-1}) available for export or for other on-farm uses.

Heat: In the pre-AD scenario all the needs of the dairy farm (15,515 kWh year^{-1}) are met by import of heat in the form of LPG. Where a digester and CHP unit are added, the parasitic heat requirements of both bringing the slurry to operating temperature (76,672 kWh year^{-1}) as well as maintaining the temperature of the digester (22,060 kWh year^{-1}), are fully met by the heat captured by the CHP unit (106,283 kWh year^{-1}). It may be noted that the former is much higher than the latter.

The excess heat from the CHP unit (7,551 kWh year^{-1}) is used to meet the needs of the dairy (15,515 kWh year^{-1}) and the remainder is imported from outside (7,964 kWh year^{-1}).

The overall output mix of heat and electricity is realistic as the infrastructure to export heat is not available to most farms in the UK. On the other hand, the infrastructure to support the export of electricity is widely available.

All of these combined lead to a reduction in emissions by substitution of fossil fuel based energy. The GHG emissions associated with electricity and heat imported for the farm are reduced from 151 kg CO_2 eq. ha^{-1} year^{-1} to -103 kg CO_2 eq. ha^{-1} year^{-1}. This represents 35% of the AD emissions benefits.

8.9 Net emissions

The impact of AD on GHG emissions from the modelled farm is presented in Figure 28.

As discussed earlier, the introduction of AD does not impact many sources of emissions from dairy farms, specifically the enteric emission, emissions from soil management and crop production. From the sources of emissions that are impacted by the introduction of AD, a significant drop is seen.

The total emissions from sources impacted are reduced from 736 kg CO_2 eq. to 11 kg CO_2 eq., a reduction of 98.5%. This reduction is from manure management and CO_2 substitution from electricity and heat production, partially offset by embodied carbon and fugitive emissions as discussed above.

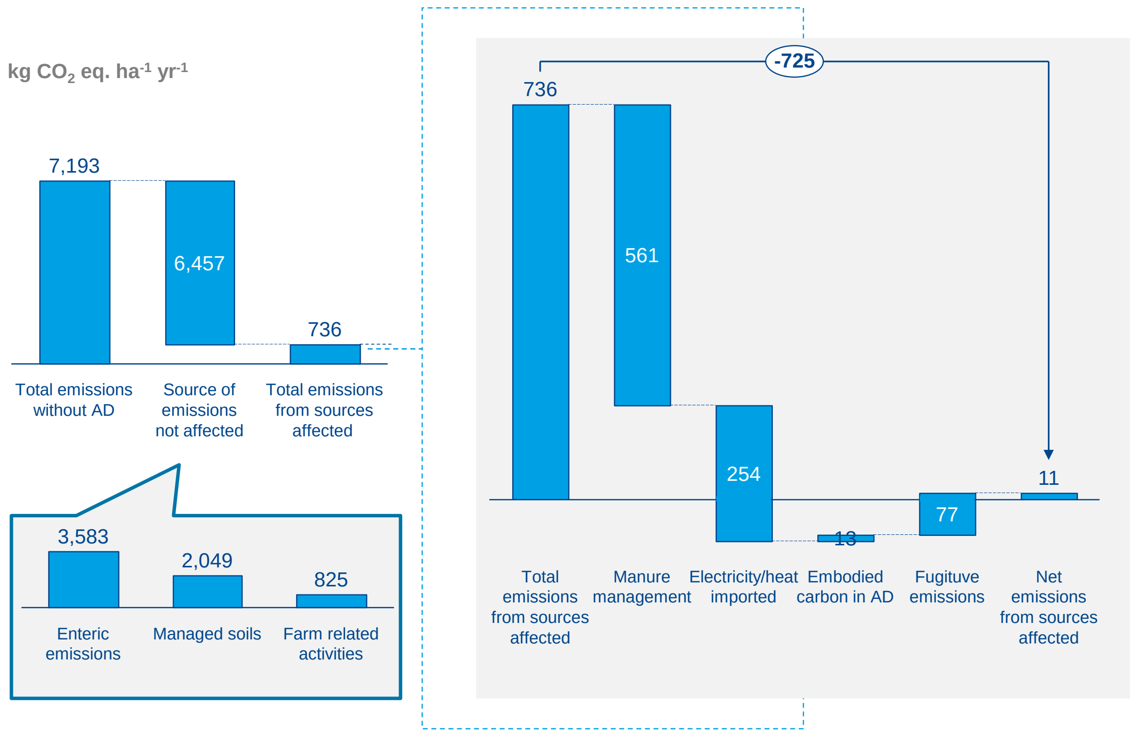

Figure 28 Impact of AD on sources of emissions on dairy farms

9. Economic model results

The introduction of AD impacts both the costs and the revenues of a farm. On the cost front, key variables that change are:

a) initial capital outlay for the installation and construction of a digester
b) capital outlay on a unit to process the biogas generated (either a CHP unit or a biogas upgrade unit)
c) operating costs of the AD setup

On the revenue front, the key change is additional revenue from the production of heat and electricity. The balance of these additional costs and revenues determines the net impact to the farmer from introduction of AD and these various factors are discussed in detail in the following sub-sections for the "Modelled farm".

9.1 Capital cost

The digester capital cost is the biggest incremental cost incurred on the introduction of AD and is the primary driver of the economics. It is linked to the digester size via a power function equation as provided in Section 5.1.

Based on Equation 46, the capital cost of a slurry based digester of size 145 m^3 is calculated to be £78,915. This implies a unit capital cost for the digester of £545 m^{-3} or £3,076 kWe^{-1}. This falls within the guideline range proposed for the UK of £400-750 m^{-3}, or £2,500 - £6,000 kW^{-1} (Redman, 2010). In the "Modelled farm" scenario, the farmer incurs a loss at the cost structure mentioned above. Hence, the revenue will need to increase to compensate for the higher per unit cost of the digester at "Modelled farm" scale.

The relationship of unit digester cost to the digester size is illustrated in Figure 29.

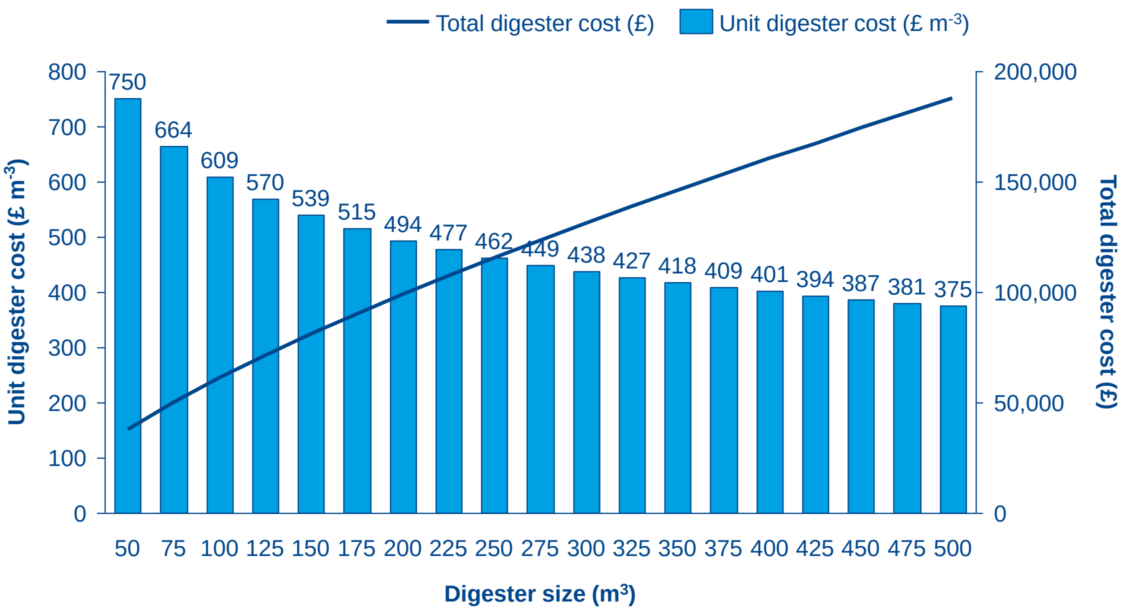

Figure 29 Digester cost (total and per unit) as a function of digester size

The capital cost of a CHP unit exhibits behaviour similar to that of the digester cost. The power factor in the case of CHP cost is 0.53 which is lower than the power factor for digester cost. For the "Modelled farm", CHP cost is calculated, using Equation 47, to be £42,810 equivalent to a unit cost of £1,648 kW^{-1} of installed capacity.

Figure 30 illustrates the detailed relationship of total CHP cost and CHP cost per kW as the CHP capacity increases.

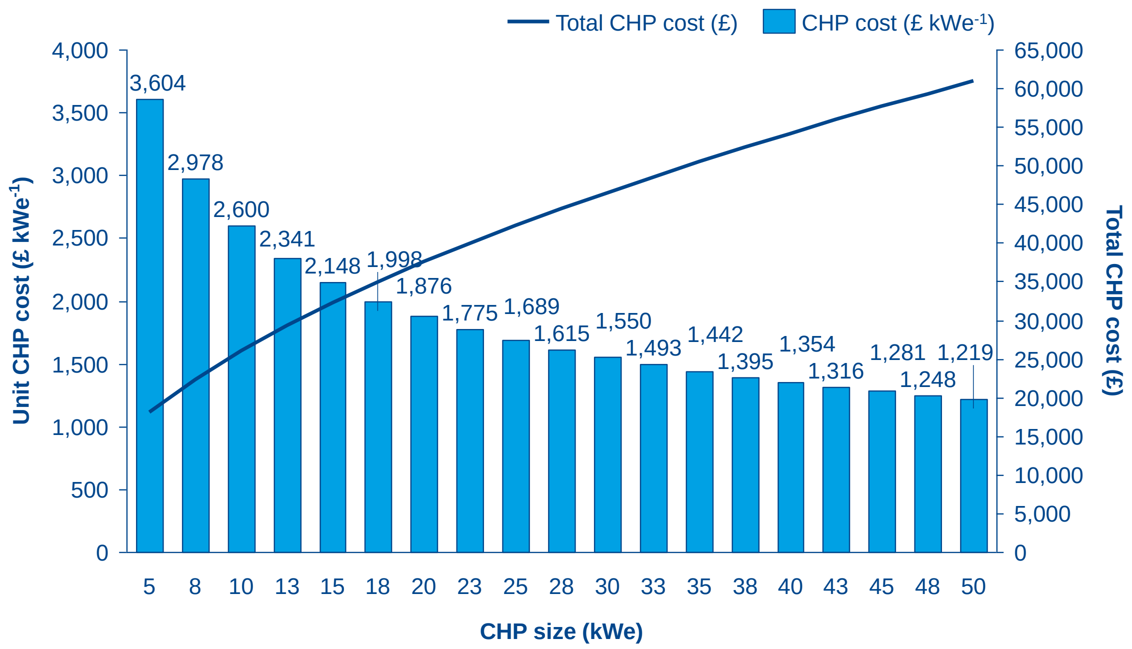

Figure 30 CHP cost (total and per unit) as CHP size increases

The CHP cost equation, as derived in Section 5.2, also implies that although per unit cost decreases as the CHP size increases, the rate of change decreases as the size increases.

The combined capital cost of installing the digester and the CHP unit for the "Modelled farm" is calculated to be £121,726.

9.2 Financing costs

Only the capital costs for installation of the digester and the CHP unit and the set-up fee are assumed to be financed via a mortgage. All other running expenses are assumed to be borne out of the operating cash flows of the farm. Adding a set-up fee of 1% as typically charged by the bank, the mortgage amount is increased to £122,943.

Based on a fixed rate mortgage with an APR of 6.5%, yearly payments of £11,158 or £80 ha^{-1} year^{-1} for 20 years have been calculated. This mortgage results in the farmer paying a total of £100,214 in interest over the lifetime of the mortgage, 81.5% of the original capital outlay required.

A monthly mortgage may change the total amount of interest paid but given the long period of the financing, the total interest paid would remain high.

9.3 Operating cost of AD

The operational expenditures due to introduction of AD to the "Modelled farm" are calculated as £5,524 $year^{-1}$ or £40 ha^{-1} $year^{-1}$. This takes the form of increased labour costs (£11.3 $ha^{-1}year^{-1}$), maintenance and repair (£19.7 ha^{-1} $year^{-1}$) and insurance (£8.5 ha^{-1} $year^{-1}$). The maintenance and repair cost of CHP are calculated to be £781 $year^{-1}$ or £5.6 ha^{-1} $year^{-1}$.

9.4 Heat and Electricity

The UK government subsidises electricity and heat from renewable sources by offering a guaranteed fixed price. The impact of sale of heat and electricity on the economics of the farm has been discussed below, with and without subsidy.

Without subsidy:

The electricity generated by CHP (78,086 kWh $year^{-1}$) is sufficient to meet the requirements of the digester (16,221kWh $year^{-1}$) and the farm (31,610 kWh $year^{-1}$), implying a saving of £3,231 $year^{-1}$ as compared to a farm without AD. Additionally, the excess electricity (30,255 kWh $year^{-1}$) is exported to the grid at a negotiated price (5.5 pence kWh^{-1}) which results in a further £1,664 $year^{-1}$ profit.

The heat generated by the CHP unit (106,283 kWh $year^{-1}$) is sufficient to cover the parasitic load (98,732 kWh $year^{-1}$) but is not able to support all of the dairy heat requirements (15,515 kWh $year^{-1}$). The difference must be imported (7,964 kWh $year^{-1}$). This results in a decrease in expenditure from £1,789 $year^{-1}$ to £918 $year^{-1}$.

With subsidy:

FIT supports both the electricity generated and the electricity exported. This results in zero expenditure on electricity, plus £9,094 $year^{-1}$ from the generation tariff and £968 $year^{-1}$ from the export tariff. As the FIT for export of electricity is only 3.1 pence kWh^{-1} compared to the import price of 10.22 pence kWh^{-1}, it makes sense for any surplus electricity (post meeting parasitic load of the digester) to be first consumed to meet the dairy needs to substitute the high price imported electricity.

There is expected to be some heat, produced by the CHP, left after covering the digester's parasitic load. The farmer is able to claim a RHI only on this excess heat used in the dairy, earning an additional £536 $year^{-1}$. The price of imported heat in the form of LPG is 11.53 pence kWh^{-1} while the RHI is 7.1 pence kWh^{-1}. Hence, use of heat onsite and claiming of RHI is economically more lucrative than export.

9.5 Net profits

The introduction of AD results in additional expenditures in the form of mortgage payment, maintenance and repair, labour and insurance; which are partially compensated by the revenues generated from heat and electricity production via the CHP unit. After taking into account the subsidies offered by the government, there is a net decrease in profit of the farm by £2,763 $year^{-1}$ or £20 ha^{-1} $year^{-1}$.

9.5.1 Net Present Value

The discount rate based on Equation 54 is 8.14%, which is in the 7-10% range as suggested by Oxera (2011). This discount rate captures the perceived risk of the project and accounts for the premium required above risk free rate.

Based on this discount rate, the NPV of the "Modelled farm" calculated as per Equation 56 is -£18,210. Thus, the introduction of AD is expected to lead to a loss under the "Modelled farm" parameters. This calculation is based on the operating cash flows of the farm post introduction of AD. The cost of a mortgage is in addition to this and would be an added burden to the farmer.

The impact of key sensitive variables on NPV is discussed in detail in the "Sensitivity analysis" later.

9.5.2 Payback period

Payback period is calculated to be 15 years. This is the payback period on a cash basis and does not account for the discounted value of future cash flows. As the NPV is negative for the "Modelled farm" case, it would not be possible to calculate a discounted payback period.

9.5.3 Internal Rate of Return

IRR for the project is 6.3%, lower than the project discount rate, which implies that the project is a poor investment choice or in other words, loss making for the farmer from a time value of money perspective. Only projects that have an IRR > discount rate for the project would make a profit for the investor. This is reflected by the negative NPV.

IRR is discussed in further detail in the sensitivity analyses as the financial implications of various different farm setups as defined by altering key inputs variables have been studied.

The results from the emissions and economic models are brought together for the calculation of marginal abatement cost as detailed in the following chapter.

10.Marginal abatement cost results

This chapter presents results and discussion for the marginal abatement cost which results from the introduction of a digester to a dairy farm. The chapter starts by examining the actual MAC achieved without subsidies and how that is affected by the introduction of subsidies. The final part of the chapter addresses the issue of variability and presents the results of a Monte Carlo analysis.

Marginal abatement cost

A summary of results from the emission and economic models are presented in Table 25.

Table 25 Summary of results from emissions and economic models

Summary results	Pre-AD	Post-AD	Difference
GHG abated (kg CO_2 eq. ha^{-1} $year^{-1}$)	7,193	6,468	724.6
Profit difference (£ ha^{-1} $year^{-1}$)			
- without subsidy	1,155	1,072	83.1
- with subsidy	1,155	1,136	19.7

The introduction of AD reduces the GHG emissions from the farm by 0.725 tonne CO_2 eq. ha^{-1} $year^{-1}$. This reduction in emissions comes at a cost of £83.1 ha^{-1} $year^{-1}$. Thus the marginal abatement cost is £114.5 $tonne^{-1}$ CO_2 eq. abated. The FIT reduces the MAC to £32.5 $tonne^{-1}$ CO_2 eq. abated, a reduction of £82 $tonne^{-1}$ CO_2 eq. abated. This is achieved primarily by the increase in revenues from FIT (an increase in profits of £60 ha^{-1} $year^{-1}$) and this reduction maps directly to the incentive from the government for the twin goals of fossil fuel substitution and carbon abatement. RHI further reduces the MAC marginally to £27.2 $tonne^{-1}$ CO_2 eq. abated.

The unsubsidised MAC calculated here differs from the work by Moran *et al.* (2008) which proposed a MAC of £26 $tonne^{-1}$ CO_2 eq. using on-farm AD for medium sized dairy farms in the UK. This MAC calculation is based on the capital cost estimates presented in FEC services (2003), which does not take into account interest costs and assumes a lower annual running cost (2% of capital cost).

The non-traded price of carbon that is in use for appraising policies that reduce/increase emissions in sectors not covered by the EU ETS is £53 per tonne of CO_2 eq. abated ± 50% for the year 2012 (£27-£80 per tonne of CO_2 eq. abated). The calculated MAC of £83.1 $tonne^{-1}$ CO_2 eq. abated is within the recommended range for feasible policies.

