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Selection and specification of technical and management-based greenhouse gas mitigation measures in agricultural production for modelling

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1 Introduction

Approximately 10 % of European (EU15) GHG emissions can be attributed to the agricultural sector (Gugele et al., 2002). Within this sector, dairy production systems represent the largest source of CH_4 and N_2O emissions and additionally an important source of the indirect greenhouse gas ammonia which indicates their large potential for GHG mitigation. Related to dairy production measures in bull fattening may also have a high GHG mitigation potential. In addition, animal production has become highly specialised, industrialised and geographically concentrated in various parts of Europe and pig production volumes in particular strongly increased over the last decades. Main European pig producing areas can be found in the north (e.g. Denmark, the Netherlands, Belgium, Brittany in France, Lower Saxony in Germany) and the south (Lombardy in Italy, Catalonia and Galicia in Spain). A significant part of the N input in pig production is emitted as NH_3, under European conditions estimated at 30 % both from animal houses and from application of manure (IPCC, 2003). Therefore, pig fattening farms in particular contribute considerably to agricultural GHG emissions and hence also have a high GHG mitigation potential.

Alongside policy measures, potential mitigation measures for reducing GHG emissions in agricultural production systems can be technical (e.g. manure storage and application techniques), management-based (e.g. organic instead of conventional farming) or are a combination of the two (e.g. anaerobic digestion). Here, mitigation options can influence a specific section of the production process but in most cases the whole farming system.

In the past, GHG emissions and the potential of mitigation options have mainly been investigated for individual gases and for separate parts of the production chain, such as CH_4 emissions from ruminant digestion or manure storages, or N_2O emissions from grazed pasture. Not all identified measures appear feasible from a technical point of view, and the efficiency of the measures is not always clear nor is the impact calculated for each measure at the farm level.

Thus, as a first step, the aim of this part of the study was to select potential, feasible and cost-effective technical and management-based GHG mitigation measures for modelling at the whole farm level with a focus on dairy production and on bull and pig fattening farming systems. The selection of potential GHG mitigation measures was mainly based on deliverable report D7a and a workshop with MEACAP partners. Since the majority of measures with highest GHG mitigation potential are related to N use efficiency, (e.g. through a closer N cycle on the farm by more efficient use of manures, fertilisers and plant residues, and by increasing productivity of animals and crops) most of the selected measures are focussed on N use efficiency at the farm level. In addition, it was the objective of the selection process to identify mitigation measures that in total cover the whole production chain of both dairy production and bull and pig fattening farms.

In a second step, which represents the main part of deliverable report D10a, it was the aim to define and specify the selected mitigation measures and to describe in detail the used emission factors/equations and estimated abatement costs that are needed for the cost-benefit analysis (D15a) of the individual measures. The calculation of the GHG reduction potential and the cost-benefit analysis of the selected mitigation measures will be carried out with the model "ModelFarm" for three dairy farms and two bull and pig fattening farms based on a cluster analysis of existing farms in North-West Germany conducted by FAL (P2).

2 Selection and modelling of mitigation measures

For the analysis of the most promising technical and management-based measures for GHG mitigation in agricultural production in Europe, the selection of measures was based on the literature review carried out for deliverable report D7a in WP3, data from additional studies and on the expert knowledge of the MEACAP partners and external experts. As a first step, potential mitigation measures for both animal husbandry and crop production were collected and described (D7a). In order to take GHG emissions of the entire farming system (including pre-chain emissions) into consideration, mitigation measures of all different sources were analysed. In a second step, all identified measures were evaluated according to the following criteria:

- GHG mitigation potential,
- Technical feasibility,
- Environmental added value,
- Cost-effectiveness,
- Social acceptance,
- Animal health and welfare / ethical acceptance,
- State of technology knowledge,
- Availability of emission factors.

In addition, all technical measures were assigned to one of the following categories:

- N efficiency,
- Animal efficiency and livestock density,
- Manure,
- Carbon sequestration,
- Biomass,
- Agricultural energy use.

For the selection process, all measures were listed and judged according to these criteria in different combined evaluation tables to derive a ranking of the most promising measures (see evaluation tables in appendix 1 of this report). All measures were given a score with respect to the above mentioned evaluation criteria (5 = best - 1 = worst score, 0 = killing assumption). A "killing assumption" would mean that these measures have at least one negative impact (e.g. negative social acceptance or that no emissions factors are available for modelling) so that such measures were automatically ruled out from the evaluation process. For all measures a total score over all criteria was calculated, which allowed a ranking of all measures. Additionally, the measures were given a weighting factor with respect to the most important criteria 'GHG mitigation potential' (3), 'technical feasibility' (2) and 'cost-effectiveness' (2 to select particularly those measures with the highest potential concerning the respective question. Finally, during a workshop together with the MEACAP partners of WP6/7 and based on the described evaluation tables, 15 potential mitigation options were defined with additional consideration given to the feasibility of the measures with respect to administrative costs of implementation, control, monitoring and enforcement. Furthermore, it was agreed that organic farming as a management-based measure will be included in the modelling process.

Within this evaluation process the following options or the comparison of different mitigation techniques or management systems were selected for modelling:

- Feeding strategies,
- Comparison of straw-based and slurry-based housing systems,
- Frequency of manure removal,

- Improved outdoor manure storage techniques,
- Manure application techniques.
- Slow- and controlled-release fertilisers and fertilisers with urease or nitrification inhibitors
- Increase of grazing in comparison to animal housing,
- Anaerobic digestion,
- Organic farming.

The following measures are considered as further potential, but optional, mitigation measures for modelling:
- Solid-liquid separation of manure,
- Groundwater adjustment for grassland,
- Increase of N fixation,
- Continuous plant cover (catch crops and intercrops),
- Precision farming,
- Carbon sequestration.

The selection of the mitigation measures shows that one focus of modelling GHG abatement options will be on the reduction of emissions from nitrogen (N_2O, NH_3) achieved by an improvement of the nutrient cycle and N efficiency. The nitrogen cycle and nitrogen efficiency are influenced by various activities in agricultural production chains referring to livestock and manure management, as well as crop production and fertilisation, and therefore provide several starting points for GHG mitigation measures. Hence, a number of mitigation measures addressing the N-cycle will be modelled for their impact on N_2O, NH_3 and CH_4 emissions.

In general, the modelling in WP3 and WP6 will mainly deal with milk production, pig and bull fattening farms and the corresponding crop production systems. Hence, a cluster analysis of farms in North-West Germany was carried out by FAL (see the description of the cluster analysis in appendix 2 of this report) to select three typical dairy farms (DF1, DF2, DF3), two bull fattening farms (BF1, BF2) and two pig fattening farms (PF1, PF2). Based on the results of the cluster analysis, representative model farms will be defined with respect to stocking rate, crop rotations, milk and crop yields etc. to model the impact of the implementation of the selected measures on GHG emissions.

The analysis of the GHG mitigation potential and cost-efficiency of the different selected technical and management-based measures of the seven defined model farms will be carried out with the whole-farm model "ModelFarm" (Michel, 2006). This process-orientated farm production model was primarily developed to allow quantification of all environmental and economic effects of agricultural systems with and without biogas utilisation. In the model, all internal flows (between the compartments of the farms e.g. arable land and grassland, animal housing, manure storage, biogas plant) and external flows (import of resources such as seed, feedstuffs, fertilisers and energy; export of crops, milk and meat) are calculated. For the internal flows, different direct and indirect gaseous emissions (CO_2, CH_4, N_2O, NH_3) are estimated for all compartments by the use of emission factors and equations basing on various studies and reports, such as IPCC (IPCC, 1997, 2000), MIDAIR (the FarmGHG model, Olesen et al., 2004) or ALFAM (Søgaard et al., 2002). The energy used for production of machinery and buildings is additionally considered in the model and is mainly based on data by KTBL (2004, 2006). The environmental effects from the prechains are estimated by using the results of the Ecoinvent Data 1.1 (Ecoinvent Centre, 2004). Financial data and labour expenses base on KTBL (2002, 2004, 2006), Mittelfränkische Landwirtschaftsverwaltung

(2004), Bioland (2005) and ZMP (2005). The model ModelFarm is described in detail in Michel (2006).

The following chapters of D10a (WP3) contain the descriptions of the details with respect to the modelling of the nine selected technical and management-based GHG mitigation measures including additionally used emission factors/equations and costs to model the selected mitigation measures. This will provide the essential data of the cost-benefit analysis from D15 for WP6.