The calculated MAC is comparable to that of on- and off-shore wind energy, reported by Committee on Climate Change (2008) at £55-£133 $tonne^{-1}$ CO_2 eq. abated and £85-£152 $tonne^{-1}$ CO_2 eq. abated, respectively. It is lower than the reported MAC of £193 $tonne^{-1}$ CO_2 eq. abated for marine power.

A MAC of £27.2 $tonne^{-1}$ CO_2 eq. after the subsidy is currently borne by the farmer. The focus of the FIT is primarily CO_2 abatement through the replacement of fossil fuels. By digesting the cattle slurry the farmer has, however, contributed to GHG reduction through the abatement of CH_4 and N_2O. Arguably, the residual MAC of £27.2 $tonne^{-1}$ CO_2 eq. is the cost of abating CH_4 as well as N_2O.

The impact of other incentive structures is discussed below:

10.1.1 Impact of ROCs

The UK Government also provides subsidy to the renewable energy industry in the form of ROCs. Plants with installed capacity of 5 MW or more may be eligible for ROCs. For a purely slurry based digester on a farm size of 140 hectares, the installed capacity is 26 kW. Hence, a farm size of 5,833 hectares would be required to feed a CHP of 1MW installed capacity. Given that most UK dairy farms are significantly smaller than this, they are unlikely to be eligible for ROCs and would need to rely on FIT based incentives.

10.1.2 Claiming carbon credits

In order to claim carbon credits, there are a number of procedural requirements that a facility needs to meet. These include but are not limited to contract negotiation and writing, internal monitoring, mandatory checks on design, validation and verification. The upfront cost of these can be quite significant. Disch *et al.* (2010) estimated these costs to be $5,000 (£3,200) for project assessment, $40,000 - $50,000 (£25,600 – £32,000) for document preparation, $30,000 - $50,000 (£19,200 - £32,000) for validation and $3,000 - $5,000 (£1,920 - £3,200) in the form of legal costs. This report also estimated the monitoring costs to be $20,000 – $40,000 (£12,800 - £25,600) every 2 years, issuance fees of 2% of the issued credits and an additional registration fee.

The traded price of carbon has varied between £4 $tonne^{-1}$ and £25 $tonne^{-1}$. The government has now set the carbon floor price at £16 $tonne^{-1}$ for 2013 effective April 2013 and it is expected to reach £30 $tonne^{-1}$ by 2020 (Ares, 2012). The current price of carbon is £4 $tonne^{-1}$ which is lower than the floor price.

As per the model, the modelled farm can claim up to 100 carbon credits. The additional benefit from these would be £1,600 $year^{-1}$ based on the floor price. At this rate it would take over 40 years to claim back the upfront cost which is more than the lifetime of the digester.

Thus, given the low price of carbon and high transaction costs (in terms of accreditation, registration, etc.) by claiming carbon credits, the farmer would actually incur a loss. The number of carbon credits would have to be much higher to change this into a profit making proposition.

10.1.3 Growing maize to improve the biogas methane yield

An alternative option for the farmer to reduce MAC borne is to grow maize and increase the methane yield of the slurry by adding maize to it. For the modelled farm, the farmer is bearing a cost of £27.2 $tonne^{-1}$ CO_2 eq. abated for the 0.725 $tonne^{-1}$ CO_2 eq. ha^{-1} $year^{-1}$ abated for a farm size of 140 ha. Thus, the farmer would need an additional £2760.8 $year^{-1}$ as revenue to break even. Assuming that FIT and RHI are claimed, 27924 kWh of energy will be required to generate this revenue. Based on a specific methane yield of 0.33 CH_4 kg^{-1} VS added (Cornell, 2011), maize yield of 40 tonnes fresh matter ha^{-1} $year^{-1}$ (Countryside, 2010), gross calorific value of methane of 15.4166 kWh kg^{-1} and density of 0.717 kg m^{-3} (DECC, 2010b), an additional area of about 0.8 hectare will be required to compensate for the MAC. There may be some additional area requirement to compensate for the bigger digester, additional equipment to process and store the maize, and the additional parasitic load. The financial feasibility of growing maize may come at a considerable environmental cost if grassland or permanent pasture is converted to cropland. Conversion of one hectare of grassland to cropland results in release of 1.14 tonne CO_2 eq. (IPCC, 2006).

10.2 Alternative AD operational setups

In this section, alternative AD setups that could be considered by farmers are studied. A high level assessment of three setups is made:

Flaring of biogas produced post-AD ('Flare'): In this setup, the farmer flares the biogas produced post digestion without making the required investments to manage a CHP unit. All the biogas produced is burned and not processed by a CHP unit. In this scenario, AD is treated purely as an emissions abatement technology and not a source of incremental income. The emissions from flaring of biogas are not considered as they are a part of the natural biological carbon cycle as explained in Section 4.8.

Use of biogas in a boiler to meet on-farm heating needs ('Boiler'): In this setup, the farmer could install a boiler on-site to use the biogas to meet his on-farm needs which include the needs of the dairy and the parasitic load of the digester. All the surplus biogas is assumed to be flared.

Biogas upgrade for exporting to the grid ('Upgrade'): In this case, the biogas produced in the digester is assumed to be transferred to a biogas upgrade unit, which would enhance the quality of biogas to match the properties of natural gas that can be exported to the gas grid.

Table 26 provides key metrics under the "Modelled farm" scenario with "CHP" and each of the alternative scenarios described above.

Table 26 Comparison of key metrics under various options for using biogas

Key metrics	CHP	Flare	Boiler	Upgrade
GHG abated (kg CO_2 eq. ha^{-1} $year^{-1}$)	725	471	499	666
Initial capital cost (£s)	121,726	78,915	92,049	177,473
- Digester cost (£s)	*78,915*	*78,915*	*78,915*	*78,915*
- CHP cost (£s)	*42,810*	*0*	*0*	*0*
- Upgrade unit cost (£s)	*0*	*0*	*0*	*98,558*
- Boiler cost (£s)	*0*	*0*	*13,134*	*0*
Revenue from electricity/heat/biogas (£s $year^{-1}$)	9,680	-18,061	-3,787	3,502
- Electricity revenue (£s $year^{-1}$)	*10,062*	*-4,888*	*-4,888*	*-4,888*
- Heat revenue (£s $year^{-1}$)	*-382*	*-13,173*	*1,102*	*8,390*
Marginal Abatement Cost (£s $tonne^{-1}$ CO_2 eq.)	27	431	220	171
NPV (£s)	-18,210	-328,891	-171,932	-187,384
Internal rate of return (%)	6.3%	N/A	N/A	N/A

These results are discussed in detail below:

10.2.1 Flaring of biogas produced post-AD

In this scenario, there is GHG abatement from the capture of emissions during manure management but as all the biogas is flared, there is no electricity or heat production and hence no fossil-fuel substitution which reduces the emissions benefit. Some of the GHG abated is negated by the embodied carbon (13 kg CO_2 eq. ha^{-1} $year^{-1}$) in the digester and the fugitive emissions (77 kg CO_2 eq. ha^{-1} $year^{-1}$) during digestion. The net effect is still some reduction in GHG emissions (471 kg CO_2 eq. ha^{-1} $year^{-1}$), though without any financial benefits as in the case of CHP setup.

From an economic viewpoint, the farmer incurs the cost of installing the digester and does not make any incremental income from the sale of heat and electricity. The interplay of lower GHG abatement and reduced profit leads to a significantly higher MAC compared to the MAC under the “CHP” setup. This is not an attractive setup for the farmer but is the case when the CHP is down for maintenance and repair.

In practice, some farmers use part of the biogas for local cooking and heating needs via use of Raeburn cookers. Though this alleviates the financial burden of the household cooking bills, the economic benefits may not be enough to sufficiently reduce the MAC to make the overall enterprise profitable.

10.2.2 Use of biogas in a boiler to meet on-farm heating needs

This setup is relatively easy to install and can be a lower capital investment proposition for the farmer. Boilers are easy to procure and install and for the "Modelled farm". The biogas produced (224,771 kWh $year^{-1}$) has sufficient net energy to satisfy the heating needs of the dairy (15,515 kWh $year^{-1}$) and thermal parasitic load of the digester (98,732 kWh $year^{-1}$).

The GHG abatement is found to be higher compared to the "Flare" option as the imported heat based on fossil-fuel based sources is completely replaced. From the economic viewpoint, the heating bills are reduced to zero and there is some additional income from RHIs for the heat used in the dairy. The NPV is significantly lower compared to the "CHP" setup and the MAC is significantly higher. This can be explained by the loss of electricity FIT revenue that the "CHP" setup provides.

This setup can be useful where local heating needs are substantial, for example a brewery or a farm with large attached cottages. The boiler setup is easy to install and maintain and may be considered in certain scenarios. The use of biogas locally to heat water in a boiler, however, has limited applications in summer months and in absence of an alternative, the biogas produced may end up being flared, which weakens the case for installing a boiler.

10.2.3 Biogas upgrade for exporting to the grid

Biogas upgrade as an alternative to CHP is economically unviable for the "Modelled farm" scenario. The MAC is much higher than CHP at £171 $tonne^{-1}$ CO_2 eq. For the amount of biogas produced at the "Modelled farm", an upgrade unit of capacity 3.9 m^3 $hour^{-1}$ is required. The cost for this unit based on the equation developed in Section 5.3 would be close to £100K which is more than 2x the cost of the CHP required for the "Modelled farm" case.

Electricity: In terms of economics, the loss of revenues from electricity sale reduces the NPV compared to the CHP case. Additionally, the parasitic load of the digester and biogas upgrade unit needs to be imported and creates a significant economic disincentive for the farmer as well as adding to the net carbon footprint. Thus the electricity import related carbon footprint is zero under the CHP scenario as electricity generated covers both the parasitic load and dairy needs. Under the biogas upgrade scenario, however, the carbon footprint is 123 kg CO_2 eq. ha^{-1} $year^{-1}$ for the fossil fuel based imported electricity.

Heat: Under the biogas upgrade scenario, the thermal parasitic load as well as that required by the dairy can be fully met by burning the biogas produced. The surplus biogas after meeting both the thermal parasitic load and dairy needs is exported as bio-methane to the grid. This helps in reducing emissions from the fossil fuel based heat import under the CHP scenario. The heat import related carbon footprint is 29 kg CO_2 eq. ha^{-1} $year^{-1}$ under the CHP scenario and -166 kg CO_2 eq. ha^{-1} $year^{-1}$ under the "Biogas upgrade" scenario. Revenue earned from RHI is significantly lower than the net income from electricity generation under the CHP scenario. Overall the farmer makes less operating income if he/she chooses to install an upgrade unit as compared to a CHP unit.

Currently, there are only three functional biogas upgrade facilities in the UK, one of which is at Didcot sewage works, a large scale facility that primarily processes sewage. The Adnams Brewery processes high energy brewery waste and local food waste. The third facility, Rainbarrow Farm, is the only agricultural gas to grid facility in the UK and was commissioned in 2012. The AD plant processes maize, grass, slurry, manure and other farm wastes adding up to 38,000 tonnes per annum (Defra, 2013).

Farm scale biogas upgrading facilities are still in a developmental stage are not financially viable. A typical farm in the UK does not have access to the gas grid or the capabilities to monitor the quality and mix of gas exported. The infrastructure for distribution of biogas as vehicular fuel is not available widely in the UK. All of these issues make biogas upgrading an unattractive proposition for an average UK farm.

10.3 Sensitivity analysis

Sensitivity analysis has been used to study the impact of each input parameter on the MAC individually. The sensitivity of the models to different input parameters has been presented in the following sections.

10.3.1 Generation FIT

The Generation FIT is a source of incremental revenues that the farmer makes on the electricity produced which is in excess of the parasitic load of the digester. In the case of the "Modelled farm", the farmer is able to produce electricity from a CHP unit in excess of the dairy's needs as well as the parasitic load as discussed in Section 8.8. Generation FIT is directly linked to the amount of electricity generated and hence, is a key variable impacting the revenue for the farmer.

Higher revenue would affect the output economic variables like MAC, IRR and NPV. The MAC, which provides a measure for cost to the farmer per tonne of CO_2 eq. abated, is directly and negatively correlated to FIT as shown in Figure 31.

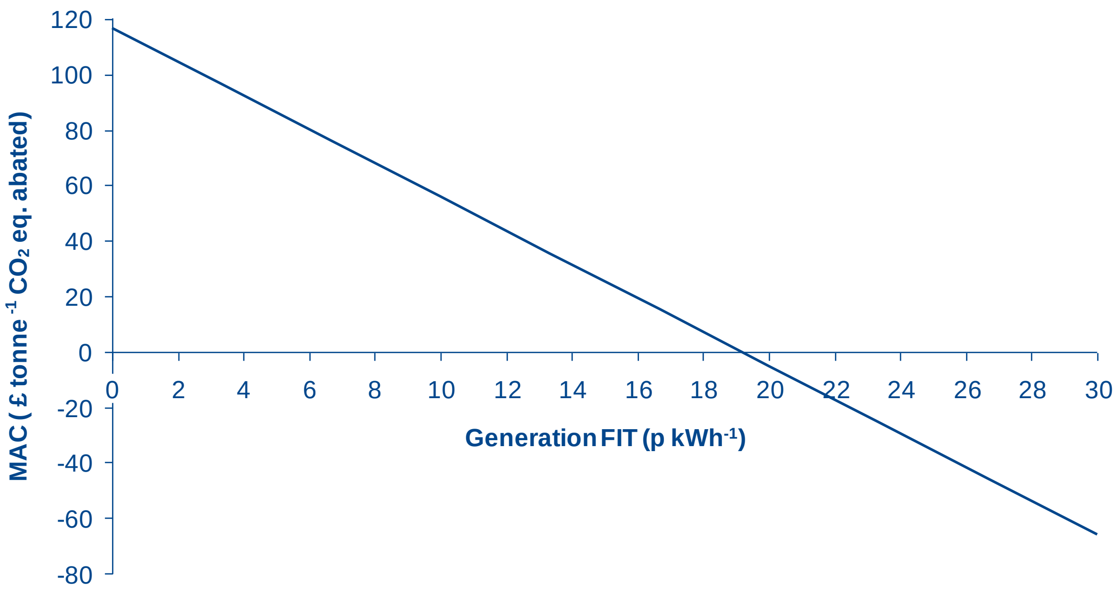

Figure 31 Variation of MAC with change in FIT

In order for the MAC to be zero a $FIT_{generation}$ of 19.2 pence kWh^{-1} is required in addition to the FIT_{export}, 3.2 pence kWh^{-1}.

A broader sensitivity analyses of the NPV and IRR metrics to changes in Generation FIT is presented in Figure 32

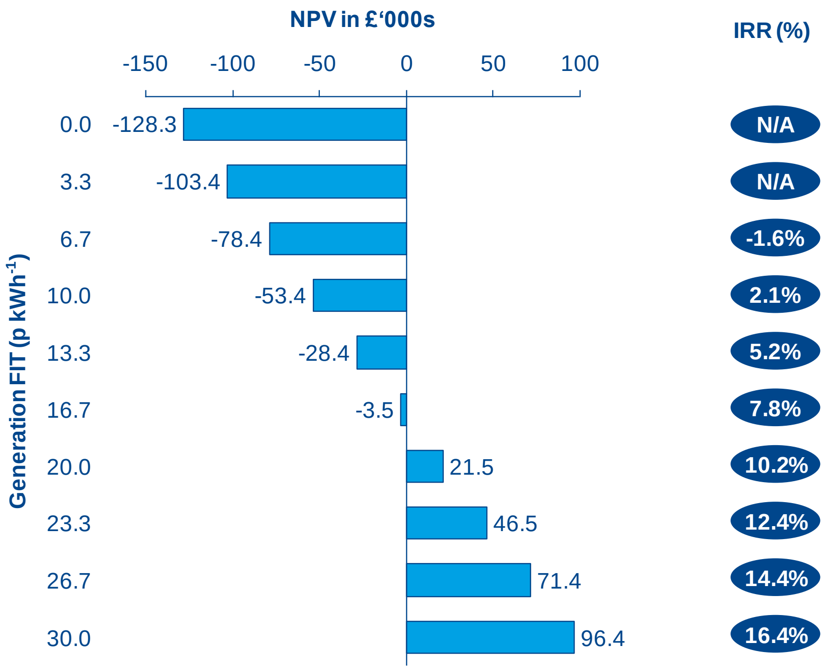

Figure 32 Impact of change in generation FIT on NPV and IRR

The distribution implies that if the value of Generation FIT is below 20.0 pence kWh^{-1}, the NPV remains negative and the IRR below the discount rate of 8.14%. At these lower values for Generation FIT the farmer makes a loss through the introduction of AD using the biogas for CHP.

The Generation FIT, however, has a very significant impact on the farm economics. Both the NPV and IRR increase significantly with every pence increase in the Generation FIT. Thus increasing the generation FIT from the current level of 14.7 pence kWh^{-1} by 1p, the NPV moves from -£18,210 to -£10,718, a 41% increase. This sensitivity remains similar at other price points.

The export FIT is assumed to be constant in the model as its impact on the MAC is expected to be the same as that of generation FIT. Increasing the export FIT would have limited impact due to the small amount of electricity available for export on the modelled farm, as well as the smaller impact from the export FIT of 3.2 pence kWh^{-1} as compared to the generation FIT of 14.7 pence kWh^{-1}.

10.3.2 Farm Size

Farm size has a direct bearing on the digester size and overall economic and environmental potential of the farm. For this part of the analysis, it has been assumed that the livestock density (1.6 LU ha^{-1}) as well as the ratio of dairy cows to followers is maintained as the farm size changes. Keeping all else constant, a larger farm size would imply a larger number of cattle and hence the need for a larger digester to process the manure. The calculation to estimate the digester size from farm size has been discussed earlier in Section 3.5. The impact of a change in farm size on the overall economic and environmental variables on introduction of AD is discussed here.

From the emissions viewpoint, GHG emissions in the absence of AD grow linearly with farm size as they are directly proportional to the herd size.

On introduction of AD, parasitic load and embodied carbon related to digester size are introduced. Parasitic load is made up of two components, of which electric parasitic load is linearly linked to digester size and the heat parasitic load is related to the surface area and volume of the digester. As a result, on a per unit volume basis, the heat parasitic load goes down. As digester volume is linearly linked to farm size, heat parasitic load per hectare has an inverse relationship with farm size as shown in Figure 33.