3 Specification of selected mitigation measures for modelling

3.1 Feeding strategies

In general, the nutrition and performance of animals have a significant influence on the amount of N excretion. Adjusting feed composition to decrease the amount of nitrogen excreted could be one of the most sustainable methods of reducing not only ammonia emissions but also other forms of agricultural nitrogen losses to water and air.

On average only about one third of feed N is transformed into the protein of animal products, while approximately two thirds of N intake is excreted in urine and faeces of different productive livestock (Table 1; Jongbloed, 1991; Kirchgessner et al., 1994). About ¼ of this N may be emitted as ammonia directly after excretion from the animals and during manure storage. The problem is that the extent to which ammonia emissions can be reduced through feeding strategies will be crucially dependent on current feeding practices (reference system). The reference varies greatly across Europe and in many cases is not documented.

Table 1: Typical nitrogen balances of pig farms (according to Jongbloed, 1991).

	Nitrogen [kg animal^{-1}]		
	Starter pigs (9-25 kg live weight)	**Growing pigs (25-106 kg live weight)**	**Sows, including nursing piglets**
Intake [kg pig^{-1} or kg sow^{-1} a^{-1}]	0.94	6.36	27.6
Excretion [kg pig^{-1} or kg sow^{-1} a^{-1}]	0.56	4.48	22.5
Retention [kg pig^{-1} or kg sow^{-1} a^{-1}]	0.38	1.88	5.07
Efficiency of retention [%]	40.5	29.5	18.4

Low nitrogen feed assumes changes in the composition of the feed at such a rate that the nitrogen content decreases. This can be achieved by 1) the reduction in the level of nitrogen applied to grassland or substitution of grass by silage (cattle), 2) a better tuning of compound feed to the nutrient needs of the animals (in particular for pigs and poultry), 3) changes in the composition of the raw materials (in particular for pigs and poultry), 4) pelleting of feeds to improve feed efficiency through increased energy and protein digestibility (in particular for pigs), 5) supplementing diets with e.g. synthetic amino acids (in particular for pigs and poultry) and 6) replacement of grass and grass silage with maize (cattle) (Klaassen, 1991; Wijnands & Amadei, 1991).

NH_3 and N_2O emissions are largely dependent on the amount of nitrogen excreted by animals. Since a lower nitrogen content of the fodder reduces the nitrogen excretion per animal, consequently NH_3 and N_2O emissions from livestock will decrease accordingly (assuming a constant livestock population) (Velthof et al., 1998).

Dairy cows

Dairy cows excrete N via milk, manure and urine. Milk N represents about 30 %, manure N 30-40 % and urinary N about 20-40 % of total N intake. The amount of N excretion depends closely on the feed intake and therefore also on the targeted milk yield of the cows (Gruber & Steinwidder, 1996; Clemens & Ahlgrim, 2001). Generally, there exists a linear increase between N excretion and milk yield due to the requirements of higher intake of nutrients (Kirchgessner et al., 1993). This results in an asymptotic decrease of the specific N excretion per kg milk because the part for maintenance remains constant (Kirchgessner et al., 1993; Gruber & Steinwidder, 1996; Clemens & Ahlgrim, 2001). In addition, the N excretion is influenced by the crude protein content of the diet (Kirchgessner et al., 1993). Increasing crude protein contents causes higher N excretion by urine, while the N content of the faeces

increases with the intake of dry matter above protein contents of 13 % (Kirchgessner et al., 1993). According to Smits et al. (1995), Paul et al. (1998), Kröber et al. (2000) and Külling et al. (2001, 2002) for each percentage point increase in the crude protein content of a dairy cow ration, nitrogen emissions increased within the range of 10-20 %. James et al. (1999) reported similar results for heifers. Castillo et al. (2000) suggested that reducing the crude protein content of a cattle diet from 200 to 150 g kg^{-1} dry matter would reduce the annual nitrogen excretion in faeces by 21 % and, more importantly, in urine by 66 %.
Values for different milk yields and protein contents are tabulated in Table 2 according to Kirchgessner et al. (1993) and Clemens & Ahlgrim (2001).

Table 2: N excretion in kg N a^{-1} (Kirchgessner et al., 1993) and specific N excretion in g N kg^{-1} milk (Clemens & Ahlgrim, 2001) of dairy cows (lactation period: 310 days, dry period: 55 days) affected by milk yield and crude protein content of the ration.

Crude protein content of the ration [%]	**Milk yield [kg a^{-1}]**		
	4000	**5000**	**6000**
	N excretion [kg N a^{-1}]		
13	65	66	—
15	80	85	89
17	95	103	111
	Specific N excretion [g N kg^{-1} milk]		
13	16.3	13.2	—
15	20	17	14.8
17	23.8	20.6	18.5

These results confirm that increasing milk yields also require an increasing crude protein content of the diet and that high-producing dairy cows require a proper balance of rumen non-degradable protein and rumen degradable protein to meet their requirements for metabolisable protein. Metabolisable protein is the protein that the cow actually absorbs and uses for production. The requirement for rumen degradable protein for lactating dairy cows is 35-38 % of total crude protein (LPES, 2006). When cows were precisely fed to meet rumen non-degradable protein and rumen degradable protein requirements, they excreted 101 kg of N per year (LPES, 2006). When cows were fed simply to meet their total crude protein requirement, however, they excreted 118 kg of N per year (Table 3).

Table 3: Daily and yearly excretion of N by 635 kg Holstein dairy cow (LPES, 2006).

	0-30 days in milk	**31-100 days in milk**	**101-305 days in milk**	**60-day dry period**	**Yearly total**
Milk [kg cow^{-1}]	45.4	31.8	22.7	Dry	9866
Dry matter intake [kg cow^{-1}]	25.3	21.0	17.8	11.4	6560
kg N excreted per day					
Total N (low protein degradability)	0.40	0.33	0.27	0.16	101
Total N (high protein degradability)	0.47	0.39	0.32	0.20	118

In grazing systems the intake of protein may be relatively high, resulting in a surplus N in the diet which is mainly excreted in the urine (Figure 1). Kebreab et al. (2001) described the following relationships between N intake (NI) and N outputs in milk, faeces and urine (g N cow^{-1} d^{-1}) on the basis of several feeding experiments:

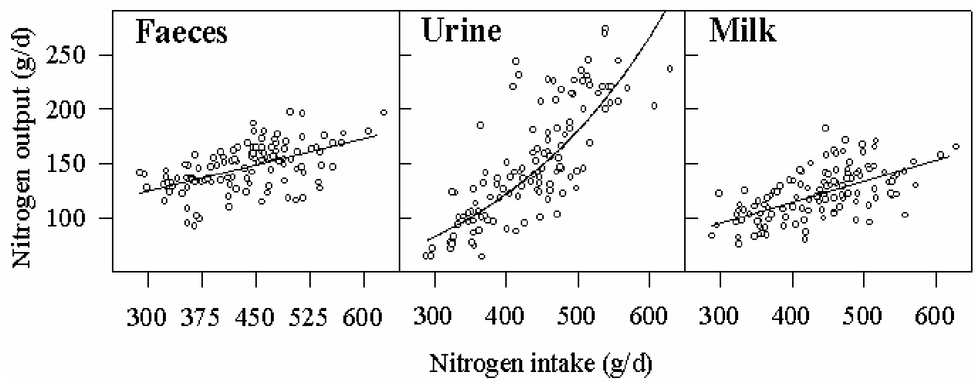
$$N_{milk} = 34.8 + 0.17(NI) \quad (1)$$

$$N_{faeces} = 78.0 + 0.15(NI) \quad (2)$$

$$N_{urine} = 0.0052(NI^{1.7}) \quad (3)$$

These equations show that reductions in N intake will have a relatively large impact on urinary N. For example, decreasing the daily N intake per cow on the given ration from 500 to 400 g N would decrease N in milk protein by 14 %, but urinary N by 32 % (Figure 1).
Petersen et al. (1998) found urea N as a percentage of total urinary N to vary between 65 and 95 % in most cases. Hence, reducing the N surplus in the diet will significantly influence the excretion of urinary urea.

Figure 1: Relationship between total N intake and the proportion of subsequent N outputs in faeces, urine and milk from Holstein dairy cows fed 30 different diet types (Kebreab et al., 2001).

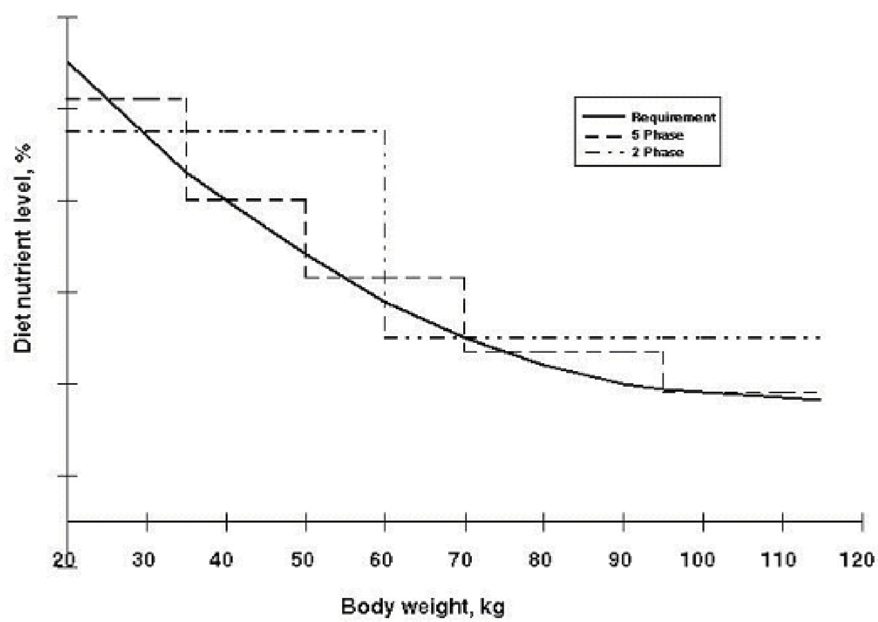


For cattle fed mainly on roughage (grass, grass silage etc.), a certain protein surplus is often inevitable (particularly during summer) due to an imbalance between energy and protein in young grass. This surplus might be reduced by adding components with lower protein content to the ration (e.g. maize). The latter option will be partly limited in grassland regions where roughage is the only feed available. Here, as an alternative or additionally the proportion of concentrate in the ration can be increased. According to UBA (1994) such an N adapted feeding of dairy cows may in total reduce N excretion of dairy cows by 10-20 %.

Fattening Pigs
For fattening pigs there are also different promising feeding strategies to reduce N losses. In general, a better feed efficiency is the most obvious strategy for reducing N excretion. A proper processing of feeds represents a very practical means of decreasing nutrient excretion through improvements in feed digestibility. Pelleting of feeds has been estimated to improve feed efficiency through increased energy and protein digestibility and subsequently to reduce N excretion. But the key to minimising nutrient output is to match the supply of available nutrients to animals' requirements. Over- or underfeeding nutrients relative to the animals' requirements will increase N output since animals will simply excrete all of the nutrients they are unable to use for maintenance and growth. Accurate estimates of nutrient requirements are therefore essential to optimise the production system but in the majority of cases they are moving targets, depending on factors such as energy density of the diet, stage of development, genetic potential, sex, health status and environmental conditions.
One of the reasons for high N losses from pig production arises from the fact that the protein demand of the animals changes considerably during the course of the production cycle (pregnancy/lactation, start/end of fattening), while the protein content of the feed is often kept constant at the level of maximum requirement. Indeed, such a feeding technique appears to be beneficial especially with respect to working management, but it automatically produces a considerable protein surplus along the whole production cycle. The excessive amounts of protein ingested by the animal have to be eliminated by degrading the protein N mainly to urea and by excretion via the urine.

In pig fattening, the dietary protein content necessary to meet the animals' requirements decreases steadily during the course of the production chain. In addition, during the growth of the fattening pigs the gain of fat is greater than the gain of protein. Therefore, the energy requirements are also greater than those of protein. In the case of feeding, with the same compositions in the diet during the whole fattening period, there exists a surplus of protein at the end of the fattening period, which cannot be utilised and from which the N is excreted via the urine and faeces. Therefore, the food composition with regard to the protein content and the decline in protein requirement should be adapted more accurately to the actual demand several times during the fattening period by using several types of feed with different protein content by (multi)phase feeding (Figure 2; Gruber & Steinwidder, 1996; Kaiser et al., 1998). Also the use of separate diets for pregnancy and lactation compared to one uniform diet has been reported to reduce nitrogen excretion without influencing reproductive traits between sows on one- and two-phase feeding.

Figure 2: Effect of number of feed phases on nutrient excess relative to nutrient requirement (Murphy & de Lange, 2004).



Thus, phase feeding, in which diets can be automatically adjusted by means of a computer controlled feeding system (Henry & Dourmad, 1993), represents an ideal example of how a herd manager can reduce N excretion and subsequently GHG emissions and may increase profitability at the same time. Phase feeding is generally applicable for all livestock and could mostly be implemented in the short term. In Germany, for instance, only 50 % of fattening pigs are fed by an N-adapted feed plan (Osterburg, 2002).

Phase feeding with the addition of (synthetic) amino acids can additionally reduce N losses and therefore predominantly ammonia emissions (Spiekers & Pfeffer, 1990; Heber et al., 1996). The balance of amino acids in the protein is the most important factor affecting the utilisation of dietary protein. Pigs do not have a requirement for protein per se, but for the amino acids that make up the proteins. The closer the amino acid composition of the diet matches the requirement for maintenance and production the less protein the animals effectively need and the less nitrogen is excreted in urine and faeces. Plant proteins rarely supply amino acids in the required ratio. Therefore, feedstuffs are combined to meet the animals' needs for the most limiting amino acids. This practice usually results in a higher than required protein content of the diet due to the presence of other amino acids in excess. These amino acid requirements also vary with age and physiological status. Therefore, to keep unnecessary nutrient losses to a minimum, nutrient supply has to change almost continuously, which can be achieved by phase feeding. Especially for grower-finisher pigs the greatest improvements in the efficiency of N utilisation can be achieved from improving the dietary amino acid balance so that the diet more closely reflects the true balance in which amino acids are required. The use of commercially available synthetic amino acids therefore provides a means for increasing the efficiency of utilisation of dietary protein, allowing lower protein inclusions and a better utilisation of the protein in the diet.

In a study by Baidoo (2001), the dietary crude protein was reduced from in excess of 20 % to 13 % for growing-finishing pigs with no effect on performance. In a similar study, the low protein diets reduced nitrogen excretion by 40 % in manure when compared to high protein diets. Feeding according to amino acid requirements, rather than total protein requirements can additionally reduce the nitrogen content of manure. About one percentage unit reduction in total protein content of growing-finishing pig diets is possible with the use of synthetic amino acids. According to Baidoo (2001) a reduction in dietary protein can result in a decrease of 16-35 % for the growing period and 19-20 % for the finisher period in urinary nitrogen excretion. Multiple phases improved feed efficiency by approximately seven percent. Calculations show that by changing from one feed system to a 2-phase system, the N needs would be met more precisely resulting in a reduction in N in manure of 12 %. Going from a 1-phase to a 3-phase feeding programme should reduce N excretion by about 17.5 %. The use of multi-phase feeding systems for growing-finishing pigs with a reduction in protein content of the diets but supplemented with synthetic amino acids, resulted in a reduction in nitrogen by at least 12 % (Baidoo et al., 1995; Kotchan & Baidoo, 1997).

Regarding sows, there exist considerable differences in the requirements of energy and protein between pregnant and lactating sows. Therefore, at least a two-phase feeding should be used, which can reduce N excretion by 12 % (Heinrichs, 1994).

In total, N emissions from fattening pig management may be reduced by phase feeding in the order of approximately 10-20 % (Roth & Kirchgessner, 1993; Windisch, 2001) or up to 40 % if amino acids are added (Spiekers & Pfeffer, 1990; Kirchgessner et al., 1994; Heber et al., 1996) compared to universal diets.

With the current cost of synthetic amino acids, it does not make sense to include synthetic amino acids other than lysine in grower pig diets but this will change as the availability and price of other amino acids improves (Murphy & de Lange, 2004).

3.1.1 Definition of mitigation measure

The modelling of feeding strategies by adjusting the feed composition to lower nitrogen feed will be carried out for two dairy model farms and two pig fattening model farms.