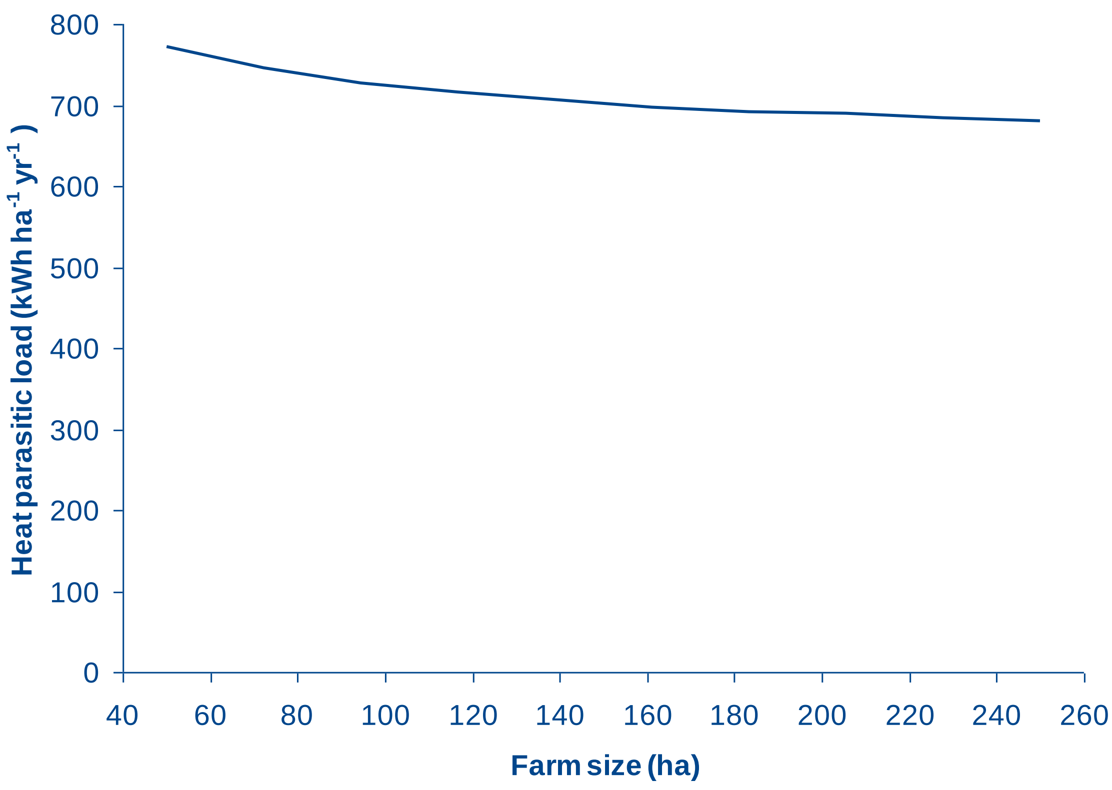

Figure 33 Change in thermal parasitic load with increasing farm size

In addition, embodied carbon is related to the volume of the construction materials used in setting up the digester and has a relationship to digester volume and hence to farm size. The interplay of all of the above mentioned factors implies that GHG emissions per hectare decline with increase in farm size in a post-AD setup as demonstrated in Figure 34.

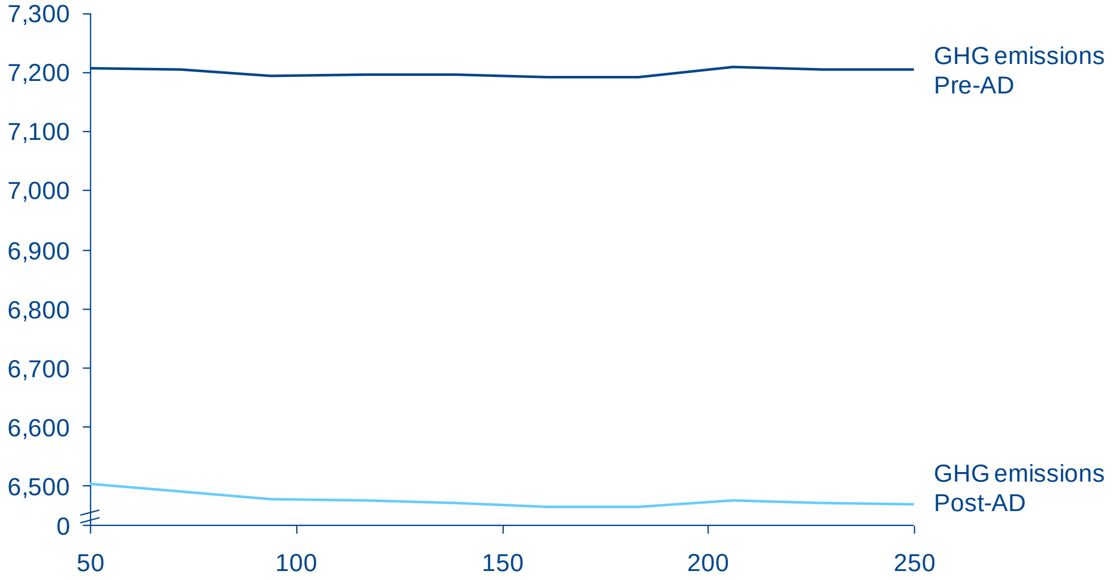

Figure 34 Variation in GHG emissions with increase in farm size

The dynamic between farm size and economic variables is also complex given the different, non-linear relationships between input and output variables.

On the revenue front, the farmer gets a near linear increase with increase in farm size. This is driven by a growth in the revenue from Generation FIT which in turn is linked to the electricity production increase as shown in Figure 35.

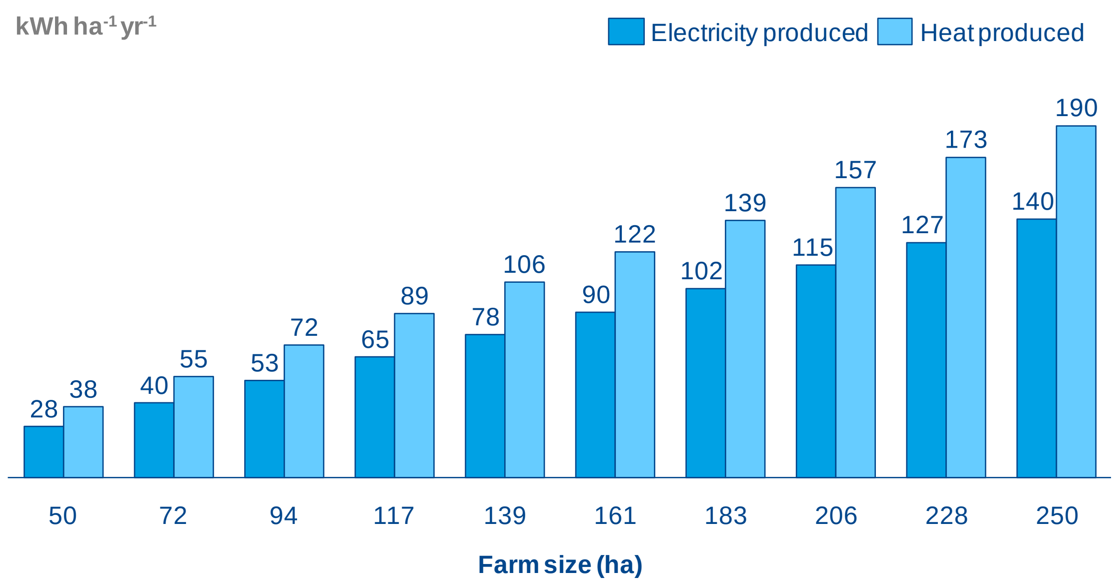

Figure 35 Change in energy produced with change in farm size

On the other hand, costs do not increase linearly with increasing farm size. As discussed in Section 9.1, though total digester costs increase with the size of the digester, the rate of increase reduces as the digester gets bigger. As a result, on increasing the farm size, unit cost of AD reduces. The interplay of these two factors increase the profits per hectare as farm size increased as shown in Figure 36.

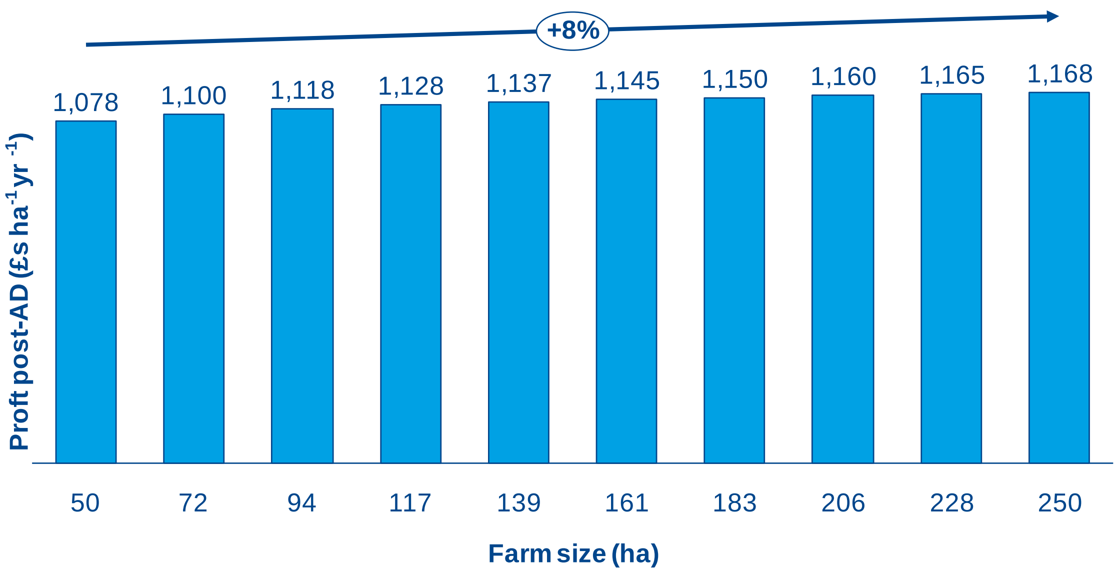

Figure 36 Change in profit with increase in farm size

Figure 37 and Figure 38 show the impact of increase in farm size on NPV and IRR, respectively.

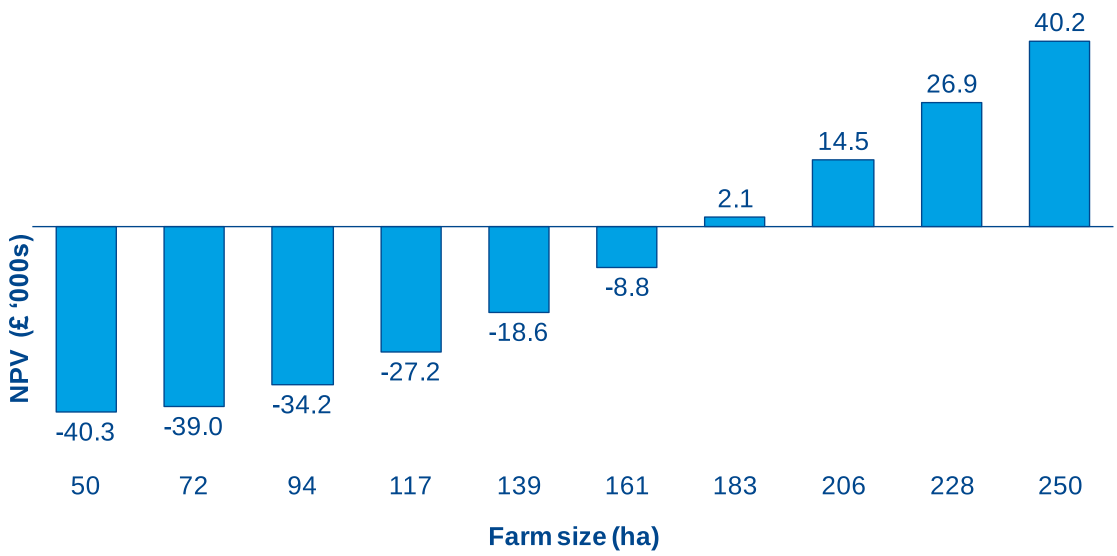

Figure 37 Change in NPV with increase in farm size

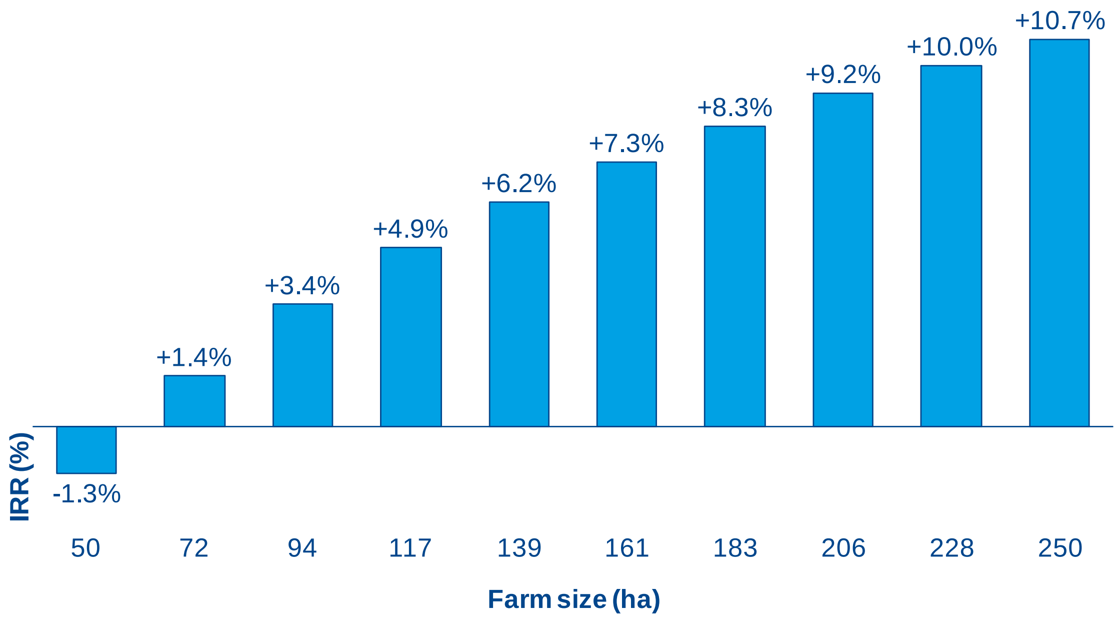

Figure 38 Change in IRR with increase in farm size

Though the profitability increases, NPV does not become positive for farm sizes below 183 hectares. Correspondingly, IRR remains below the discount rate of 8.14%.

Bringing both the economic and environmental impacts together, the impact of increasing farm size on the MAC can be seen. The MAC reduces from £116 to -£9 as farm size changes from 50 hectares to 250 hectares.

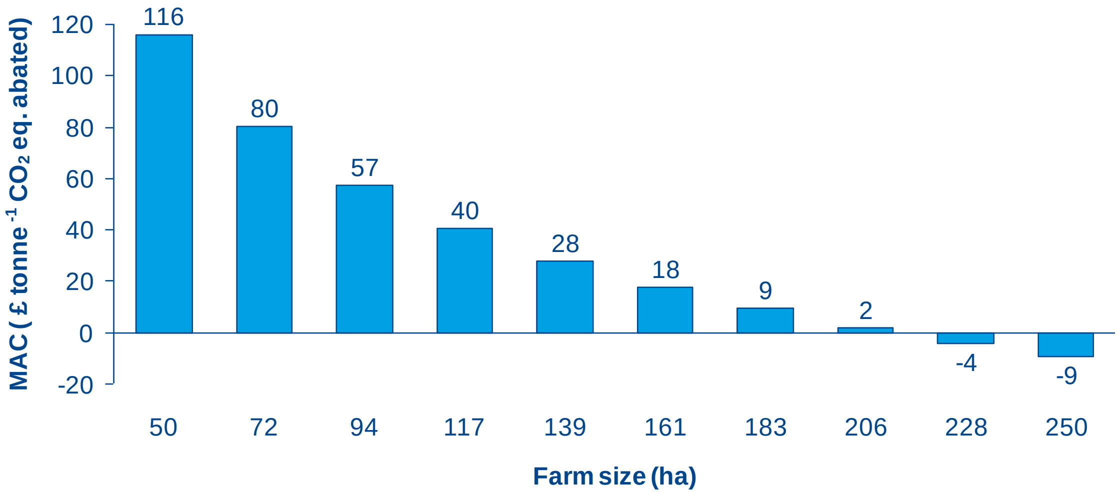

Figure 39 Change in MAC with variation in farm size

10.3.3 Housing percentage

Emissions of CH_4 from manure are higher when manure from housed animals is stored in a slurry tank/lagoon (anaerobic conditions that encourage CH_4 production) than when it is deposited by grazing animals (aerobic conditions). Introducing AD allows the farmer to capture as well as utilise the increased biogas produced under housed conditions, helping both electricity production and GHG abatement. The net result is that as housing percentage is increased, the GHG emissions in the post-AD scenario decrease compared to GHG emissions in the pre-AD scenario. This implies that abatement increases as percentage of housing increases. Specifically, the increase in GHG emissions per hectare per year is 2% in pre-AD scenario when housing percentage is increased from 60% to 100%. The same change in housing percentage leads to a 3% decline in emissions in post-AD scenario as shown in Figure 40.

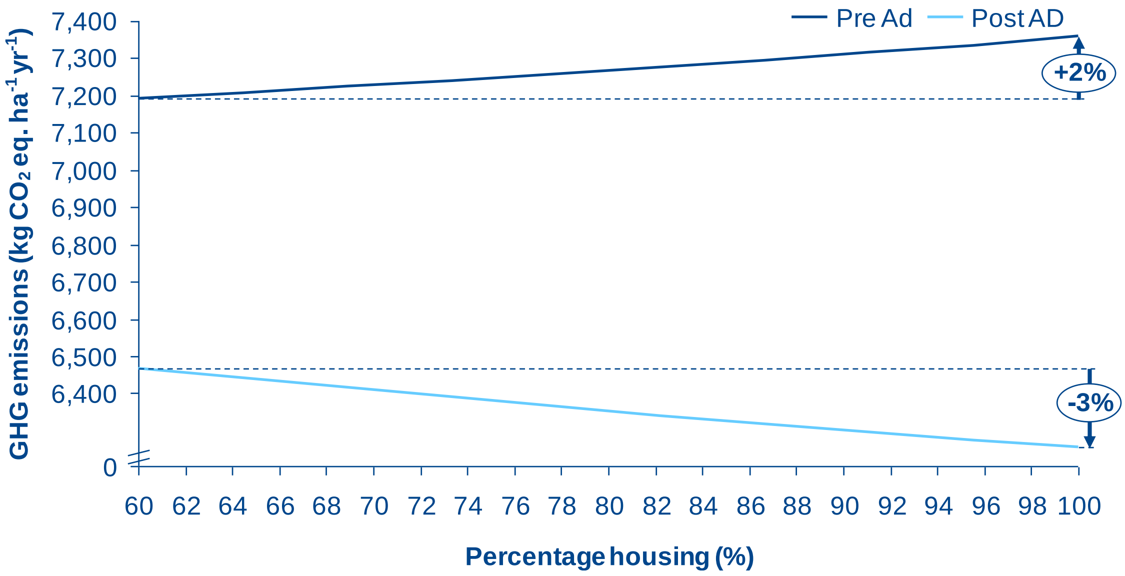

Figure 40 Change in GHG emissions on increasing the housing percentage

From the economic viewpoint, increasing housing adds to running costs as a result of increased bedding and silage requirements and the farm activities associated with their cultivation. Increased housing also results in the higher capital cost of a larger digester, although this is partially compensated for by the increased electricity and heat production. The result is that increased housing reduces the profits for the farmer in both pre- and post- AD scenarios as shown in Figure 41.

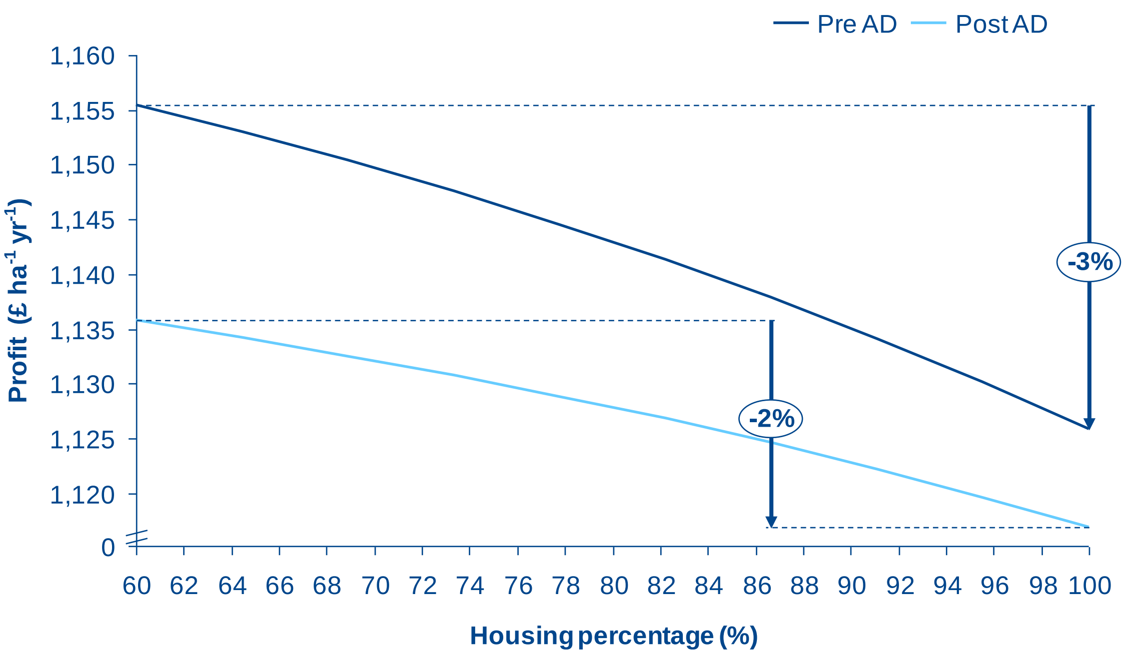

Figure 41 Net profit (first year) trend as housing percentage is increased

Marginal abatement cost results

The bigger drop in GHG abated amount (3% drop in post AD scenario) compared to a lower drop in profits (2% in post AD scenario) helps in reducing the overall MAC for the “Modelled farm” when housing percentage is increased as demonstrated in Figure 42.

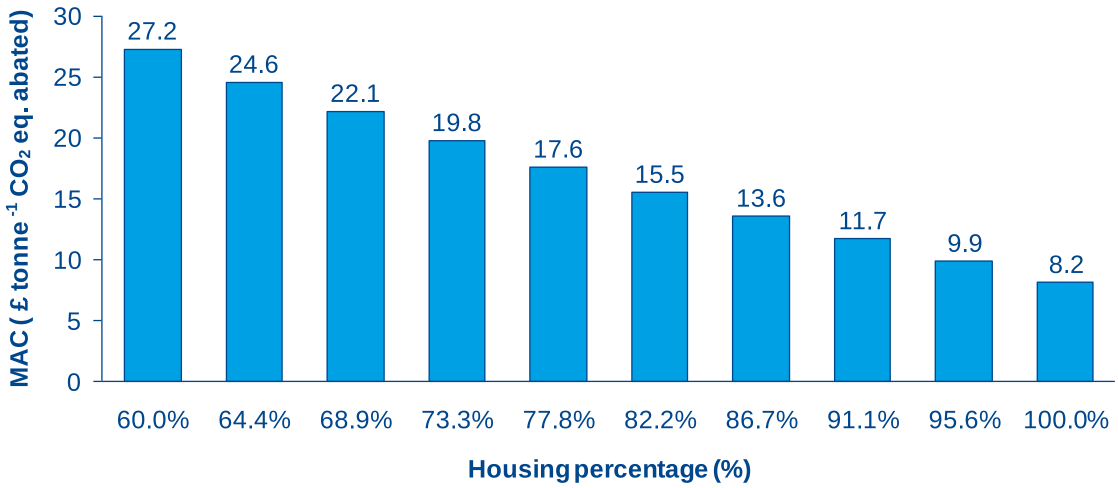

Figure 42 Change in MAC when housing percentage is increased

10.3.4 Specific methane yield

Specific methane yield is a characteristic of the slurry. In the pre-AD scenario, the amount of this potential yield that is released is defined by different methane conversion factors applied under various manure management methods as reported in Section 4.2. Post AD, the specific methane yield affects both the environmental and economic aspects of a farm.

Methane yield impacts the total GHG emissions in a pre-AD scenario by directly affecting the amount of CH_4 released for both grazed and housed cattle.

In the post-AD scenario, a higher yield results in increased CH_4 production which is captured and can be used by the CHP to produce additional electricity and heat, thus substituting more fossil fuel based energy. For example, an increase in specific methane yield from 0.13 to 0.15 m^3 CH_4 kg^{-1} VS added leads to an in increase in CH_4 production from 18,744 to 21,628 m^3 $year^{-1}$ in the “Modelled farm” scenario.

Any change in the amount of methane produced from each tonne of slurry affects the size of CHP unit required by the model and the resultant amount of electricity and heat generated. The net balance is economically efficient, however, as increased revenues from electricity and heat fully compensate the increased costs of the CHP unit. For example, using the "Modelled farm" data, a change in the specific methane yield from 0.13 to 0.15 m^3 CH_4 kg^{-1} VS added to the digester would lead to an increase in electricity production of 11,076 kWh $year^{-1}$ and heat of 15,076 kWh $year^{-1}$. The result is an increase in GHG abated from 675 to 766 kg CO_2 eq. ha^{-1} $year^{-1}$ and change in profit from -£37 to -£6 ha^{-1} $year^{-1}$. Hence the MAC changes from £54 $tonne^{-1}$ CO_2 eq. abated to £7 $tonne^{-1}$ CO_2 eq. abated.

Figure 43 shows the sensitivity of NPV and IRR to specific methane yield. Farmers can positively influence methane yield by using fresh slurry; old slurry has lower biochemical methane potential leading to a decrease in the biogas production (Bywater, 2011) and hence reduction in revenues.

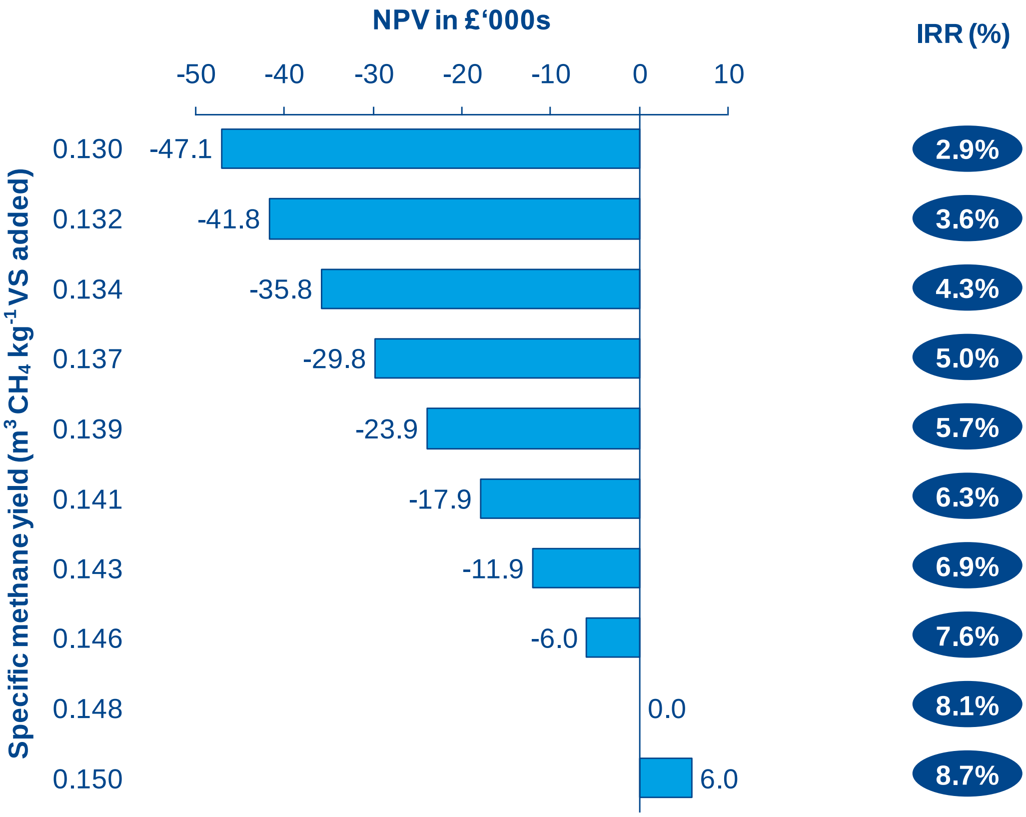

Figure 43 Sensitivity of NPV and IRR to specific methane yield

An optimal AD setup would aim to maximise the capture of specific methane yield. This can, however, be quite difficult to achieve as there are multiple operational steps in which the CH_4 yield is irrecoverably lost, such as during mixing of the slurry before it is fed into the digester or during CHP maintenance and repair downtime. It may be noted that there are various interventions which can help improve methane production. The focus of this research is understanding the impact if such an improvement is achieved rather than the drivers of the same. The sensitivity of the MAC to specific methane yield (assuming optimal biogas production) is shown in Figure 44.

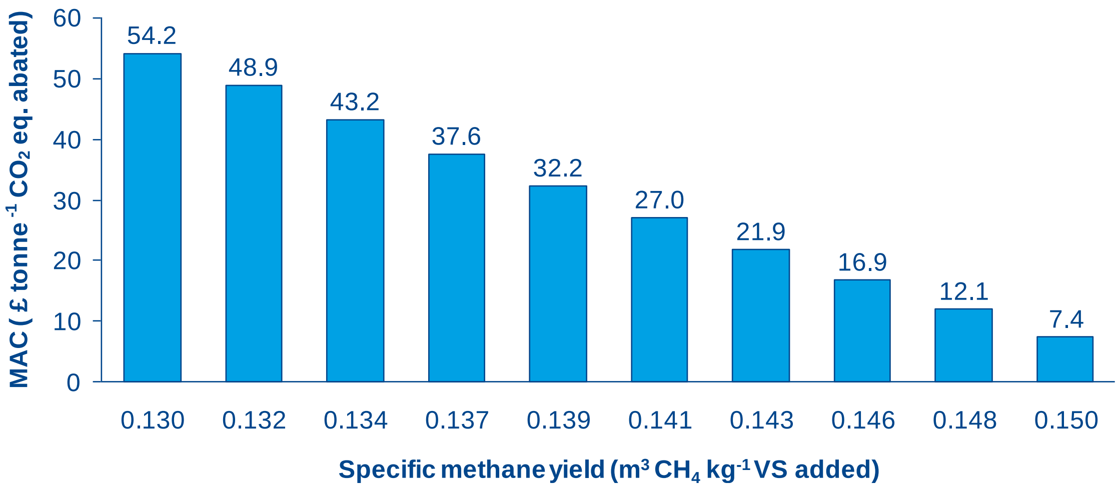

Figure 44 Sensitivity of MAC to specific methane yield

As expected, a higher specific methane yield implies a lower MAC. The MAC does not become negative for yields used in this sensitivity analysis but reduces from 54.2 tonne^{-1} CO_2 eq. abated to 7.4 tonne^{-1} CO_2 eq. abated. As discussed earlier, in spite of its environmental and economic advantages, higher specific methane yield is quite difficult to achieve in practice.

10.3.5 Organic Loading Rate

Organic loading rate directly affects the digester size and retention time as discussed in Section 3.5. These variables have a significant impact on the environmental and economic aspects of introducing AD to the farm. Digester size is a key determinant of the digester cost which is the highest capital cost component of an AD setup and must be chosen to optimise loading rate and retention time. For cattle slurry, 69% of the methane potential has been found to be achieved in the first 10 days (Cornell, 2011). The subset of methane potential that is not captured during digestion will need to be captured post-digestion, when the digestate is stored in air-tight enclosures. This could be a potential challenge but for the purpose of this analysis, it is assumed that the full methane potential is captured either in the digester or from the digestate storage. Table 27 presents the impact of change in organic loading rate on key metrics for the "Modelled farm" case.

Table 27 Change in digester size and cost when loading rate is varied

Loading rate (kg VS m^{-3} day^{-1})	1.5	2.5	3.5	4.5	5.5
Digester size (m^3)	290	174	124	97	79
Digester cost (£s)	128,110	89,642	70,854	59,439	51,660
Retention time (days)	43	26	18	14	12

For a given amount of slurry, a low loading rate requires a large digester which may be economically unfeasible from a cost perspective.

From the economic viewpoint, the digester size and hence digester capital cost declines rapidly with increase in organic loading rate. As presented in Table 27, an increase in organic loading rate from 1.5 kg VS m^{-3} day^{-1} to 5.5 kg VS m^{-3} day^{-1} leads to digester cost falling from £128,110 to £51,660, a decline of ~60%.

Taken all together, the interplay of lower GHG abated and lower costs results in a decrease in MAC with increase in organic loading rate. Figure 45 provides the change in MAC when organic loading rate is increased.

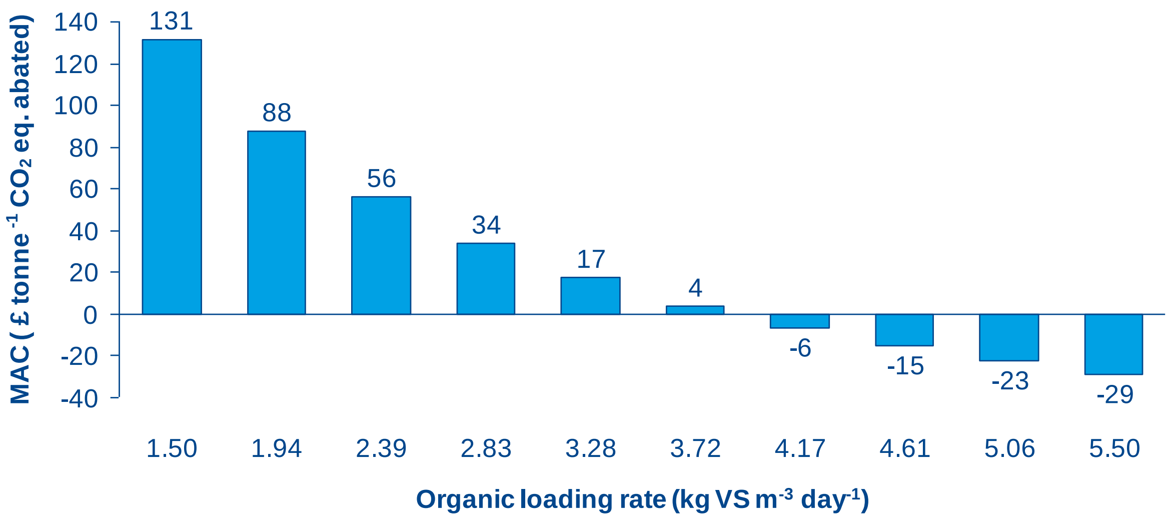

Figure 45 Impact of increasing organic loading rate on MAC

10.3.6 Livestock Density

The livestock density of a dairy farm is limited by the nitrogen that can be applied to the land as per the NVZ regulations (Defra, 2009). The average livestock density including followers in England is 1.6 livestock unit (LU) per hectare of farm land (Defra, 2011a). This number assumes a dairy cow represents 1 LU and a follower represents 0.6 LUs as discussed in Section 3.1. Higher livestock density leads to more slurry being collected and digested and hence more revenue from the sale of electricity. It also leads to higher capital cost due to a larger digester. On balance, the overall MAC reduces as livestock density is increased as shown in Figure 46.

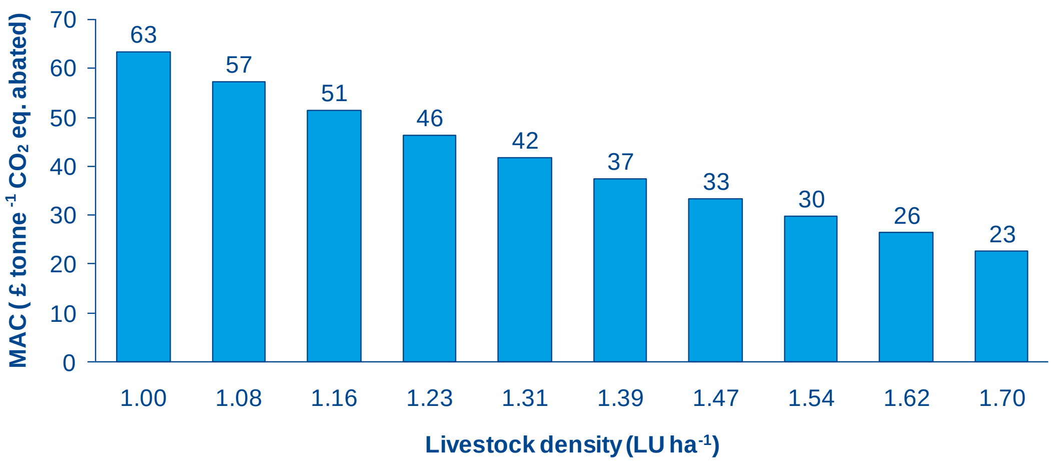

Figure 46 Change in MAC with varying livestock density

10.3.6.1 Higher livestock density beyond NVZ regulations

The “Modelled farm” case assumes the livestock density is equal to the UK average based on Defra (2011a).