The feed plans of dairy farm 2 (DF2) and 3 (DF3) will be adapted in such a way that the crude protein contents match to the animal's performance and the feed efficiency is improved. In addition, the feed will be adapted to meet also the animals' requirements of energy, crude protein and metabolisable protein. Since the cows of the model farms DF2 and DF3 are mainly fed on grass and grass silage, there is a surplus of protein which will be adjusted by adding more maize silage with lower protein content and by increasing the proportion of concentrates in the ration. The detailed changes in the feed plan with respect to changes in the crop rotations due to the adapted feed plans and to the adjustments in terms of feed imports will be determined and described in detail when the definition of the model farms is finalised.

According to StMLF (2003) such a change in the feed plan will reduce N excretion by 10 %. This factor will be used in ModelFarm although other studies (UBA, 1994; LPES, 2006) have reported reductions of N excretion of up to 14-20 %. In addition, the N excretion by faeces and urine will be calculated for the different feed plans according to Equation (4) basing on Equations (2) and (3) by Kebreab et al. (2001) and applicable to dairy cows. The influence of the reduced N excretion on GHG emissions from cows and the whole farm will be calculated by the model.

$$N_{excreted} = 78 + 0.15\,(NI) + 0.0052\,(NI^{1.7}) \quad (4)$$

An adaptation of the feeding system for fattening pigs will be calculated for both defined model farms (PF1 and PF2). The reference model farms PF1 and PF2 per definition have a common two-phase feeding system with one feed for starter pigs (10-25 kg) and one feed for growing pigs (25-110 kg). For the comparison of a better feed efficiency the model farms were additionally defined to have only a universal diet for all fattening pigs (1-phase) or a 3-phase feeding system. Furthermore, the 3-phase feeding system will distinguish between the addition of amino acids and without.

According to several recent studies (Table 4) the N excretion of the 2-phase and 3-phase feeding system will be modelled to be on average 10 % and 15 % lower than the one feed system. The addition of amino acids will be calculated to reduce the N excretion by 40 % (Table 4).

Table 4: Reduction in N excretion through changes in pig feeding practices (basing on ALFAM, 2002).

	Reduction [%]		Reference
	N excreted	NH_3 loss	
Low N-diet	—	20-30	Dourmad et al., 1999 Menzi et al., 1997
Phase feeding (2 phases)	8.5 9 10 12	—	Heber et al., 1996 Spiekers & Pfeffer, 1990 Henry & Dourmad, 1993 Kirchgessner et al., 1993 Heinrichs, 1994 Baidoo, 2001
Phase feeding (3 phases)	13.3 15 17.5 18	—	Heber et al., 1996 Spiekers & Pfeffer, 1990 UBA, 2005 Baidoo, 2001 Ratschow, 1994
Phase feeding (3 phases) + amino acids	40 41.5	—	Spiekers & Pfeffer, 1990 Heber et al., 1996
Adding lysine	40	—	Kirchgessner et al., 1994
Change from 18 % crude protein to 10 % + essential amino acids	>40	—	Sutton et al., 1997
Reduced N in feed	—	46 (growing) and 46 (finishing	Kay & Lee, 1997
N crude protein reduction	—	10-12.5 % reduction per % decrease in dietary crude protein (interval 15.5-12.5 % crude protein)	Canh et al., 1998

3.1.2 Abatement costs

Cost differences of the N adapted feeding strategy for dairy cows are basically caused by the adjusted plant production dependent on the changed feed plan and also by the extensification of the grassland production (e.g. reduced fertiliser application) due to the reduced need for roughage fodder (grass and grass silage). In contrast, possible higher expenses for more maize silage production and increasing imports of concentrates have to be considered too. Cost changes for the different production of crops and the import of concentrates will be calculated by ModelFarm basing on the data stored in the database of the model (KTBL, 2002, 2004; ZMP, 2005). The animal housing and personnel costs with respect to the feeding management were estimated to be the same for both feeding strategies.

Since the N-adapted feeding of fattening pigs will cause both cost savings and partly increased costs, the changes in feeding costs will be neglected. According to Eurich-Menden et al. (2002) the cost adaptations of a phase feeding system including the additional work needed for the preparation of the different compositions of the diet will be considered to be 2.8 € pig place^{-1} a^{-1} higher for a 3-phase feeding system compared to the reference universal diet and was estimated to be 2.4 € pig place^{-1} a^{-1} higher for the 2-phase feeding system. The additional use of amino acids was calculated to cost 1.6 € pig place^{-1} a^{-1}.

3.2 Comparison of straw- and slurry-based housing systems

Animal housing systems vary enormously across Europe and levels of ammonia, methane and nitrous oxide emissions depend on the animal and housing system. In general, animal husbandry can be divided into slurry- and straw-based (deep litter) systems or is in some cases a combination of both.

Slurry-based systems are more common in most European countries than straw-based systems. In Germany, only 16 % of dairy cows were kept in straw-based systems and 84 % in slurry-based systems (33 % in tied systems, 51 % in loose housing systems) in 1999 (Osterburg et al., 2002). But for financial and animal welfare reasons or because of the likely increase of organic farming, straw-based systems may become more popular in the future. Typical straw-based systems are applicable to cattle, pig and poultry farming.

An additional straw-based system is the deep litter system that is also applicable to cattle, pigs and poultry. In deep litter housing systems, animals are kept on a thick layer of a mixture of manure with sawdust, straw or wood shavings as dry absorbent material (litter). Deep litter based production systems have been developed as an alternative to intensive housing types and are often used because of their ease and speed of construction, high flexibility as well as their relatively low capital cost. Large amounts of manure are allowed to accumulate in the litter since the litter is generally removed only 1-2 times a year.

Commonly-used bedding materials include various species of grain and grass straws, peat (Table 5), sawdust, shredded paper, reusable plastic, hardwood bark, and wood shavings (White & McLeod, 1989; Brake et al., 1992; Thompson, 1995). The Dairy Housing and Equipment Handbook (1995) lists water absorption of straw at 1.05, pine sawdust at 1.25, and pine shavings at 1.0 kg of water per kg of bedding. The dairy manual by Adams (1995) reports water absorption in kg of water per kg of bedding of 1.2 for chopped oat straw, 1.5 for chopped mature hay, 1.25 for pine sawdust, and 0.65-0.75 for wood shavings. Long straw is less absorbent than short or chopped straw (by a factor of 10 or more). Wheat and barley straw systems combined absorb 150 % more water than barley.

Table 5: Bedding utilisation rates according to Šileika (2000).

Animal	Animal housing type	Bedding input [kg day^{-1}] *	
		straw	**peat**
Mature cattle	Tied	2.5	4.0
	Loose with bedding	5.0-8.0	3.0-5.0
	Loose in cubicles	0.3	1.0
	Loose in combi-cubicles	1.5	2.0
Calf under 6 months	In individual pens	1.5	—
	In group pens	1.5	3.0
	Loose in cubicles	0.2	0.6
Cattle yearling 6-18 months	Tied	2.0	2.0
	Loose with bedding	3.0-4.0	6.0-8.0
	Loose in cubicles	0.3	0.8
Beef cow with calf	Loose with bedding	5.0-6.0	8.0-10.0
Fatling pig	In shallow pigsty	0.15	0.25
	On deep litter	3.0	4.5
Sow with piglets	In shallow pigsty	1.4	—
Sheep	With bedding	0.3	—
Hens and replacement pullets from 19 weeks	With bedding	0.05	—
Geese and replacements	With bedding	0.10	—
Turkeys	With bedding	0.05	—

* Humidity of straw used for litter (15 %), humidity of peat (45 %). Rate of litter has to be increased if its humidity is higher.

The addition of straw can influence the microbial activity in farmyard manure (FYM) or bedding material in animal housing, e.g. by the improvement of the C : N ratio in order to reduce NH_3 emissions (Enquete-Kommission, 1994; Jeppsson et al., 1997). The use of bedding material such as straw additionally has been shown to reduce the release of ammonia due to a lowered pH value of the manure (Andersson, 1995; Jeppsson et al., 1997).
A literature review by Bussink & Oenema (1998) shows that the absorption of urine by straw may effectively reduce NH_3 losses. Ammonia emissions are influenced by the bedding material (straw, peat etc.), the amount of bedding material and how often the material is applied (Van den Weghe, 2001). Some studies also show that a high straw content can give rise to lower ammonia emissions than some traditional slurry-based housing systems. Reductions of NH_3 emissions in cattle housings by 0-50 % were reported in different studies (Andersson, 1996; Kaiser & Van den Weghe, 1999). For pig fattening, the NH_3 emissions of straw-based systems are on average reduced by 20-50 % compared to conventional slurry systems (Amon et al., 1998).
But if bedding material is added it must be considered that the amount of material used for bedding may have an impact not only on emissions from the buildings but also on subsequent emissions during storage and spreading (Table 6; Pain & Jarvis, 1999).