Another potential case is that the amount of N that can be applied to the field governs the upper limit for livestock density. As ~70% of UK farms fall in NVZs, this assumption governs a large proportion of the farms. At a livestock density of 1.7 LU ha^{-1}, the N application rates that are based on the N excretion rate and the organic N application limit of 170 kg ha^{-1} are maximised. This case has been called “Modelled farm (NVZ limited)”.

A farmer may choose to maintain a higher livestock density for additional income and rent adjacent farm land to apply excess N. It may be noted that this rental of additional land is purely for the application of excess N. The nutrient needs of the cows would still be met by the original farm land which can support livestock density of up to 1.85 LU ha^{-1}. This scenario has been called “Modelled farm (Feed limited)” with the limiting factor being the feed that can be produced on the original farm land.

Another scenario is for farms that do not fall in NVZs. For these farms, the organic N application limit is 250 kg ha^{-1} thus allowing a livestock density of up to 2.5 LU ha^{-1}. This scenario has been called "Modelled farm (Non-NVZ)".

As the number of cattle increase in the same farm area, the financial impact from introduction of AD can be significant. For the "Modelled farm" size of 140 hectares, the financial performance of AD improved significantly by increasing livestock density. Table 28 provides a detailed comparison of key metrics as they change when livestock density is increased.

Table 28 Comparison of key metrics when livestock density is increased for the "Modelled farm"

Key metrics	Modelled farm (NVZ limited)	Modelled farm (Feed limited)	Modelled farm (Non-NVZ)
Farm size (ha)	140	140	140
Livestock density (LU ha^{-1})	1.7	1.85	2.5
Number of dairy cows	155	168	227
GHG emissions pre-AD (kg CO_2 eq. ha^{-1} $year^{-1}$)	7,435	7,745	9,419
GHG emissions post-AD (kg CO_2 eq. ha^{-1} $year^{-1}$)	6,659	6,903	8,275
GHG abated (kg CO_2 eq. ha^{-1} $year^{-1}$)	776	841	1,145
Profit difference (£s ha^{-1} $year^{-1}$)	-17.5	-14.5	1.8
Electricity+Heat revenue (£s $year^{-1}$)	10,443	11,422	16,017
- Electricity revenue (£s $year^{-1}$)	*10,755*	*11,642*	*15,729*
- Heat revenue (£s $year^{-1}$)	*-312*	*-219*	*288*
Marginal Abatement Cost (£s $tonne^{-1}$ CO_2 eq.)	23	17	-2
NPV (£s)	-14,031	-8,359	21,403
Internal rate of return (%)	6.8%	7.4%	9.7%

Increased cattle numbers, resulting from increased livestock density, leads to more emissions from both cattle (via enteric emissions) and manure. In the post AD scenario, the increased emissions from manure are largely captured and converted into heat and electricity. The net impact is that a larger amount of GHGs are abated and increased net revenues from electricity and heat production are earned. The interplay of these factors leads to an improvement in MAC as livestock density is increased; MAC reduces from £23 $tonne^{-1}$ CO_2 eq. to -£2.0 $tonne^{-1}$ CO_2 eq. as livestock density is changed from 1.7 to 2.5 LU ha^{-1}.

If the farmer has to pay rent on the additional land under the “Modelled farm (Feed limited)” scenario, the costs of rental could be substantial. In the “Modelled farm (Feed limited)” scenario, the farmer would need an additional 12 hectares of land to apply the excess N. It may be noted that this additional rented land is considered outside of the farm boundary and the slurry applied here is considered to be exported from the farm.

Cropland rental costs are £170 year^{-1} (Nix, 2012) leading to an additional burden of £15 ha^{-1} year^{-1} (based on the original farm size of 140 hectares only, not counting the additional rented land) which negates the profit increase of ~£2 ha^{-1} year^{-1} that the farmer may achieve by increasing the livestock density.

Consequently, unless the farmer has other economic benefits from the rented land, the increased revenue from AD alone would not justify the rental costs of neighbouring land. In this analysis the potential of additional income from crop revenues or other farm activities on the additional rented land have not been considered.

10.3.7 Renewable Heat Incentive

Renewable heat incentive is available to the farmers for all the heat based energy that is generated in surplus of the parasitic load of the digester and put to an eligible use as defined in Section 2.3.10.2. In the “Modelled farm” case, most of the heat generated (106,283 kWh year^{-1}) is consumed as parasitic load of the digester (98,732 kWh year^{-1}) while some is available (7,551 kWh year^{-1} or 53.9 kWh year^{-1} ha^{-1}) to fulfil part of the dairy usage. There is, however, no heat left to export. The amount consumed by the dairy attracts a RHI and provides incremental revenue to the farmer. The revenue is small given the current RHI price of 7.1 pence kWh^{-1} and even if the RHI prices were to rise, the incremental benefit to the farmer would be very limited compared to revenue from electricity as demonstrated in Figure 47.

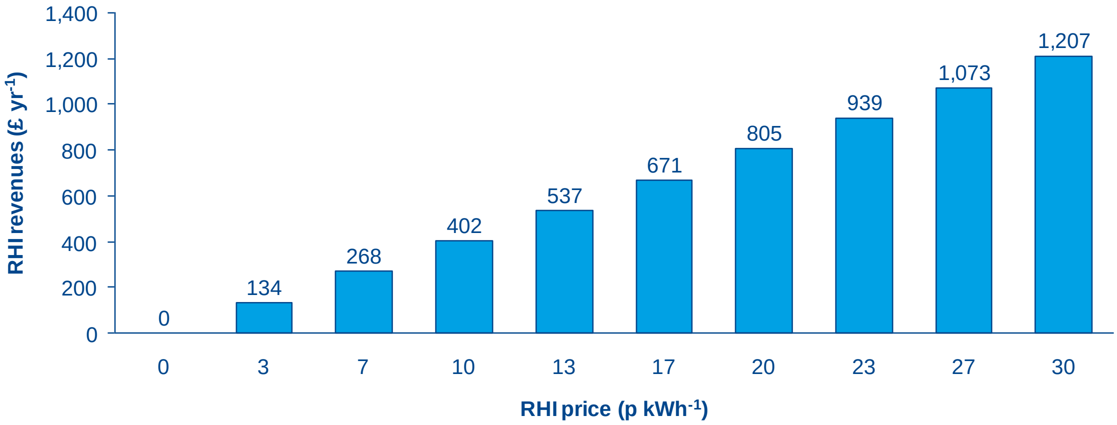

Figure 47 Change in revenues from heat generation with change in RHI

Given the current prices and subsidies, the farmer would benefit from installing a CHP unit with higher electrical efficiency (if this choice does not change the CHP capital cost). Though the electricity import price (10.2 pence kWh^{-1}) is lower than the LPG based heat import price (11.5 pence kWh^{-1}) thus making heat production more lucrative from a substitution perspective, the excess electricity attracts a much higher FIT of 14.7 pence kWh^{-1} against an RHI of 7.1 pence kWh^{-1} for excess heat. This dynamic means that in absence of FIT or RHI, the farmer would have benefitted from producing more heat on the farm as he would have saved money from an energy import substitution perspective. However, given that the FIT earned from the electricity surplus (14.7 pence kWh^{-1} plus 3.1 pence kWh^{-1}) is more than twice the amount earned on surplus heat in terms of RHI (7.1 pence kWh^{-1}), the farmer is significantly incentivised to maximise electricity production. Figure 48 demonstrates the energy surplus dynamic if the electrical efficiency of the CHP unit is varied to favour either more electricity production or more heat production.

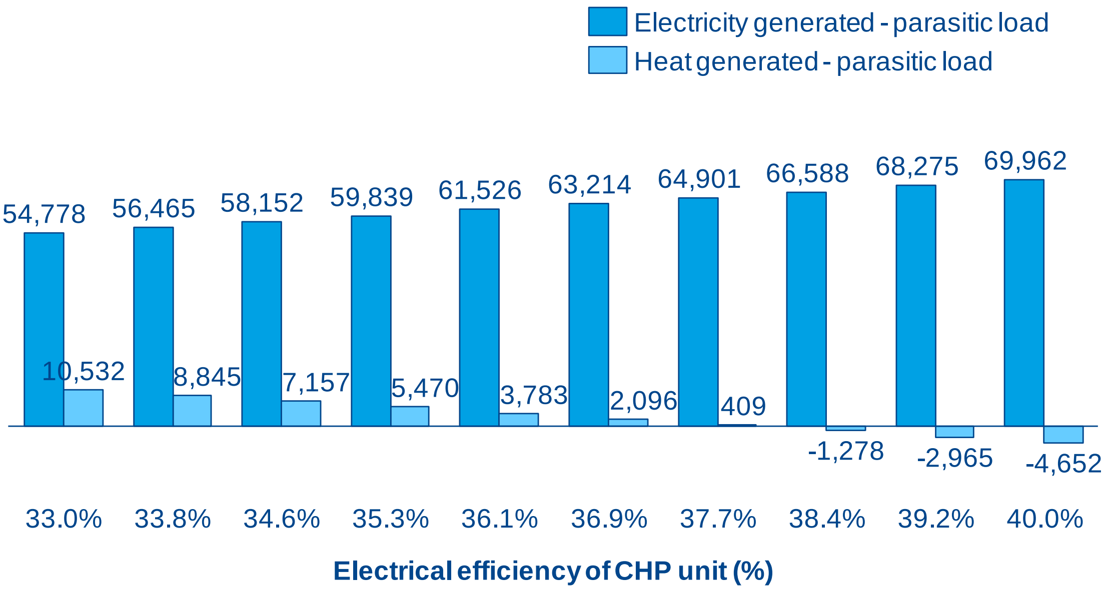

Figure 48 Electricity and heat generated (above parasitic load) (kWh $year^{-1}$) if electrical efficiency of the CHP unit is varied

10.3.8 Fugitive Emissions

Fugitive emissions during the operation of an AD plant represent the leakage in the system as described in Section 2.3.7.3. These reduce both the environmental and economic benefits of introduction of AD on the dairy farm.

From an environmental perspective, fugitive emissions negate the benefit of introduction of AD. As part of the digestion process, more CH_4 is generated than without a digester. The biogas is, however, captured and converted into usable energy by the CHP unit in a well-functioning setup. If there are leaks and this CH_4 is released into the atmosphere, the higher GWP of CH_4 can outweigh the environmental benefits of fossil fuel based energy substitution as demonstrated in Figure 49.

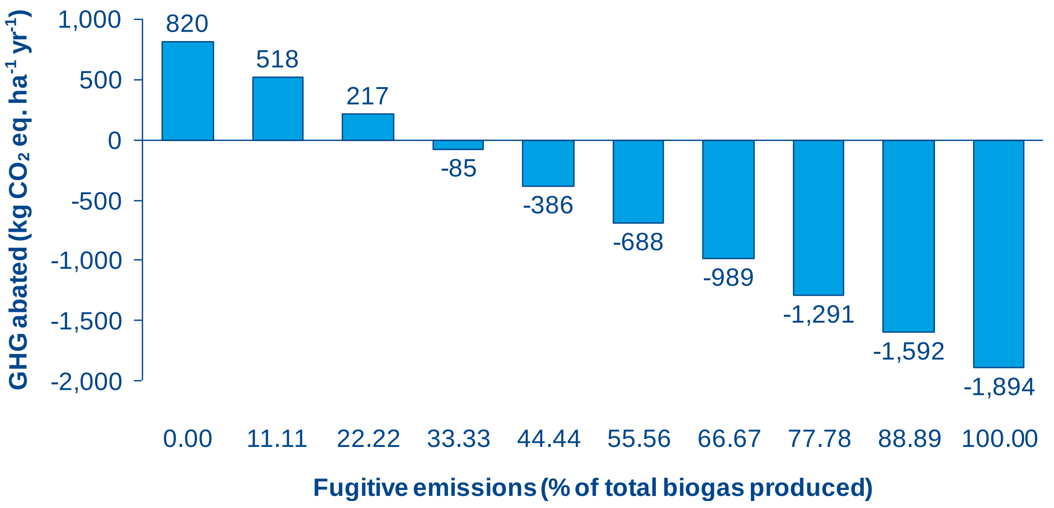

Figure 49 Change in GHG abatement as fugitive emissions vary

If unmanaged, fugitive emissions can even "produce" emissions compared to the pre-AD scenario. For the "Modelled farm", if fugitive emissions exceed 22%, the farm has more emissions in the post-AD scenario than the pre-AD scenario.

Any fugitive emissions also reduce the amount of biogas that goes into CHP and hence reduce the amount of heat and electricity that are generated. As a result, higher fugitive emissions would reduce electricity and heat revenues and reduce the NPV as demonstrated in Figure 50.

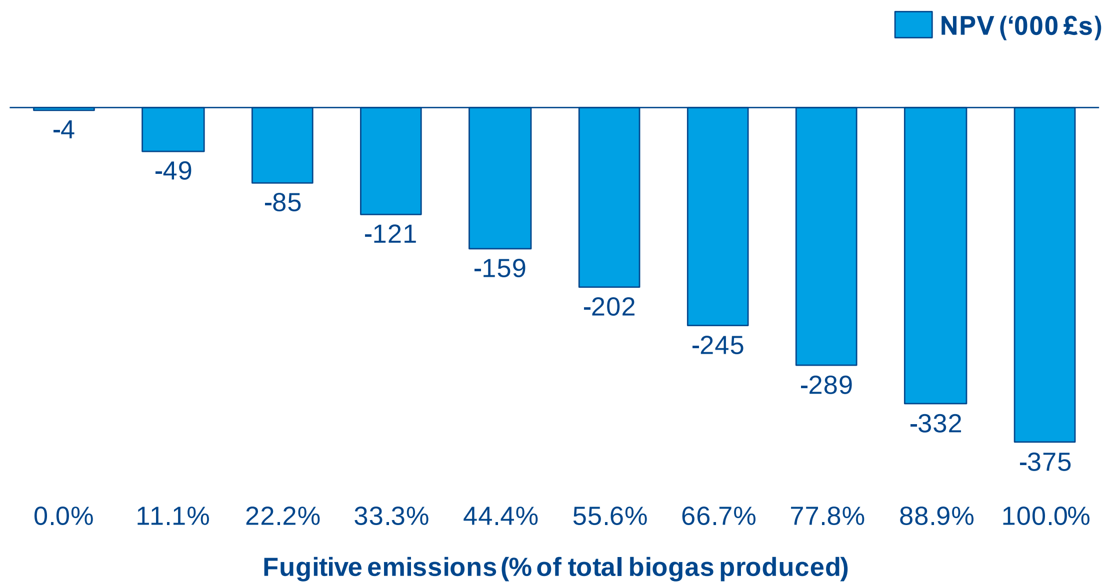

Figure 50 Change in NPV when fugitive emissions vary

10.3.9 Total solids load

The total amount of manure excreted by the cows and followers is calculated using a manure excretion rate of 19.3 and 14.6 tonnes year^{-1} for dairy cows and other cattle, respectively (Defra, 2010b). Total solids represent the dry matter in the manure and have been assumed to be 8% (Nijaguna, 2002). It is assumed that the percentage of volatile solids as a proportion of dry matter is constant for all cases considered. As a result, a change in the total solids concentration results in a change in the total amount of volatile solids. This will lead to variation in the potential methane produced affecting energy production and the overall economics. The total solids produced per LU may vary with change in the diet of the cow, the season, farming practices and manure collection methods. Hence, sensitivity of MAC to total solids has been studied.

In the pre-AD scenario, a high concentration of solids could lead to difficulties in pumping and spreading of slurry (due to crust formation). Crusting of slurry in open slurry storage tanks could also lead to N_2O emissions thus worsening the carbon footprint of the farm (discussed in further detail in Section 8.2). Additionally, the penetration of slurry into the soil decreases if the total solids concentration is high. If the solids are too low the slurry may cause anaerobic conditions in the soils on application leading to increased methane emissions.

Similar observations are made with digestate.

Within the suggested range of 7-9% by Nijaguna (2002), the farm environmental and economic performances in the post-AD scenario improve with increase in total solids. If the digester size is based on loading rate then Increasing or decreasing the total solids (and the equivalent volatile solids) changes the digester size, the retention time, the amount of biogas captured and, therefore, the heat and electricity produced. This impacts both the emissions abated by the digester and the profits made from it.

An increase in total solids (and therefore volatile solids) leads to increased biogas production, increased CHP capacity, and further substitution of fossil fuel based heat and electricity thus increasing the GHG abated. Some of this GHG abated is negated by the increased fugitive emissions from higher biogas production and the increased embodied carbon from a larger digester. The emission benefits from substitution of fossil fuel based energy, however, outweigh the increased emissions from other sources as shown in Figure 51.

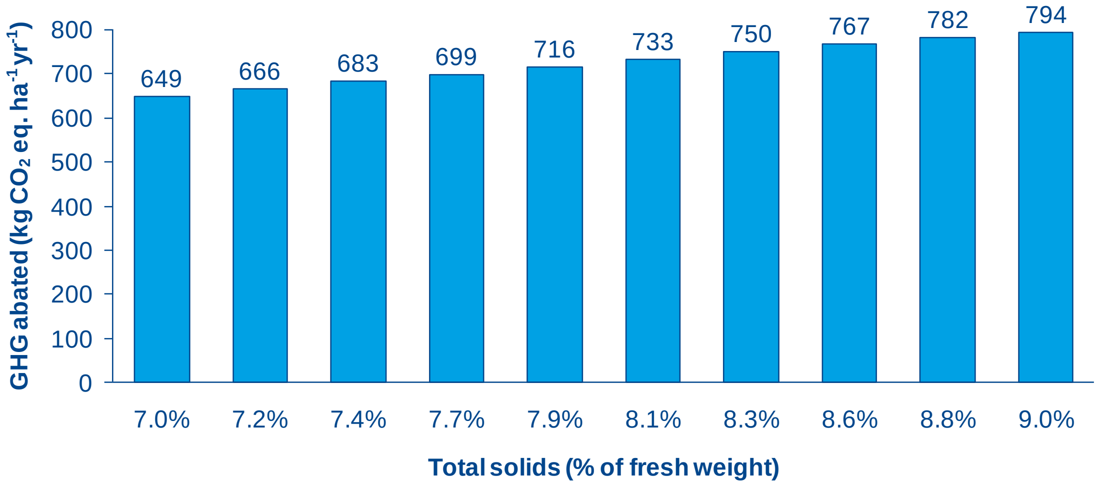

Figure 51 Change in GHG abated with change in total solids

From an economic viewpoint, more volatile solids imply a bigger digester and a bigger CHP unit and hence higher capital costs. These increased expenditures are, however, more than compensated by the increased revenues from higher production of heat and electricity due to more biogas being converted. Taken all together, higher GHG abated and improved revenues imply a higher NPV for the farmer and thus a lower MAC as demonstrated in Figure 52.

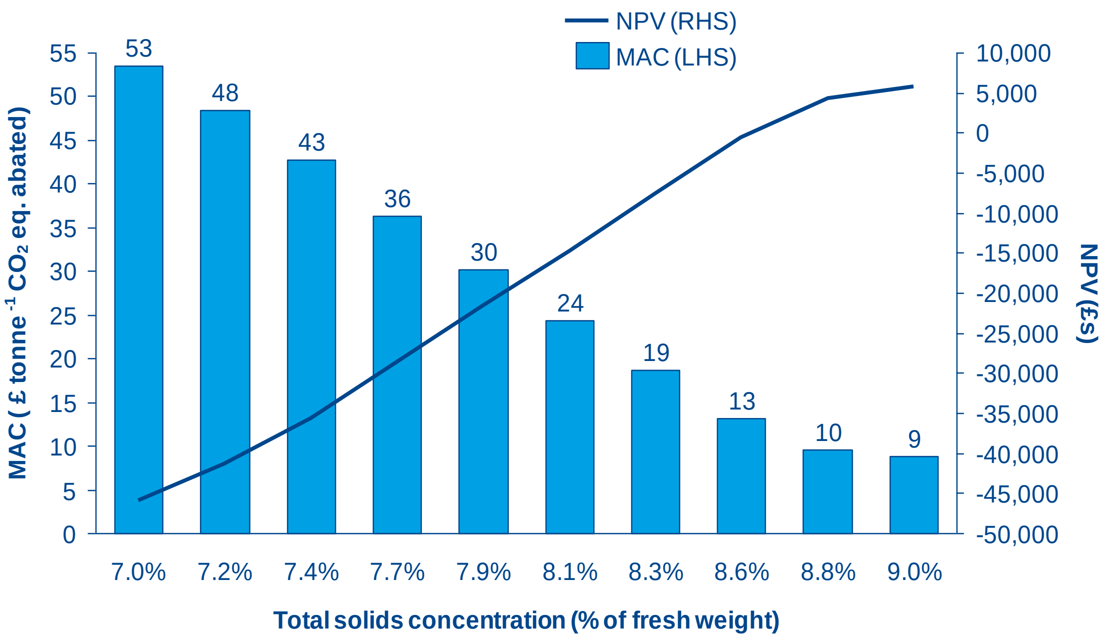

Figure 52 Effect of change in total solids concentration on MAC

10.3.10 Overall summary of sensitivity analysis

The environmental benefit of AD (on a per hectare basis) of slurry increases with increase in specific methane yield and livestock density. It is partially negated by the fugitive emissions which if uncontrolled, can take away a large part of the AD benefits. The farm size has a marginal effect on the emissions abated per hectare but has a favourable impact on the economic aspects of introducing AD on the farm. Introducing AD on a farm is economically beneficial in more intensive farming setups e.g. a larger farm allows for lower capital cost per m^3 and as a result a more favourable net economic result.

Considering both environmental and economic aspects together, it is seen that MAC varies the most with a change in FIT rates. Farm size and organic loading rate also have a material impact on MAC, reflecting the interplay of economic and environmental benefits.

The sensitivity analyses discussed earlier in this chapter provide a detailed overview of the impact on environmental and economic aspects on change of one variable at a time. The next chapter presents the effect of changing multiple of these variables concurrently on the impact of introducing AD.

10.4 Monte Carlo simulation

From the Monte Carlo simulation, the range of outcomes that various different dairy farming scenarios may imply can be understood. The base scenario is the “Modelled farm” and for the five variables considered for Monte Carlo simulation, all permutations of changing these five variables across 5 values in their respective expected range are considered.

The rest of this chapter is divided into multiple sections to discuss the various results:

- Section 10.4.1 – This section provides commentary and discussion on results obtained for the GHG abated, heat surplus, electricity surplus and MAC values.
- Section 10.4.2 – This section discusses the values of input parameters in the synthetic scenario that leads to highest and lowest post-AD GHG abated outputs - “boundary scenarios”. The relation between input

parameters with the GHG abated output values is discussed and explanation for the boundary conditions is provided.

- Section 10.4.3– Similar to the previous section, this one discusses the values of input parameters in the synthetic scenario that leads to highest and lowest NPV outputs. Once the “boundary scenario” input parameters are identified, their relation with NPV values is discussed and explanation for the boundary conditions is provided.
- Section 10.4.4– This section takes a similar approach to the above two sections for MAC results.
- Section 10.4.5– Comparative analysis of anaerobic digestion of slurry with other renewable energy or low carbon technologies is undertaken in this section using levelised costs and carbon content of unit electricity as indices.

10.4.1 Profiles of GHG abated, heat surplus, electricity surplus and MAC

In this section, the full profile of GHG abated, heat and electricity produced in the post AD scenario and resultant MAC in 3125 synthetic UK farm scenarios that are studied under the Monte Carlo simulation is discussed. A summary of the results of Monte-Carlo simulation is presented in Table 29.

Table 29 Summary results from Monte Carlo simulations

Results	GHG abated (kg CO_2 eq. ha^{-1} $year^{-1}$)	Heat surplus (kWh $year^{-1}ha^{-1}$)	Electricity surplus (kWh $year^{-1}ha^{-1}$)	MAC (£s $tonne^{-1}$ CO_2 eq.)
Mean	761	-47	271	46
Median	741	-48	260	33
Standard Deviation	186	58	88	51
Skew	0.33	(0.01)	0.49	0.88
Minimum	388	-218	107	-35
Maximum	1,226	136	522	243

The frequency distribution for GHG abated under the various scenarios is shown in Figure 53.

Marginal abatement cost results

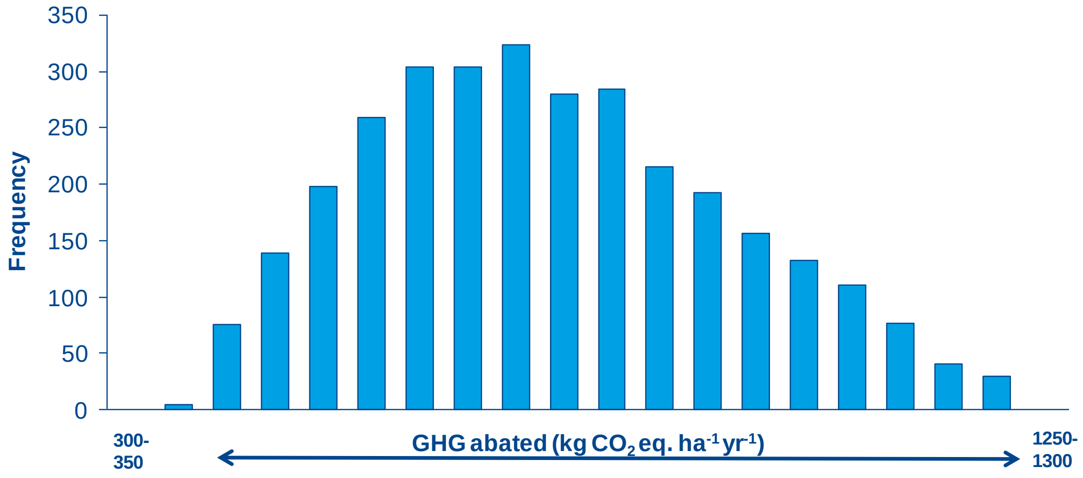

Figure 53 GHG abated distribution under Monte Carlo simulations

The frequency plot of GHG abated is a positive skew Gaussian curve with a mean at 761 kg CO_2 eq. abated ha^{-1} year^{-1}. The frequency distribution demonstrates that the net GHG abated under the studied range of farm scales, setups and operating conditions remains positive. Although the range varies between 388 kg CO_2 eq. abated ha^{-1} year^{-1} and 1,226 kg CO_2 eq. abated ha^{-1} year^{-1}, the positive result under each of those configurations is a key contribution to GHG abatement resulting from the introduction of AD on a dairy farm.

Heat and electricity production play a major role in the GHG abatement and in the financial feasibility of AD. In order to explore this further, the production of these have been studied in detail. Figure 54shows the frequency plot of each of these on the same axis.

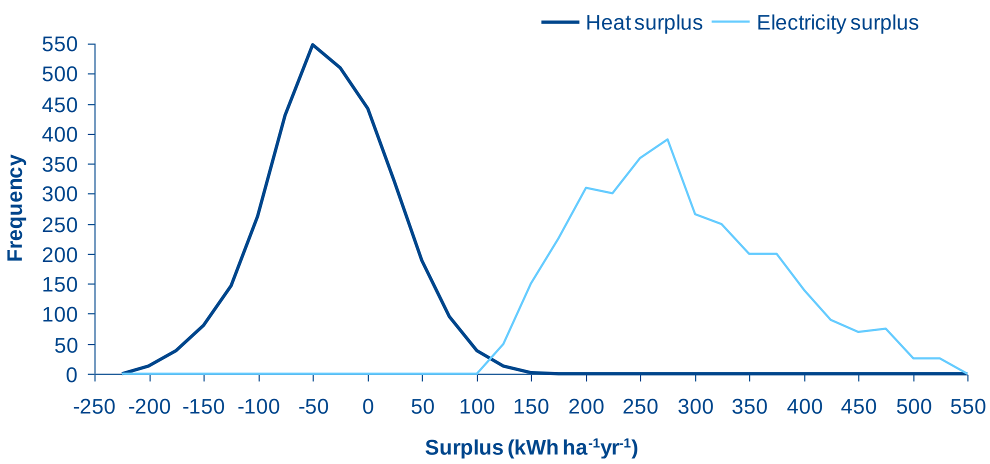

Figure 54 Monte Carlo simulations – Heat and electricity surplus

Though both the distributions of heat and electricity surplus (after accounting for parasitic load and dairy usage) are Gaussian, they peak (highest frequency point) at very different points on the x-axis, 250 to 275 kWh ha^{-1} $year^{-1}$ for electricity and -75 to -50 kWh ha^{-1} $year^{-1}$ for heat. Electricity surplus was found to be always positive and the heat surplus was negative under 78% of the scenarios.

The heat and electricity produced are correlated inversely for a given amount of biogas. If the electrical efficiency of the CHP unit is high, more electricity would be produced but less heat, because the total energy production is limited by the amount of biogas. In any of the 3,125 modelled scenarios, a part of the economic benefit of electricity production is found to be negated by cost of the heat deficit.

The interplay of the GHG abated and revenue difference from electricity and heat production leads to a Gaussian MAC distribution as represented below in Figure 55.

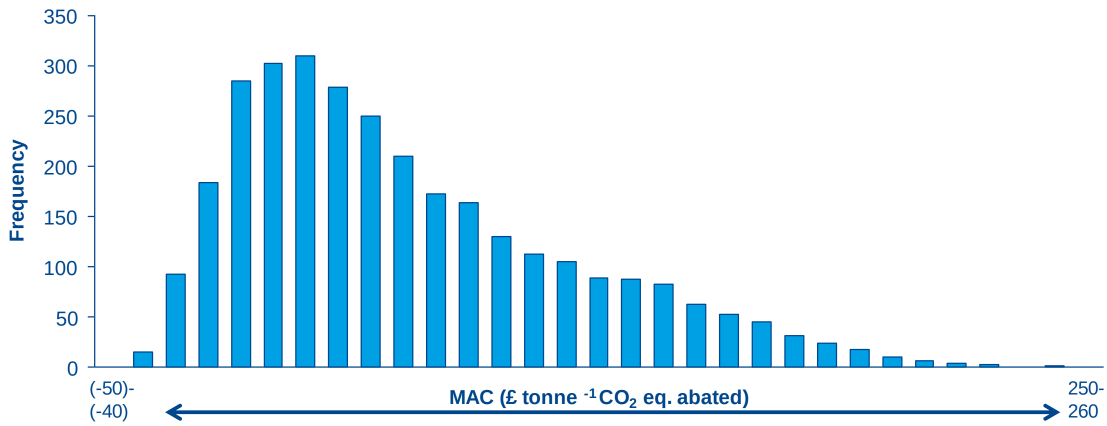

Figure 55 Distribution of MAC under Monte Carlo simulations

The mean for the MAC distribution, presented in Figure 55, is £46 $tonne^{-1}$ CO_2 eq. abated with a standard deviation of 51. This mean value is lower than that calculated for the "Modelled farm" MAC of £27 $tonne^{-1}$ CO_2 eq. abated and the positive skew of the distribution implies that in a larger number of the scenarios, MAC is positive. A positive MAC is essentially a cost borne by the farmer for GHG abatement. This on its own makes introduction of AD an unprofitable enterprise for a large proportion of farms in the UK, unless further incentives are introduced based on the GHGs abated in addition to the current ones based on renewable energy produced.

In 82% of the 3,125 modelled scenarios, the MAC turns out to be positive implying a cost to the farmer for GHG abatement. MAC is negative under the remaining 18% of the scenarios, implying a net profit from GHG abatement. This is typically the case for larger farms (>150 hectares). Under these 18% of the modelled scenarios, farmers make a profit by taking advantage of the FIT related subsidy by introducing AD on the farm.

To achieve a negative MAC, a farm size of 150 hectares would require a specific methane yield of 0.14 m^3 CH_4 kg^{-1} VS added or higher along with a high housing percentage (>80%) and a high organic loading rate (>3.25 kg VS m^{-3} day^{-1}). As discussed in various sensitivity analyses, the farmer would need to optimise most variables if the farm size is small.

For a bigger farm (for example, 250 hectares), the MAC can be negative under many different scenarios. For example, for a 250 hectare farm, if the specific methane yield is 0.15 m^3 CH_4 kg^{-1} VS added, the livestock density can be as low as 1 LU ha^{-1}.

10.4.2 GHG abated boundary scenarios

From an environmental viewpoint, higher electricity sale would suggest a higher fossil fuel substitution leading to higher GHG abatement while other considerations such as fugitive emissions and embodied carbon in a bigger digester may negate the benefit of some of the fossil fuel substitution. The balance of these various factors could result in an economically sub-optimal but environmentally optimal set of parameters, as is the case here. The cases resulting in the minimum and maximum GHG abated and the input parameters that lead to these boundary conditions are shown in Table 30.

Table 30 GHG abatement boundary scenario input parameters

	Min scenario	Max scenario
GHG abated (kg CO_2 eq. ha^{-1} $year^{-1}$)	388	1226
NPV (£s)	-49,239	104,637
MAC (£ $tonne^{-1}$ CO_2 eq. abated)	243	-26
Farm size (ha)	50	200
Livestock Density (LU ha^{-1})	1.0	1.7
Housing percentage (%)	60%	100%
Organic loading Rate (kg VS m^{-3} day^{-1})	2.5	3.5
Specific methane yield (m^3 CH_4 kg^{-1} VS added)	0.130	0.150

The range of GHG abated observed under Monte Carlo simulation is 388-1,226 kg CO_2 eq. ha^{-1} $year^{-1}$, implying a significant environmental contribution under all farming scenarios. This shows a consistent GHG abatement advantage of AD.

The GHG abated minimum scenario occurs under the least intensive farming setup used in the simulation. The parameters for the maximum GHG abatement scenario are a mirror image of those for the minimum GHG abatement. This reflects the unidirectional nature of the GHG abatement relationship with the key input parameters that were varied under the Monte Carlo simulation.

10.4.3 NPV boundary scenarios

The NPV boundary scenarios occur under different input values as compared to GHG boundary scenarios, as shown in Table 31. NPV is purely an economic metric and at times may not be optimal from an environmental perspective. MAC, in comparison, captures the cost of per unit GHG emission abatement and is able to balance both the environmental and economic aspects of running a digester.

Based on the Monte Carlo simulation, it was found that the financial effects could outweigh the environmental considerations in many scenarios and under the current incentive structures may lead the farmer to run sub optimal AD setups from an environmental perspective.

Marginal abatement cost results

Table 31 NPV boundary scenario input parameters

	Min scenario	Max scenario
NPV (£s)	-74,882	156,449
MAC (£ $tonne^{-1}$ CO_2 eq. abated)	85	-34
GHG abated (kg CO_2 eq. ha^{-1} $year^{-1}$)	629	1,226
Farm size (ha)	150	250
Livestock Density (LU ha^{-1})	1.0	1.7
Housing percentage (%)	100%	100%
Organic loading Rate (kg VS m^{-3} day^{-1})	2.5	3.5
Specific methane yield (m^3 CH_4 kg^{-1} VS added)	0.130	0.150

The boundary conditions of NPV are represented by the least and most intensive setups from a specific methane yield perspective. As the amount of methane produced is directly linked to specific methane yield, it directly impacts the heat and electricity production and corresponding revenues. Of the other metrics, the digester cost and heat/electricity revenues balance under difference combinations and the highest and lowest NPVs are achieved at intermediary values and not the extreme ends of the value range.

10.4.4 MAC boundary scenarios

The input variables leading to the MAC boundary scenarios under the Monte Carlo simulations are shown in Table 32.

Table 32 MAC boundary scenario input parameters

	Min scenario	Max scenario
MAC (£ $tonne^{-1}$ CO_2 eq. abated)	-35	243
NPV (£s)	103,612	-49,239
GHG abated (kg CO_2 eq. ha^{-1} $year^{-1}$)	821	388
Farm size (ha)	250	50
Livestock Density (LU ha^{-1})	1.7	1.0
Housing percentage (%)	60%	60%
Organic loading Rate (kg VS m^{-3} day^{-1})	3.5	2.5
Specific methane yield (m^3 CH_4 kg^{-1} VS added)	0.150	0.130

It may be noted that the minimum MAC scenario is not the best scenario for the farmer from a purely economic standpoint as NPV is higher in some other scenarios as demonstrated in Table 32. Minimum MAC ensures that the farmer is getting the best compensation for abating each unit of GHGs. Similarly the maximum MAC scenario does not represent the highest loss configuration for the farmer, which was represented by lowest NPV scenario as discussed in the previous section. It is simply the highest cost that the farmer may have to bear for abating each unit of GHG.