Table 6: Effects of the amount of straw bedding (at 1, 0.75 and 1.25 times the standard rate of use) on annual ammonia emissions from beef cattle systems (Pain & Jarvis, 1999).

	Increasing straw usage →		
	0.75	**1.00**	**1.25**
	[kg NH_3-N per 500 kg liveweight]		
Housing	2.7	1.6	1.2
Storage	0.9	0.9	0.4
Spreading	2.7	1.2	0.7
Total	6.3	3.7	2.3

With respect to GHG emissions there is also an effect if animals are kept in loose housing systems or in a tying system with stall partitions between each animal to guide movements in stalls and in that way preventing unnecessary contamination of the stall surfaces with faeces and urine. In general, it is agreed that this type of tying system improves stall hygiene and reduces NH_3 emissions in stalls. According to different studies a NH_3 reduction of approximately 65 % seems to be possible (Hartung & Monteny, 2000).
According to Döhler et al. (2002b), deep litter systems for pigs should not be promoted as they are likely to result in an increase in NH_3 emissions, and as they do not offer separate dunging and lying areas, which is required by pigs. Ammonia emissions of deep litter systems for fattening bulls and heifers can be 20 % higher compared to full slatted floors (Döhler et al., 2002b).
Groenestein & van Faassen (1996) also concluded that deep litter systems for fattening pigs may reduce NH_3 emissions compared with housing on fully slatted floors, but emissions of air-polluting nitrogen gases tend to be higher due to the formation of N_2O (Table 7). This is caused by the fact that the addition of organic material such as straw may increase N_2O emissions by nitrification due to the increased surface to air (Amon et al., 1998). As a result of the addition of absorbent material an increase in N_2O emissions from denitrification is also possible, especially in the case of using only small amounts of straw and litter, so that very wet and dense areas (anaerobic zones) may form in the litter of manure (Ahlgrimm et al., 1998; Döhler et al., 1999; Amon et al., 1998). Furthermore, deep litter systems tend to become very warm during summer and as a result release considerable N_2O emissions.

For pig housing, the variation in N_2O emissions is mainly caused by the type of housing system (Table 7). Fattening pigs kept on partly or fully slatted floors (slurry systems) emit very little N_2O (0.02-0.31 kg per livestock unit per year), whereas higher emissions (1.09-3.73 kg per livestock unit per year) were reported for fatteners in deep litter and compost systems (Groenestein & van Faassen, 1996).
Also Sneath et al. (1997) identified a significant increase of N_2O emissions from straw-based systems in their N_2O inventory for the UK. Slurry systems, however, produce no or only little N_2O because slurry generally contains neither nitrate nor nitrite (Hüther, 1999). Sneath et al. (1997) also reported very low N_2O emissions at the detection threshold of the measuring instrument (Table 7). Therefore, Sneath et al. (1997) suggest as a mitigation option, changing from farmyard manure to slurry systems.

Table 7: N_2O emission (g LU^{-1} d^{-1}) from cattle and pig housing systems (according to Hartung & Monteny, 2000; Hartung, 2002).

Housing system	Emission	Reference
Cattle in tying stall	0.14-1.19 (0.62)	Amon et al., 1998
Cattle in deep litter (straw)	2.01 2.9	Amon et al., 1998 Hahne et al., 1999
Cattle in loose housing system	1.6 0.8	Jungbluth et al., 1999 Sneath et al., 1997
Fattening pigs on fully slatted floors	0.15 0.02-0.04 0.15	Hahne et al., 1999 Kaiser, 1999 Stein, 1999
Fattening pigs on partly slatted floors	0.02	Sneath et al., 1997 Groot Koerkamp & Uenk, 1997
Fattening pigs on fully or partly slatted floors without straw	0.15 0.31	Hoy et al., 1997 Thelosen et al., 1993
Fattening pigs on deep litter / compost	1.9-2.4 2.48-3.73 0.59-3.44 1.55-3.07 1.43-1.89 1.09	Döhler, 1993 Groenestein & van Faassen, 1996 Hoy, 1997 Kaiser, 1999 Stein, 1999 Thelosen et al., 1993
Fattening pigs on straw	0.05	Kaiser, 1999
Fattening pigs on a straw flow system	1.6-2.4	Hesse, 1994

In addition, deep litter systems may also increase CH_4 emissions. The data in Table 8 (Hartung & Monteny, 2000; Hartung, 2002) illustrate that CH_4 emissions from cattle houses (CH_4 emissions originate from both the animals and the excrements stored indoors) range from between 120 and 390 g d^{-1} LU^{-1}, with somewhat higher values for dairy cows in loose-housing systems (cubicle houses). This range of data is comparable with the range of CH_4 emissions used as normative values for dairy cattle in the Netherlands (63-102 kg per year per animal, corresponding to 173-279 g d^{-1} per animal) (Van Amstel et al., 1993). The highest CH_4 emissions occur during feeding and rumination (Brose et al., 1999). The emission levels are mainly influenced by the animal weight, the diet, and the milk yield. Furthermore, details of the housing system design (e.g. air conduction, type of flooring, type and dimensions of manure removal and storage of excrements) play an important role. The large number of influencing factors shows that realistic normative values for the calculation of CH_4 emissions (e.g. in national studies or emission inventories) should be differentiated with regard to

housing systems, the age of the animals, the type of feed, diet and feeding level and the lactation stage.
Similar to deep litter stalls for cattle, significant CH_4 emissions from pig husbandry exclusively originate from deep litter or compost systems (Table 8). Excrements temporarily stored indoors are the main source of methane emissions. According to Ahlgrimm & Bredford (1998) the quantity of methane emitted by the animal itself should not be neglected because it may amount up to 8 litres of CH_4 per pig and day. The amount of methane emitted from stalls for fattening pigs is influenced by the diet (digestibility), the daily weight increase of the animals, the air temperature, and the type of housing system (Ahlgrimm & Bredford, 1998; Hüther, 1999). The data in Table 8 show clear variation. With regard to CH_4, this is mainly caused by the different animal species and housing systems. Methane emissions from fattening pigs range from 1.5 to 11.1 kg per animal place per year, whereas emissions of 21.1 and 3.9 kg per animal place per year were reported for sows and weaners, respectively. Hahne et al. (1999) found higher CH_4 emissions in autumn and winter, when the air exchange rates are lower. They suggested that the CH_4 production might be influenced by the availability of oxygen over the emitting surfaces.
The CH_4 emission factors from Freibauer & Kaltschmitt (2001) in Table 9 which are based on a relatively small numbers of recent studies are lower as emissions in contrast to Table 8 originate only from manure. The results clearly show that slurry-based systems emit more methane than straw-based systems.

Table 8: CH_4 emission (g LU^{-1} d^{-1}) from cattle and pig housing systems (system level) (according to Hartung & Monteny, 2000; Hartung, 2002).

Housing system	Emission	Reference
Dairy cows in tying stalls	327	Kinsman et al., 1995
	120	Groot Koerkamp & Uenk, 1997
	194	Amon et al., 1998
Dairy cows in loose housing	320	Sneath et al., 1997
	265	Groot Koerkamp & Uenk, 1997
	200-250	Jungbluth et al., 1999
	267-390	Seipelt et al., 1999
Fattening bulls on slats	147	Groot Koerkamp & Uenk, 1997
Beef cattle on slats	121	Groot Koerkamp & Uenk, 1997
Fattening pigs on fully slatted floors	55-88	Hahne et al., 1999
	20-114	Gallmann et al., 2000
Fattening pigs on partly slatted floors	82	Sneath et al., 1997
	217	Groot Koerkamp & Uenk, 1997
Fattening pigs on fully or partly slatted floors without straw	29-59	Ahlgrimm & Bredford, 1998
	41-63	Brehme, 1997

Table 9: Emission factors for CH_4 from manure in animal house (Freibauer & Kaltschmitt, 2001).