Though the highest MAC occurs under the least intensive farming setup modelled, the lowest MAC scenario is under most intensive setup for all variables except housing percentage. This is driven by a balance of reduced unit digester cost and increased crop production costs (higher housing percentage leads to more silage requirements). Due to the dynamic of these two competing variables, for a farm size of 250 hectares and under intensive farming conditions as described by the highest value of other variables modelled, increase in housing percentage leads to an increase in MAC.

10.4.5 Comparative analysis with other technologies and other sources

Due to the limited data available on the MAC of AD, comparative analyses of slurry based digesters with other renewable energy technologies were conducted using the following two indices:

- Levelised cost or the cost of producing unit electricity
- Carbon content of unit electricity produced via AD on a dairy farm

The results of these analyses for AD are presented in Table 33.

Table 33 Summary results of levelised cost and carbon content of electricity from Monte Carlo simulation

Results	Levelised cost (£ MWh^{-1})	Carbon content (g CO_2 eq. kWh^{-1})
Mean	71	-362
Median	68	-362
Standard Deviation	15	13
Skew	0.82	0
Minimum	43.2	-388
Maximum	129.5	-334

10.4.5.1 Levelised cost comparison

The levelised cost from the Monte Carlo simulation has a mean of £71 MWh^{-1} with a standard deviation of £15 MWh^{-1} and a 90% confidence interval of £41-101 MWh^{-1}.

Marginal abatement cost results

The levelised cost range of £41-101 MWh^{-1} obtained from the model lies on the lower end of DECC estimates for AD as shown in Table 34. The DECC estimates include various kinds of inputs like slurry, energy crops and food waste which may explain the broader and higher range. The Monte Carlo simulation results are comparable to the Mott McDonald result of £101 MWh^{-1}.

Table 34 Levelised costs of various technologies

Technology	Levelised cost (£ MWh^{-1}) (Mott MacDonald, 2011)	(DECC, 2011) 2010 prices
Solar PV	343 – 378	202-380
On-shore wind	83-90	75-127
Off-shore wind	169	155-196
Nuclear	96	
Dedicated biomass		127-154
Biomass co-firing		82-105
Biodiesel		288-357
Advanced Conversion Technologies		(35)-80
Landfill Gas		39-50
Sewage gas		57-122
Energy from Waste		(52)-11
Hydro	69	67-215
AD Slurry	101	75-194 (21 years, 84% load factor)
AD Energy Crops	171	
AD Food waste	147	
Geothermal	159	132-341

Some waste-to-energy technologies like landfill gas and sewage gas have a much lower levelised cost compared to on-farm AD which suggests a more cost efficient renewable energy potential from these technologies. This is largely driven by the fact that infrastructure already exists for the capture of the waste gas and hence, minimal additional infrastructure is required for utilising it while none is required for its production.

Levelised cost of AD is significantly lower than estimates of popular renewable energy technologies like solar and off-shore wind, which are £202-380 MWh^{-1} and £155-196 MWh^{-1}, respectively. It is, however, comparable to the levelised costs of the on-shore wind technology from the two sources considered above in Table 34.

Taken all together, from the levelised cost comparison perspective, AD on a dairy farm would fall in the lower quartile. This makes it an attractive investment area for meeting renewable energy goals.

10.4.5.2 Carbon footprint comparison

AD on dairy farms is very effective in abatement of GHGs from manure management. As a result the net impact of the overall setup is a negative carbon cost of producing electricity from this technology, a fact that distinguishes AD from most other renewable energy technologies.

As per Table 33, the carbon content of energy produced by introduction of AD on a dairy farm has a mean of -362 g CO_2 eq. kWh^{-1} (negative result implying carbon abatement) with a standard deviation of 13 g CO_2 eq. kWh^{-1} and a 90% confidence interval of -388 to -336 g CO_2 eq. kWh^{-1}. A comparison of the carbon content with other technologies is shown in Table 35.

Table 35 Carbon content of various technologies

Fuel	Parliamentary Office of Science and Technology (2011)(g CO_2 kWh^{-1})	DECC (2012f) (t CO_2 GWh^{-1} electricity supplied)
Coal	786-990	912
Gas	365-488	392
Solar PV	75-116	
On-shore Wind	20-38	
Off shore wind	9-13	
Nuclear	26	
Hydro	2-13	

The carbon footprint of coal is the highest of the technologies considered here. Traditionally, coal has been the largest provider of electricity for the UK and is increasingly being phased out by gas based electricity, a technology with about half the carbon footprint.

Among the renewable/low-carbon technologies considered, solar PV has a high carbon footprint owing to significant fossil fuel based energy consumption in producing solar PV cells. On the other hand, the materials required to produce a digester and CHP are widely available and the manufacturing process for these tends to be less energy intensive, which helps limit the embodied carbon content of an AD setup, keeping the GHG footprint low. In fact, the embodied carbon in AD is more than compensated for by the GHG abatement from AD, leading to a negative GHG footprint in all the modelled cases. This makes AD a strong candidate for prioritisation with regards to investment towards GHG abatement.

10.5 Case study: Hillsborough digester

Validation of the emissions model and assumed variables was conducted using data from the demonstration anaerobic digester at Agri-Food and Biosciences Institute (AFBI) Hillsborough, Ireland. The digester was designed, supplied and constructed by BiogenGreenfinch, Ludlow, Shropshire in 2007-08. It is a continuously stirred steel digester of size 660 m^3 operating in the mesophilic temperature range. Cattle slurry was digested for the first 27 months of its operation.

As very limited data is available on the Hillsborough digester operations, the comparative analysis is limited on many of the economic and emissions aspects. Table 36 provides a high level comparison of key metrics.

Table 36 Comparative analysis of Hillsborough digester empirical data and modelled outputs

Input parameter	Hillsborough digester empirical data	Model inputs
Size of digester (m^3)	660	578
Slurry digested (tonnes $year^{-1}$)	7,300	7,295
Total solids (% of fresh weight)	6.9 %	6.9%
Volatile solids (% of TS)	77%	77%
Operating temperature (C)	37.1	37
Overall efficiency (%)	78%	78%
CHP Size (kW)		78
Electrical efficiency (%)	27%	27%
Thermal efficiency (%)	51%	51%
Loading rate (kg VS m^{-3} digester day^{-1})	2.02	2.02
Retention time (days)	27	26.3
Capital cost (£s)	Not Known	£207,695 (digester) + £77,237 (CHP)
Methane yield (m^3 CH_4 kg^{-1} VS)	0.16	0.16
Electrical parasitic load (kWh $tonne^{-1}$ slurry)	5.4	5.4

Marginal abatement cost results

Results parameter	**Hillsborough digester empirical data**	**Model outputs**
Electricity produced (kWh $year^{-1}$)	167,624 (67% CHP uptime)	178,649
Heat Produced (kWh $year^{-1}$)	313,384	337,448
Thermal parasitic load (kWh $tonne^{-1}$ slurry)	32	43.8

There are some minor differences in the input variables between input parameters and the Hillsborough empirical data as some of the model inputs shown in the table are intermediate derived variables. For example, the digester size in model case is derived based on the total amount of slurry digested as the primary input. The comparative results may vary if a different starting assumption is made.

The results in the modelled outcome are very close to the empirically observed data. The electricity and heat produced as calculated by the model are within 5% and 10% of the empirical data. The difference between measured and calculated values can be attributed to operational losses and other minor configuration differences.

The thermal parasitic load from the model is roughly 38% higher than the empirically observed value. The average temperature of site has not been reported for the period of operation when slurry was digested. Hence, the model is based on an average UK temperature of 8.8 °C (The Met Office, 2013). This may have led to the deviation of the calculated value of thermal parasitic load from that measured.

The difference in thermal parasitic load may also be attributed to the different materials and thicknesses assumed under the two cases. The Hillsborough digester is an epoxy coated steel tank with 100 millimetre (mm) of mineral wool installation and a 1 mm plastic coated steel outer protection. The model, on the other hand, assumes a concrete digester with polyurethane insulation with thicknesses of 300 mm and 60 mm, respectively.

Though, other economic and environmental metrics are not available for the Hillsborough digester, key metrics from the model as per the above mentioned setup in Table 37 are provided for reference.

Table 37 Key metrics from modelling of Hillsborough digester

Key metrics	
GHG abated (kg CO_2 eq. ha^{-1} $year^{-1}$)	659
Marginal Abatement Cost (£s $tonne^{-1}$ CO_2 eq.)	7

11.Conclusions and future work

The introduction of anaerobic digestion on a dairy farm is effective in reducing emissions from manure management and can be beneficial in reducing the GHG footprint of the dairy farming industry. Based on this research, the technology can reduce the GHG footprint of manure management on most UK dairy farms by 10%. This abatement includes a reduction in CH_4 and N_2O emissions from the dairy farm as well as the reduced emissions by substitution of fossil fuel based electricity and heat.

To maximise the environmental and economic potential of the introduction of AD on a farm, the farmer needs to optimise the use of biogas produced which is a material factor in the overall impact. "Flaring" of the biogas produced or using the biogas to satisfy local heating needs are simpler to implement for the farmer but economically loss making and hence not advised. "Biogas upgrade" is a high capital cost proposition and is currently not suitable from an economic perspective for the UK dairy farms. Based on this research, using a CHP unit to convert the biogas produced into heat and electricity is the most effective technology and of the technologies considered, has the best economic and environmental potential to be used for dairy farm based AD setups.

The introduction of CHP can provide additional benefits by reducing farm operating costs and has the potential of generating incremental revenues. The initial capital expenditures of installing a digester and a CHP unit are significant deterrents for the farmer and potentially the primary reasons for low adoption of AD in the UK. Based on this research, under the current subsidy framework, the majority of the UK dairy farms are likely to make a loss by introducing this technology. The Monte Carlo simulation indicate that the UK farmer is expected to incur a MAC of £46 ± £51 $tonne^{-1}$ CO_2 eq. abated, which would be essentially the cost the farmer is incurring to abate GHG gases.

This research has studied the impact of various variables linked to farming practices and some of these can be optimised to reduce the MAC for the farmer. Of the key farming practice related variables for which a detailed single variable sensitivity analysis was conducted, it was found that a higher organic loading rate, high specific methane yield, high livestock density and higher solids production are key factors that help reduce the MAC and bigger farms are more likely to have a lower MAC compared to smaller farms.

It may be noted that a holistic evaluation of the farm should be undertaken and this study does not suggest an "optimal" farming setup but merely provides an impact assessment of certain farming practices on MAC. For example, for a certain farm setup, a higher housing percentage may reduce the MAC, but social and animal welfare considerations should be taken into account before choosing to increase housing for the cattle.

It is also important to mention that achieving the optimal output in certain cases, though desirable, may not easily be feasible from a practical perspective. For example, a higher specific methane yield is not necessarily easy to obtain, though it would be helpful in reducing the MAC.

The current subsidy framework in terms of FIT is very effective in reducing the MAC, though the benefit of subsidy in the form of RHI is limited as a low amount of heat (in excess of parasitic load) is produced from introduction of AD. Current levels of FIT are not enough to make introduction of AD a profitable proposition for 75% of the modelled farms and hence at current FIT levels, it is unlikely that adoption of slurry based digesters would rise in the UK.

The FIT framework is designed to drive maximum production of electricity rather than a reduction in GHG footprint which is where the real benefit of AD lies. Fossil fuel based energy sources primarily emit CO_2 and a FIT base subsidy system, in rewarding renewable energy generation to substitute fossil fuel based energy sources is effectively rewarding CO_2 abatement from an environmental perspective. Digestion of slurry is effective in abating CH_4 and N_2O, GHGs with GWP of 21 and 310, respectively. The current FIT subsidy is not able to reward the farmers for this GHG abatement and it is left uncompensated. A compensation system for N_2O and CH_4 abatement could help improve the farm economics from introduction of AD and will likely improve its adoption as a technology capable of supporting the UK agriculture related GHG emission abatement goals.

11.1 Future work

- This study considers a digester based solely on slurry. The existing structure of the model can be expanded to include the following scenarios
    - Consideration of other farms e.g., beef, pig or poultry farms
    - Addition of food waste or other types of organic matter
    - More detailed analysis of adding crop residues
    - Addition of other energy crops like maize
- Though the model can be scaled to larger farms, the economies of scale in digester and CHP cost may be under/over represented by a power function based capital costs calculation methodology used in this research. A more nuanced and detailed pricing study can provide additional insights for different farm sizes and farming practices.
- Enteric emissions are a significant part of GHG emissions on a dairy farm. When the cattle are housed, if the CH_4 in enteric emissions can be extracted and passed to a CHP unit to convert into heat and electricity, this can potentially make a significant difference to both environmental and economic benefits of introducing AD on a dairy farm. Further research into technologies and modelling of the same will be very useful.

A detailed study to quantify the emissions from digestate and differences vis-à-vis emissions from slurry, including a comparison across different farming practices, soil types and weather conditions should also help refine the conclusions reached in this research.

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Appendices

Appendix 1 - “Code” for Monte Carlo simulation

```
Sub runMonteCarlo()

  Sheets("Analysis").Select

  copyPaste "Analysis", "KM6", "Analysis", "B5"

  i = 3

  Sheets("MonteCarlo").Select

  farmAreaMin = Range("C5").Value

  farmAreaMax = Range("D5").Value

  liveStockDensityMin = Range("C6").Value

  liveStockDensityMax = Range("D6").Value

  percentageHousingMin = Range("C7").Value

  percentageHousingMax = Range("D7").Value

  loadingRateMin = Range("C8").Value

  loadingRateMax = Range("D8").Value

  maxMethaneYieldMin = Range("C9").Value

  maxMethaneYieldMax = Range("D9").Value


  farmAreaCount = 0

  liveStockDensityCount = 0

  percentageHousingCount = 0

  loadingRateCount = 0

  maxMethaneYieldCount = 0


  For farmAreaCount = 0 To 4
```

```
   farmAreaValue = farmAreaMin + (farmAreaMax - farmAreaMin) * farmAreaCount / 4

   For liveStockDensityCount = 0 To 4

   liveStockDensityValue = liveStockDensityMin + (liveStockDensityMax - liveStockDensityMin) * liveStockDensityCount / 4

   For percentageHousingCount = 0 To 4

   percentageHousingValue = percentageHousingMin + (percentageHousingMax - percentageHousingMin) * percentageHousingCount / 4

   For loadingRateCount = 0 To 4

   loadingRateValue = loadingRateMin + (loadingRateMax - loadingRateMin) * loadingRateCount / 4

   For maxMethaneYieldCount = 0 To 4

   maxMethaneYieldValue = maxMethaneYieldMin + (maxMethaneYieldMax - maxMethaneYieldMin) * maxMethaneYieldCount / 4


   Sheets("Analysis").Select

   Range("KM7").Value = farmAreaValue

   Range("KM8").Value = liveStockDensityValue

   Range("KM9").Value = percentageHousingValue

   Range("KM13").Value = loadingRateValue

   Range("KM14").Value = maxMethaneYieldValue


   Sheets("MonteCarlo").Select

   Range(ColumnLetter(i) + "21").Value = farmAreaValue

   Range(ColumnLetter(i) + "22").Value = liveStockDensityValue
```

```
   Range(ColumnLetter(i) + "23").Value = percentageHousingValue

   Range(ColumnLetter(i) + "24").Value = loadingRateValue

   Range(ColumnLetter(i) + "25").Value = maxMethaneYieldValue


   copyPaste "Analysis", "B38:B85", "MonteCarlo", ColumnLetter(i) + "28" + ":" +
ColumnLetter(i) + "75"

   i = i + 1


   Next

   Next

   Next

   Next

   Next


   copyPaste "Analysis", "E6", "Analysis", "B5"


End Sub
```

Appendix 1

Appendix 2 – Vienna Conference Paper

JAIN, S., SALTER, A. M. & BANKS, C. J. Calculating the economic cost of mitigating GHG emissions from UK dairy farms by anaerobic digestion of slurry. International Symposium on Anaerobic Digestion of Solid Waste and Energy Crops 28 August - 1 September 2011 Vienna, Austria.

Calculating the Economic Cost of Mitigating GHG Emissions from UK Dairy Farms by Anaerobic Digestion of Slurry

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Abstract

This study analyses anaerobic digestion (AD) as a renewable energy technology by quantifying the emissions avoided and the cost incurred in the process. The quantitative model developed and demonstrated uses basic farm information to evaluate dairy farms from an environmental and economic perspective. Based on the cost of installing and operating an anaerobic digester and the emissions avoided using this technology, the marginal carbon abatement cost (MAC) is calculated. The MAC thus obtained is used to analyse current policy incentives thereby bridging the gap between the environmental impacts, the economic (dis)incentives and sustainable farming practices.

Keywords
Anaerobic Digestion; Dairy farming; Emissions; Economics; Policy

INTRODUCTION

A change in farming practice in the UK could have a positive impact on reducing the country's greenhouse gas (GHG) emissions, both directly and also indirectly by offsetting fossil fuel usage. Directly, farms contribute 36% of the UK's methane (CH_4) emissions from livestock and livestock manures and 67% of nitrous oxide (N_2O) emissions from the use of either livestock manures or artificial fertilisers (Defra, 2009a). The UK's Low Carbon Transition Plan 2009 (HM Government, 2009) aims to cut by 2020 the GHG emissions from waste and farming by 6% based on 2008 levels. Indirectly, farming could also offset fossil fuel usage by both being a net producer of renewable energy and by reducing its dependence on inorganic fertilisers which have a high energy demand in their production. The Renewable Energy Directive (Directive 2009/29/EC) ('RED') will require the UK to source 15% of its energy needs from renewable sources by 2020 which will require a major step change to bring this about from the 2.2% production reported for generation from renewable and waste sources (DECC, 2009a).

On-farm anaerobic digestion (AD), in conjunction with good farming practices and support from the government, can make a contribution to meeting both of these targets. Another benefit is the role that AD can play in development of the rural economy by providing additional revenue to the farmers through the sale of energy, usually in the form of heat and electricity.

Following a major shift in carbon valuation policy, DECC (2009b) has moved away from the social cost and shadow price of carbon based on the Stern review, to the cost of mitigating emissions. For evaluating policies related to emissions not covered by EU Emissions Trading Scheme (the 'non-traded sector'), a short term non-traded price of carbon has been set at €72 tonne^{-1} CO_2 eq until 2020 with a range of +/- 50%, based on the marginal abatement cost (MAC) required to meet a specific emissions reduction target (DECC, 2009b). Policy that delivers mitigation cheaper than the non-traded price of carbon is considered to be cost effective.