Class of livestock	Housing system	CH_4 loss kg $head^{-1}$ a^{-1} CH_4
Dairy cows	Slurry-based systems	16
	Farmyard manure systems	13
Other cattle	Slurry-based systems	7.3
	Farmyard manure systems	4.4
Pigs	Slurry-based systems	3.6
	Farmyard manure systems	1.1

The reduced CH_4 emissions of straw-based compared to slurry-based systems are also in line with results from Döhler et al. (2002b) who summarised NH_3, N_2O and CH_4 emissions for cattle, bull and pig housing systems (Table 10). The N_2O emission factors also show minor differences between the systems whereas ammonia volatilisation is similar for the slurry- and straw-based animal housing types.

Table 10: Emission factors for NH_3, CH_4 and N_2O of cattle, bull fattening and pig housing systems (according to Döhler et al., 2002b).

Animal housing		Average emissions in kg per animal and year		
		NH_3-N	N_2O-N	CH_4
Cattle tied system	Slurry	4	0.3	90
	FYM	4		45
Cattle loose housing system	Slurry	12	0.5	90
	FYM	12	—	70
Bull fattening tied system	Slurry	2.0	—	—
	FYM	2.0	—	—
Bull fattening loose housing system	Slurry	2.5	—	—
	FYM	3.0	—	—
Pig slurry-based system	Slatted floor	3	0.1	4
	Partly slatted floor	3	0.05	4
Pig straw-based system	Deep litter (incl. compost)	4	2.5	3.5
	Two area floor, Danish system	4	0.1	2.5

3.2.1 Definition of mitigation measure

The comparison of slurry- and straw-based systems will be carried out for one dairy (DF1) and one pig fattening model farm (PF1). The dairy model farms will be calculated having either a tied stall (separate/slurry) or a loose housing system with a slatted floor (slurry) or a litter floor (deep litter). For pig fattening only a deep litter system will be compared with a slurry-based model farm with a slatted floor.

The use of bedding material (straw) depends on the house type, manure management and the time animals spend in the house and is estimated according to Poulsen et al. (2001) and Olesen et al. (2004a) (Table 11).

Table 11: Use of water and bedding for the manure management types considered in ModelFarm (according to Poulsen et al., 2001 and Olesen et al., 2004a).

Manure management	Bedding [kg animal^{-1} d^{-1}]		
	Cow	Heifer	Pig
Separate	1.2	0.9	0.15
Slurry	0.0	0.0	0.0
Deep litter	12.0	4.2	3.0

In the IPCC methodology, no emission factor is specified for ammonia volatilisation from animal housing. Therefore, ammonia emissions are based on standard values reported by Poulsen et al. (2001) and Olesen et al. (2004a).

Table 12: Ammonia emissions from different housing types in % of excreted N (Poulsen et al., 2001, Olesen et al., 2004a).

Housing type	Manure	Ammonia emission [%]
Tied stall	Separate	5
Tied stall	Slurry	3
Loose housing system with slatted floor	Slurry	8
Loose housing system with litter floor	Deep litter	6

The methane emissions from slurry- and straw-based and deep litter systems will be calculated with the IPCC methodology calculated according to the following equation (Equation (5))

$$E_{CH_4} = k_{MCF} \cdot VS \cdot B_o \cdot 0.67 \quad (5)$$

where E_{CH4} is the methane emission (kg CH_4), k_{MCF} is the methane conversion factor calculated according to the factors in Table 13 of IPCC (2000), VS is the amount of volatile solids or organic matter input to the house (kg), B_o is the maximum methane producing capacity ($m^3\ kg^{-1}$ VS) of the different manure types (Table 14) and 0.67 converts from volume to kg of methane ($kg\ m^{-1}$). The methane conversion factor depends on the climate region and on duration of storage in the house. All the model farms belong to the cool region, and only the cool region conversion factors are therefore shown in Table 13. There is also a difference between the values proposed in the original methodology (IPCC, 1997) and in the Good Practice Guidelines (IPCC, 2000) which are currently again under consideration. For the calculation of the model farms the current emission factors of IPCC (2000) will be used.

Table 13: Methane conversion factors (k_{MCF}) for different manure types stored in animal houses in a cool region (IPCC, 1997, 2000).

Manure type	Storage duration	k_{MCF}	Source
Manure	≤ 1 month	0.05	IPCC (1997)
Manure	> 1 month	0.10	IPCC (1997)
Liquid manure, slurry, deep litter	≤ 1 month	0.00	IPCC (2000)
Liquid manure, slurry, deep litter	> 1 month	0.39	IPCC (2000)
Solid manure		0.01	IPCC (2000)

Table 14: Methane production factors (B_o-factors) of different manure types.

Substrate	B_o [$m^3\ CH_4\ kg^{-1}$ VS]
Slurry from heifers (6-25 months)	0.182
Slurry from cows and bulls	0.200
Cattle solid manure	0.250
Slurry from pigs	0.270
Pig solid manure	0.290

Nitrous oxide emissions are estimated according to the IPCC (2000) methodology from both slurry- and straw-based (deep litter) systems as a proportion of excreted N. The N_2O emission factor (EF) also depends on the manure storage duration in the house (Table 15).

Table 15: Nitrous oxide emission factors for slurry- and straw-based systems in house (IPCC, 2000).

Manure type	Storage duration	EF [kg N_2O-N kg^{-1} N]
Slurry and liquid manure	≤ 1 month	0.001
Slurry and liquid manure	> 1 month	0.001
Solid manure and deep litter	≤ 1 month	0.005
Solid manure and deep litter	> 1 month	0.020

For the different manure handling of these housing systems appropriate emission factors of manure stores also have to be considered. The CH_4 emissions were calculated by ModelFarm according to Equation (5) and the emission factors of IPCC (2000) in Table 16. The N_2O emissions will be calculated using the emission factors of Table 17 depending on manure type and treatment. As there is no ammonia emission factor from manure stores specified in the IPCC methodology, the NH_3 emissions will be calculated by the emission factors (EF) in

Table 18 according to Olesen et al. (2004a) and basing on Hutchings et al. (2001), Poulsen et al. (2001), Sommer & Dahl (1999), Sommer (2001) and CORINAIR (2002).

Table 16: Methane conversion factors (k_{MCF}) for stored manure in a cool region (IPCC, 2000)

Manure type	Treatment	k_{MCF}
Slurry	None	0.390
Slurry	Digestion	0.100
Liquid	None	0.390
Solid	None	0.015
Solid	Composting	0.005
Deep litter	None	0.005

Table 17: Nitrous oxide emission factors for the storage of different manure types as proportion of N content (IPCC, 1997, 2000).

Manure type	Treatment	k_{MCF}
Slurry	None	0.001
Slurry	Digestion	0.001
Liquid	None	0.001
Solid	None	0.020
Solid	Composting	0.020
Deep litter	None	0.020

Table 18: Ammonia emission factors for manure stores depending on manure type, treatment and storage cover (Olesen et al., 2004a).

Manure type	Treatment	Cover	EF
Slurry	None	None	0.080
Slurry	None	Solid	0.008
Slurry	None	Straw	0.016
Slurry	None	Crust	0.024
Liquid	None	None	0.160
Liquid	None	Solid	0.020
Liquid	None	Straw	0.040
Liquid	None	Crust	0.040
Solid	None	None	0.100
Solid	Composting	None	0.200
Deep litter	None	None	0.200
Deep litter	Composting	None	0.200

3.2.2 Abatement costs

For the calculation of the costs for the implementation of slurry- and straw-based housing systems for dairy cows and fattening pigs the cost differences of various animal houses, labour costs for bedding and manure handling as well as of manure application have to be considered. Table 19 shows the animal housing and labour costs for dairy cows in tied and loose housing systems. The loose housing systems are differentiated in two straw-based systems (FYM and deep litter) and one slurry-based system whereas the tied system only distinguishes between a common FYM and slurry system. In Table 20 the animal housing and labour costs for typical housing systems of fattening pigs are presented. For fattening pigs only one slurry-based system with slatted floor will be compared with one deep litter animal house representing the straw-based system. The calculated costs are based on KTBL (2004, 2006) and Mittelfränkische Landwirtschaftsverwaltung (2004) and reflect the average cost conditions in Germany applicable for the construction and operation of animal houses of the defined model farms. The application and labour cost and diesel use connected with the application of slurry, liquid and solid manure are summarised in Table 21. The higher or lower costs for the use of mineral fertiliser will be adapted according to the different nitrogen losses from manure of the different housing systems. The use of mineral fertiliser will be adapted by ModelFarm in compliance with the crop demand and the different crop rotations.

Table 19: Animal housing and labour costs for animal husbandry of dairy cows in straw- and slurry-based loose housing and tied systems (basing on KTBL, 2004, 2006 and Mittelfränkische Landwirtschaftsverwaltung, 2004).

Housing system		Manure	Animal housing costs [€ cow^{-1} a^{-1}]	Labour costs [€ cow^{-1} a^{-1}]
Loose housing system	Straw-based	FYM	115	362
		Deep litter	109	455
	Slurry-based	Slurry	123	351
Tied system	Straw-based	FYM	143	373
	Slurry-based	Slurry	163	332

Table 20: Animal housing and labour costs for animal husbandry of fattening pigs in straw- and slurry-based housing systems (basing on Mittelfränkische Landwirtschaftsverwaltung, 2004 and KTBL, 2004, 2006).

Housing system	Animal housing costs [€ pig^{-1} a^{-1}]	Labour costs [€ pig^{-1} a^{-1}]
Straw-based (deep litter)	11.3	36.6
Slurry-based with slatted floor	15.2	30.7

Table 21: Costs for the application of slurry, liquid and solid manure, labour costs and use of diesel (according to KTBL, 2004).

	Manure amount	Costs [€ ha^{-1}]			Labour costs [€ ha^{-1}]	Diesel use [l ha^{-1}]
		Fixed costs	Variable costs	Total costs		
Slurry	20 m^3	16.64	22.09	38.73	26.4	6.9
Liquid manure	20 m^3	19.76	18.68	38.44	32.1	7.6
Solid manure	20 t	49.45	33.12	82.57	40.0	15.0

3.3 Frequency of manure removal

On average only about one third of, for instance, the dairy cow feed N is transformed into the protein of animal products, while the rest is excreted in urine and faeces (Kirchgessner et al., 1994). About one fourth of this N may be emitted as ammonia into the atmosphere directly after excretion from the animal and during manure storage in animal housing. If excrements are not removed immediately from fouled animal housing surfaces and manure pits into closed manure stores, NH_3 can be emitted from housing systems with the exhaust air.

In general, in animal houses the volatilisation of ammonia is related to the NH_4^+ concentration, pH and surface area of the manure (slurry) store in the house, to the area fouled by the animals and to the temperature and ventilation rate in the housing system. Ammonia is emitted from both the floors fouled with urine and faeces and from the slurry channels and pits under slatted floors. The larger the area fouled by the animals, the larger is the NH_3 loss. Thus, decreasing the surface area of animal housing fouled by manure has a high ammonia emission reduction potential. These NH_3 emissions from cattle, pig or poultry housing systems can be reduced through regular (weekly, daily or several times per day) washing or scraping the floor and therewith more frequent removal of manure to a closed outdoor storage system. A number of systems have recently been tested involving the regular removal of the slurry from the floor to a (covered) store outside of the building by either flushing with water, acid or diluted slurry, or scraping with or without water sprinklers (UNECE, 1999). Many of these options for reducing NH_3 and also CH_4 emissions from housing can only be implemented in newly built houses but there is some scope for implementing them in existing animal houses.

A regular manure removal for livestock housing may directly reduce NH_3 volatilisation and indirectly reduce NH_3 emissions by reducing the transformation by urease on the slats and solid floors. Various studies report a NH_3 emission reduction potential of 20-40 % by reduced contaminated surfaces (Voermans & Verdoes, 1994; Voermans et al., 1995; Voermans & Hendriks, 1996; Zeeland & Verdoes, 1998; Zeeland et al., 1999; Verdoes et al., 2001). Vacuum systems, flushing gutters and scraping systems (with or without toothed scrapers) have been evaluated to be the most efficient removal techniques for cattle, pig and poultry systems. Measurements of Navarotto et al. (2002) show that vacuum systems may reduce NH_3 emissions of fattening pigs by on average 27 % compared to a reference system.

Flushing systems reduce NH_3 emissions of fattening pigs by 20-40 % (Zeeland & Verdoes, 1998), minimise odours within buildings and are easily adapted to many existing animal housing systems. Labour requirements are low. Kiuntke et al. (2001) even measured on average a reduction of 45 % NH_3. According to Meissner & van den Weghe (2003) the reduction of CH_4 emissions of animal houses with fattening pigs compared to a reference system range in winter between 80 and 88 % and in summer by on average 75 %. If acids are used for flushing, emissions will further decrease because of a change in pH. Manure pH can be lowered by adding e.g. nitric acid. Other acids that can be used are phosphoric acid, sulphuric acid and hydrochloric acid, but nitric acid is the most popular since the other acids affect manure quality.

Scraping systems, especially with toothed scraper, have a significant (50 %) potential to reduce NH_3 from different animal housing systems. Therefore, this system was chosen to be calculated for representative farms by the ModelFarm model because it exhibited the highest mitigation potential.

According to UNECE (1999) there are no effective techniques for straw-based systems but they are available for slurry-based systems. Cubicle houses represent the most commonly researched slurry-based systems for dairy cows, where ammonia emissions arise from the manure pit beneath the floor and from urine- and manure-fouled slatted and/or solid floors. Table 22 compares the reduction of ammonia emissions of different housing types with cubicle houses as reference cattle system. Emissions from fully slatted pig houses were taken as the reference in UNECE (1999), although in some countries these systems are already banned for animal welfare reasons. This reference system was compared with other housing types for fattening pigs in Table 23, for farrowing sows including pigs in Table 24, for mating and gestating sows in Table 25 and for weaners in Table 26 (UNECE, 1999).

Table 22: Ammonia emissions and costs of different cattle housing in the Netherlands (UNECE, 1999).

Housing type	**Reduction [%]**	**Extra investment costs [Euro cow place^{-1}]**	**Extra costs [Euro cow^{-1} a^{-1}]**
Cubicle house (Reference)	0	Reference	
Tied system [1)]	40	-/-	-/- [3)]
Tied system only during winter time [2)]	60	-/-	-/- [3)]
Grooved floor	50	375	55
Flushing system without acid several times a day Scraper / slurry systems	50	217	31 102 - UK
Solid floor with straw bedding [2)]	0	-/-	-/-

[1)] Tied systems are not favoured for animal welfare reasons.

[2)] Systems with straw are favoured for animal welfare reasons. Emissions depend on the amount of straw used. Too little straw may increase emissions.

[3)] Difficult to quantify. In any case, labour costs will be higher.

Table 23: Techniques, reductions and costs of low-emission housing systems for fattening pigs [*)] (UNECE, 1999).

Housing type	**Reduction [%]**	**Extra investment costs [Euro pig place^{-1}]**	**Extra costs [Euro pig^{-1} a^{-1}]**
Fully slatted floor (Reference)	Reference	Reference	Reference
Partly slatted (some 50 %) floor	20	5	-/- 8.27 - UK
Vacuum system	25	10	4
Partly slatted floor - metal slats	40	20 - NL 57.5 - UK	6 - NL 7.82 - UK
Partly slatted, external alleys (width 1.3-1.5 m)	20	5	4
Flushing system by gutters	45	50	17
Flushing system with acid	55	54	11
Flushing system with clarified aerated slurry	55	55	12 17.21 - UK
Manure cooling system (to 12 °C max.) [1)]	60	56	9
Partly slatted floor - metal slats plus reduced manure pit surface to max. 0.18 m^2	65	5	0.2
Solid floor with straw bedding [2)]	0	-/-	-/-

[*)] Emissions and reductions refer to experience in the Netherlands. Costs are for Netherlands, unless stated that they are for the UK.

[1)] A readily available source of groundwater is required and the system may not be allowed where drinking water is extracted.

[2)] Systems with straw are favoured for animal welfare reasons. Emissions depend on the amount of straw.

Table 24: Techniques, reductions and costs of low-emission housing systems for fattening pigs (farrowing sows including pigs) (UNECE, 1999).

Housing type	Reduction [%]	Extra investment costs [Euro pig place^{-1}]	Extra costs [Euro pig^{-1} a^{-1}]
Fully slatted floor (Reference)	Reference	Reference	Reference
Partly slatted (some 50 %) floor	30	-/-	-/-
Vacuum system	40	-/-	-/-
Water / manure channel pit surface to max. 0.80 m^2	50	57	/
Flushing system with clarified aerated slurry	50	480	95
Flushing system by gutters	60	511	82
Flushing system with acid	60	469	83
Manure cooling system (to 12 °C max.) [1)]	70	288	51
Solid floor with straw bedding [2)]	0	-/-	-/-

[1)] A readily available source of groundwater is required and the system may not be allowed where drinking water is extracted.