This paper reports a method to calculate a MAC for AD by quantifying GHG emissions abated through the introduction of AD to a dairy farm and the change in revenue expected by doing so. This approach allows benchmarking policy that incentivises carbon emission reduction by rewarding mitigation and penalising emission. This paper is based on the analysis of four farming scenarios that could be employed in farming, using a modelling tool to estimate GHG emissions and an economic model for the farm and necessary investments for each scenario.

METHODS

Scenarios

The four scenarios used were based on a farm of 84.2 ha with 91 dairy cows and 101 followers (Jackson *et al.,* 2008).

Case 1: represents a partially grazed conventional dairy farm, most common practice in the UK. Dairy cows are housed for 60% of the year and grazed during the rest on permanent pasture. Winter wheat (9.6 ha) and grass silage (28 ha) are grown on farm to be used to feed the dairy cows. Followers are housed for 30% of the year and grazed during the rest.

Case 2: Farming system and land use distribution is the same as case 1 with the introduction of an anaerobic digester fed with slurry from the dairy cows and the followers. Electricity and heat produced is used in the dairy and surplus is exported to the grid. Digestate produced is used as an organic fertiliser applied using a trail hose spreader.

Case 3: Dairy cows are housed all year. Winter wheat (9.6 ha) used to feed the cows. Followers are grazed on a permanent pasture (28 ha) for 70% of the year. Rest of the land is cultivated for grass silage for the housed dairy cows and followers.

Case 4: Farming system and land use distribution is the same as case 3 with the introduction of an anaerobic digester fed with slurry from the dairy cows and followers. Biogas and digestate are handled in the same way as case 2.

Emissions Model

An emissions model was built to take into account the sources of GHG emissions identified on a dairy farm.

Enteric Emissions. It is assumed that CH_4 produced in the rumen of cattle as a by-product of fermentation is proportional to feed consumed and is all expelled enterically (IPCC, 2006). The enteric emissions were calculated based on the feed intake assuming the weight of a dairy cow is 650 kg (Defra, 2010), milk production 6,389 litres year^{-1} (Jackson *et al.,* 2008), fat content of milk 3.5% (Nix, 2007), digestibility of grass 70% (IPCC, 2006) and 6.5% of gross energy in feed converted to methane (IPCC, 2006).

Appendix 1

CH_4 emissions from manure management. It is assumed that each cow produces 1.7 tonne head^{-1} year^{-1} of excreta as volatile solids (Defra, 2010). When grazed this is distributed evenly on the pasture and when housed it is collected as a liquid slurry. The ultimate CH_4 yield is of excreta was taken as 0.24 m^3 CH_4 kg^{-1} volatile solids (IPCC, 2006). The average air temperature for the UK is 10°C (The Met Office, 2011). When slurry is used in association with AD on the farm it is fed directly to the digester from a sealed reception tank and the emissions are restricted to fugitive emissions from the digester itself. These will depend on the digester design, construction and management but were taken to be 3.5% of the gross methane production (Silsoe Research Institute, 2000).

There is limited quantitative data available in the literature on the emissions from field application of digestate and IPCC (2006) does not specify any emission factors, so the factors recommended for slurry have been used which may lead to some variability in results. The emission factor (EF) depends on soil moisture content, method of application of digestate, nitrogen application rate, soil type and type of vegetation (Sanger *et al*., 2010; Senbayram *et al*., 2009; Moller *et al*., 2009; Wulf *et al.,* 2002; Amon *et al*., 2006).

N_2O emissions from manure management. Liquid manure has a low redox potential and hence N_2O is not formed or released when in this state (Rodhe *et al.* 2009). There may, however, be N_2O emission when a dry crust forms on the surface. To account for this an EF for storage tanks with a natural crust cover was taken as 0.005 kg N_2O-N kg^{-1} N added (IPCC, 2006) and the rate of excretion of N by dairy cows as 0.27 kg N head^{-1} day^{-1} (Defra, 2010). It is assumed that there are no nitrogen losses from leaching while the manure or digestate is in a storage tank. Emissions originating from volatilisation of N from stored manure as ammonia or oxides of nitrogen have been calculated as per IPCC (2006).

N_2O emissions from managed soils. IPCC (2006) emissions factors were used taking into account the N additions to the soil. Manure to soils was estimated based on amount of manure excreted and its nitrogen content. Emissions from mineral fertiliser were based on N application rates either to meet the requirements of crops (Defra, 2010) or using guidelines set for Nitrogen Vulnerable Zones in the UK (Defra, 2009b). Indirect emissions from

volatilisation/atmospheric deposition and leaching/runoff were estimated based on IPCC (2006). No change in land use has been assumed.

GHG emissions from farm activities. All farm machinery is assumed to use diesel fuel and the energy required for the farming operations was calculated using the method and data in Salter and Banks (2009). A UK-specific emissions factor (EF) of 0.27 kg CO_2 eq kWh^{-1} was used to determine GHG emissions from the diesel consumed (DECC, 2009a). The GHG emissions from the production of mineral fertilisers were based on EF of 7.11 kg CO_2 eq kg^{-1} N, 1.85 kg CO_2 eq kg^{-1} P_2O_5 and 1.76 kg CO_2 eq kg^{-1} K_2O (Defra, 2009c).

GHG emissions from dairy energy import/export. The annual electricity consumption on a dairy farm was estimated as 306 kWh cow^{-1} (DLTech Inc, 2006). The GHG EF used for electricity consumption was 0.54284 kg CO_2 eq kWh^{-1} (DECC, 2009a).

Embodied carbon in AD. The size of the digesters, $95m^3$ and $143m^3$, was calculated using a slurry loading rate of 3 kg VS m^{-3} day^{-1}. Based on this size the embodied carbon in the digester was calculated as per Hammond and Jones (2008). In doing this it is assumed that the digester has a life of 20 years. The gas collected both from the digester and from the gas-tight digestate storage tank was used to produce electricity via a combined heat and power (CHP) unit.

Economic Model

The model assumes that livestock, land and all the dairy buildings and equipment are owned by the farmer. Annual costs for crop and milk production were calculated from Nix (2007). The current price of electricity bought is taken as 11.8 c kWh^{-1} and of gas as 3.5 c kWh^{-1} (DECC, 2009a). In order to account for the recent fluctuations in market price of wheat, a 5-year average (August 2005 - 2010) of €135.6 $tonne^{-1}$ was taken. Similarly a 5-year average of 26.5 c $litre^{-1}$ (August 2005 - 2010) was taken for the farm-gate price paid to the farmer for milk.

A useful rule of thumb for calculating capital cost investment for AD is €3,000 to €7,200 kWe^{-1} generated or €480 to €900 per m^3 of digester capacity (The

Anderson Centre, 2010). A high-end value of €900 per m[3] was used as economy of scale is expected to work against the small scale of the farms considered. The lifetime of a CHP unit varies from 8-12 years with a major rebuild after 2-3 years. The total price of the CHP unit, replacement and rebuilds, for a 20-year period is assumed to be €46,800. A mortgage rate on the investment required to set up an AD plant has been assumed at 9% over a period of 20 years (personal communication with banker), higher than the 7% recommended by the IBBK (2008) and the Anderson Centre (2010). Operating costs for the digester including labour, maintenance, repair, and insurance have been estimated at 7% of capital cost (IBBK, 2008; The Anderson Centre, 2010). Net profit is calculated based on enterprise cost, running expenses and value of produce. Current policy incentives like feed in tariffs and the renewable heat incentive have not been built into the model. The effects of these incentives are analysed using the model.

Loss in profit by introduction of AD is calculated by comparing the farms with AD with the corresponding base cases. The loss is then compared to the tonnes of CO_2 equivalent GHG emissions abated by its introduction. Thus a MAC is obtained in £ tonne^{-1} of CO_2 eq abated. Payback period is calculated assuming that a mortgage is not taken and all the upfront investment is made out of pocket. The subsequent additional profit earned by the sale of electricity and heat goes towards recovering that money.

RESULTS AND DISCUSSION

Emissions Model

The emissions for the four cases are presented in Table 1.

Table 1: Results from emissions modelling (kg CO_2 eq. ha^{-1} year^{-1})

	Case 1	Case 2	Case 3	Case 4
	Partial housing	Partial housing plus AD	Full housing	Full housing plus AD
Methane				
Enteric Emission	4,334	4,334	4,246	4,246
Dairy Cows	*2,903*	*2,903*	*2,815*	*2,815*

Followers	*1,431*	*1,431*	*1,431*	*1,431*
Manure Management	521	148	745	124
Grazing	*48*	*48*	*23*	*23*
Housing	*473*	*100*	*722*	*100*
Fugitive Emissions	0	177	0	264
Nitrous Oxide				
Manure Management	354	0	541	0
Direct	*197*	*0*	*300*	*0*
Indirect	*157*	*0*	*240*	*0*
Managed soils	1,958	1,958	1,750	1,750
Direct	*1,516*	*1,516*	*1,308*	*1,308*
Indirect	*442*	*442*	*442*	*442*
Carbon dioxide				
Farm activities	634	634	708	708
Electricity and Gas imported	195	-290	195	-541
Embodied carbon in AD	0	17	0	22
Total (kg CO_2 eq ha^{-1} year^{-1})	**7,997**	**6,988**	**8,184**	**6,574**

Enteric emissions account for nearly 50% of the GHG emissions which in the example used ranged from 2,815 to 2,903 kg CO_2 eq ha^{-1} year^{-1} for different housing conditions and are equivalent to 125 to 128 kg CH_4 cow^{-1} year^{-1}. This figure agrees with values reported in the literature which are in the range 96 to 120 kg CH_4 cow^{-1} year^{-1} (Lassey *et al.,* 1997; Bruinenburg *et al.,* 2002; Grainger *et al.,* 2009). More enteric CH_4 head^{-1} year^{-1} is emitted from grazed dairy cows as they are more active and consume more energy than housed cows, although this may be compensated for by selective grazing to increase the digestibility of fresh grass. Enteric emissions from dairy followers, modelled at 68 kg CH_4 follower^{-1} year^{-1}, fall within the 48 to 88 kg CH_4 per follower^{-1} year^{-1} range reported in literature (Pinares-Patino *et al.,* 2007). The presence of a digester does not affect the enteric emissions.

Emissions of CH_4 from manure are significantly higher when manure is stored from housed animals. In a grazed system manure excreted in the field is mainly broken down aerobically whereas slurry stored in a lagoon or tank is under predominantly anaerobic conditions which encourage the formation of CH_4. The fraction of methane yield converted for grazing cows reported in the literature ranges from 0.8 to 2.5% which is similar to the IPCC value of 1% (Holter, 1997). The methane conversion factor for a slurry based manure management system reported by Rodhe et al. (2009) is 2.7% which is much lower than the IPCC (2006) value of 10-17%. Hence, there may be an overestimation in the CH_4 emissions from slurry management calculated by the model which is based on IPCC methodology.

GHG emissions associated with storage of slurries are minimised in an AD plant if the feed slurry and the final digestate are held in gas-tight storage tanks connected to the biogas collection system. This is not always the case and if they are not then the overall emissions would be much higher than the estimates given. A poorly run or designed AD plant may also have a high level of fugitive emissions of biogas which, according to the model, would have to increase to 10% to be more damaging than open manure storage tank. It is therefore critical to monitor the performance of the AD plant on a regular basis.

N_2O emissions from manure management are in the order of 5% of the total emissions, but were shown to increase with housing as more slurry is stored in manure storage tanks. The model assumes there are no N_2O emissions from stored digestate.

N_2O ***emissions from managed soils were higher in cases 1 and 2 where partial grazing took place due to a*** higher direct loss of N from excreta deposited on the field than from the application of the slurry and digestate. ***The*** recommended fertiliser requirement for grazed grass is lower than that for grass silage due to better recirculation of nutrients in grazed grass, thus affecting the amount of fertilisers used and the emissions from their production and application. The emissions from crop production increase with the increase in housing as more grass silage is grown which requires more intervention than a grazed pasture. For the purposes of the model it is assumed that emissions from digestate spread to land were the same as from manure used in the same way.

In cases 2 and 4 the anaerobic digestion plant reduces GHG emissions by 1 and 1.6 tonnes CO_2 eq ha^{-1} $year^{-1}$. AD adds emissions from embodied carbon in the building materials used for its construction. These emissions account for 0.3% of the total emissions per hectare, as compared to other sources of emissions. In order to obtain optimum gas production, a digester requires heat to maintain temperature inside the digester and raise the feedstock to operating temperature and electricity to run the pumps and other equipment. The emissions corresponding to these are offset by the production of heat and electricity by the CHP unit. In case 2, a total of 78,988 kWh of electricity and 84,768 kWh of heat is generated by a 9 kW CHP unit. After accounting for dairy usage, 40,410 kWh of electricity and 16,359 kWh of heat are available for export resulting in an emissions reduction of 485 kg CO_2 eq ha^{-1} $year^{-1}$. Similarly, when the dairy cows are fully housed, a total of 122,262 kWh of electricity and 131,159 kWh of heat is generated by a 14 kW CHP unit. After accounting for dairy usage, 74,533 kWh of electricity and 32,431 kWh of heat are exported resulting in an emissions reduction of 736 kg CO_2 eq ha^{-1} $year^{-1}$. Thus the majority of the GHG savings resulting from the introduction of AD come from the energy produced and from avoided manure management emissions. By increasing the housing period of the dairy cows from 60% to 100%, the total GHG savings can be increased by 6%.

Economic Model

Results obtained from the economic model are given in Table 2.

Table 2: Results from economic model (€ ha^{-1} $year^{-1}$)

	Case 1	Case 2	Case 3	Case 4
	Partial housing	Partial housing plus AD	Full housing	Full housing plus AD
Costs				

(AD) Mortgage	0	173	0	229
Seeds	11	11	13	13
Fertiliser	47	47	54	54
Feed (wheat,	279	279	383	383
Concentrates	25	25	25	25
Bedding	23	23	39	39
Vet and medicine	51	51	51	51
Water	36	36	36	36
Electricity	39	0	39	0
Heat	4	0	4	0
Labour				
Crops	140	140	212	212
Dairy	459	459	459	459
AD	0	20	0	31
AD maintenance	0	36	0	53
AD insurance	0	15	0	23
Total	**1116**	**1317**	**1315**	**1608**
Value of Produce				
Electricity	0	57	0	104
Heat	0	7	0	14
Wheat	124	124	124	124
Straw	14	14	14	14
Silage	156	156	259	259
Milk	1831	1831	1831	1831
Total	**2125**	**2188**	**2228**	**2346**
Profit	**1009**	**872**	**913**	**738**

Labour costs account for 50% of the running costs on a dairy farm while the majority of the revenue comes from sale of milk. The feed produced (wheat and grass) is consumed on farm hence there is no profit or loss from its production and consumption. With increased housing, becoming more common as herd sizes and distance to grazing increase, the silage requirement and the farm activities associated with its cultivation increase resulting in a 10% drop in profit. There is an increased energy usage on farm related to maintenance of digester temperature and electrical needs of pumps and other related equipment. Increase in heat and electricity use on the farm is offset by their production for use on farm with the surplus exported. The sale of electricity and heat at 11.8 c kWh^{-1} and 3.5 c kWh^{-1} generates revenues of €107 and €161 ha^{-1} $year^{-1}$ in the two farms, by export of energy and by avoiding its import. The capital cost of AD has been estimated at €85,500 and €128,700 for digester capacities of 95 m^3 and 143 m^3 respectively. The extra revenue from the sale of heat and electricity is negated by mortgage payments of €173 and €229 per ha^{-1} $year^{-1}$ on the capital cost and additional running costs. The digestate is given no financial value as it is not sold off the farm although it has some value as a fertiliser replacement. The net profit after the introduction of AD drops by €137 ha^{-1} $year^{-1}$ in a 60% housed dairy farm while it drops by €175 ha^{-1} $year^{-1}$ in a fully housed farm. AD does not affect the medical, bedding, water requirements, milk yield and the corresponding costs and revenues in a dairy.

Introduction of AD on a typical dairy farm with cows housed for 60% of the year decreases the GHG emitted by 1 tonne ha^{-1} $year^{-1}$. Payback period if the capital investment is made out of pocket has been calculated as 29 years. The MAC for GHG is calculated to be €136 $tonne^{-1}$ CO_2 eq abated. Taking the current feed in tariff (FIT) of 13.8 c kWh^{-1} and renewable heat incentive (RHI) of 6.6 c kWh^{-1} into account, the MAC drops to €120 $tonne^{-1}$ CO_2 eq abated and the payback period to 20 years, making only a marginal difference to the farmer. Similarly, introduction of AD on a 100% housed dairy farm decreases the GHG emitted by 1.6 tonne ha^{-1} $year^{-1}$ at a cost of €175 ha^{-1} $year^{-1}$. Payback period has been calculated as 29 years and the MAC for GHG as €109 $tonne^{-1}$ CO_2 eq abated. Taking the current FIT and RHI into account, the MAC drops to €90 $tonne^{-1}$ CO_2 eq abated and the payback period to 18 years, again making only a marginal difference to the farmer. These values are on the higher side of the range of MAC range for other green technologies some of which are already subsidised (McKinsey and Company, 2007) and are also higher than the DECC recommended short term non-traded price of carbon. The profitability of AD is sensitive to the interest rate and in this case, a 7% interest would make the MAC comparable to the short term non-traded price of carbon. Based on the given scenarios, in order to make AD feasible, a FIT payment of 20-25 c kWh^{-1} would need to be introduced. This would reduce the payback period down to 10-15 years which is still quite high. The FIT and RHI may provide some support to the farmers interested in AD but do not go far enough to incentivise its adoption. Current policy structure drives maximum production of electricity rather than the reduction in carbon footprint which is where the real benefit of the technology lies. A restructured policy that rewards abatement and penalises excess emission based on MAC is required.

CONCLUSIONS

Appendix 1

According to the model, operating an on-farm digester reduces the GHG emissions from dairy farming at this scale by 1-1.6 tonne CO_2 eq ha^{-1} $year^{-1}$. MAC using an on-farm AD is €136-175 $tonne^{-1}$ CO_2 eq GHG mitigated. The FIT and RHI may provide some support to the farmers interested in AD but do not go far enough to incentivise its adoption. A green investment bank is being set up by the UK government to provide the extra support needed to green technologies through equity, loans and risk reduction. While these are steps in the right direction, we are a long way from realising the full potential of on-farm AD in the UK.

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