[2)] Systems with straw are favoured for animal welfare reasons. Emissions depend on the amount of straw.

Table 25: Techniques, reductions and costs of low-emission housing systems for fattening pigs (mating and gestating sows) (UNECE, 1999).

Housing type	Reduction [%]	Extra investment costs [Euro pig place^{-1}]	Extra costs [Euro pig^{-1} a^{-1}]
Partially slatted with individual stall or group housing system without straw (Reference)	Reference	Reference	Reference
Partly slatted, external alley (width 1.3-1.5)	30	5	-/-
Flushing system by gutters	40	154	26
Small channel, manure pit surface to max. 0.5 m^2 per sow with or without vacuum system	45	17	3
Flushing system with clarified aerated slurry	50	140	30
Manure cooling system (to 12 °C max.) [1)]	50	107	19
Flushing system with acid	60	131	25
Solid floor with straw bedding [2)]	0	-/-	-/-

[1)] A readily available source of groundwater is required and the system may not be allowed where drinking water is extracted.

[2)] Systems with straw are favoured for animal welfare reasons. Emissions depend on the amount of straw.

Table 26: Techniques, reductions and costs of low-emission housing systems for weaners (UNECE, 1999)

Housing type	Reduction [%]	Extra investment costs [Euro pig place^{-1}]	Extra costs [Euro pig^{-1} a^{-1}]
Fully slatted floor (Reference)	Reference	Reference	Reference
Partly slatted (some 30 %)	40	-/-	-/-
Vacuum system	40	-/-	-/-
Scrapers (with urine drainage)	50	65	12
Flushing system with gutters	45	250	4
Flushing system with acid	55	36	6
Water / manure channel, manure pit surface to max. 0.15 m^2	65		
Manure cooling system (to 12 °C max.) [1)]	60	24	4
Solid floor with straw bedding [2)]	0	-/-	-/-

[1)] A readily available source of groundwater is required and the system may not be allowed where drinking water is extracted.

[2)] Systems with straw are favoured for animal welfare reasons. Emissions depend on the amount of straw.

3.3.1 Definition of mitigation measure

For several animal categories design modifications of livestock houses are possible to prevent or reduce greenhouse gas emissions. A reduction of emissions can be achieved if either the surface area of manure exposed to the air is reduced or if the slurry is frequently removed and placed in a covered storage.

A change in the frequency of manure removal from animal housing into the outdoor manure storage mainly influences NH_3 as well as CH_4 emissions. The methane emissions from slurry-based manure management systems strongly depend on the temperature of the slurry. A higher storage temperature will increase emissions and there is a non-linear increase in emissions, which in the FarmGHG and ModelFarm model is calculated by the Arrhenius equation (Equation (6)). The Arrhenius equation describes the temperature dependence of the methane emission rate:

$$f_T = \exp\left(\frac{\Delta E}{R}\left[\frac{1}{T}\right] - \frac{1}{T_{ref}}\right) \tag{6}$$

where ΔE is the enthalpy of formation of -1.22×10^5 J mol^{-1}, R is the gas constant (8.31 J mol^{-1} K^{-1}), T is actual absolute temperature (K), and T_{ref} is the reference temperature (15 °C or 288.15 K).

The calculation of methane emissions is based on the default methodology of Olesen et al. (2004a) according to the following equation (Equation (7)) taking the temperature dependency into account:

$$E_{CH_4} = k_m \cdot f_T \cdot VS \cdot B_0 \cdot 0.67 \tag{7}$$

where VS is the amount of volatile solids or organic matter stored in the house (kg), k_m is a methane conversion factor at the reference temperature (15 °C) (proportion of methane producing capacity used per day), and f_T the temperature function of Equation (6). The parameter k_m is defined to be the standard value $k_m = 0.005$.

The effect of reducing slurry temperature has been demonstrated by Hilhorst et al. (2001), who found a reduction in methane emissions of 30-50 % when decreasing the storage temperature from 20 to 10°C. The slurry is typically stored at higher temperatures in the house pit than in the outside manure store depending on season and climatic conditions of the animal production region. A more frequent removal of the manure from the house to the outside storage can therefore be expected to reduce the methane emissions especially if the outside storage is covered.

A number of additional systems have been tested experimentally involving the regular removal of the slurry from the floor to a covered store outside the building (UNECE, 1999, Döhler et al., 2002). These systems involve either flushing with water, acid or diluted slurry or scraping with or without water sprinklers. But one of the most promising and practical systems to date for a frequent removal of manure involves the use of a scraper running over a solid floor (Swierstra et al., 2001). This appears to produce a clean and therefore less emitting floor surface. It has been shown that using scrapers to clean floors in loose cattle housing can reduce ammonia emissions by approximately 50 % (Swierstra et al., 2001). The ammonia emissions are estimated as a proportion of total-N excreted by the animals with respect to different housing systems (Table 27). The ModelFarm model thus assumes a reduction in ammonia emissions of 50 % for slurry-based cattle housing systems with solid floors based on Poulsen et al. (2001).

Table 27: Ammonia emissions from different loose housing systems in % of excreted N (according to Poulsen et al., 2001).

Loose housing system	Ammonia emission in %
Solid floor, no cleaning	8
Slatted floor	8
Partly slatted floor	6
Solid floor, scrapers	4

Applying a scraper to reduce the surface area fouled by the animals may be speculated also to reduce N_2O emissions as these emissions depend on sites with both aerobic and anaerobic conditions. However, there are no studies to show this effect clearly, and it has therefore been omitted from the model calculations. In addition, there is an electricity demand for operating the scrapers, which is set to 40 kWh cow^{-1} a^{-1}.

The reduction of ammonia emissions by flushing gutters for fattening pigs was estimated to be 45 % (UNECE, 1999; Kiuntke et al., 2001; Table 23). The energy use for the operation of scrapers was estimated to be 5.2 kWh pig^{-1} a^{-1}.

The more frequent removal of manure out of the pits requires neither technical expenditures nor additional energy, as the amount of manure pumped out remains constant. An improved manure removal will be calculated for the dairy model farm DF1 and pig fattening model farm PF1.

3.3.2 Abatement costs

The additional costs for the scraping systems were taken from UNECE (1999) and Eurich-Menden et al. (2002) which were calculated by the extra investment as well as the additional costs for resources (in most cases electricity) and for labour. The extra costs for scraping systems were estimated to be 33.2 € $cow^{-1}a^{-1}$ (according to Swiestra in Eurich-Menden et al., 2002 and in contrast to 102 € $cow^{-1}a^{-1}$ in Table 22) and 17 € $pig^{-1}a^{-1}$ for a flushing system by gutters for fattening pigs. The costs for a higher frequency of pumping operations were set to zero as the manure amount remains constant. Minor changes in labour costs were neglected.

3.4 Improved outdoor manure storage techniques

Ammonia and methane emissions and dependent on the formation of the manure surface additionally nitrous oxide emissions from slurry or FYM stores can be decreased either by reducing the surface per unit volume of manure stores (de Bode, 1991; Sommer, 1992; Hüther, 1999) or by different cover techniques of the manure storage (Sommer, 1992; Sommer & Hutchings, 1995; Wanka et al., 1998; UNECE, 1999; Döhler et al., 2002).

If, for example, lagoons are replaced by tanks, NH_3 and possibly also N_2O and CH_4 emissions may be reduced due to the lower surface area per unit volume. In general, about 90 % of manure's CH_4 potential and about 80 % of NH_4^+-N can be lost to the atmosphere from open lagoons (especially under warm conditions in south Europe).

Table 28 shows the NH_3 emissions of storage tanks compared to lagoons and FYM heaps (Döhler et al., 2002). The results show that the NH_3 emissions of pig slurry storages are higher compared to cattle slurry. This is caused by the higher NH_4^+-N content of pig slurry and by the surface crust of cattle slurry that reduces NH_3 emissions considerably. Thus, the mitigating effect for the replacement of lagoons by tanks would be higher for pig slurry than for cattle slurry.

Table 28: NH_3 emissions during open storage of slurry and FYM (according to Döhler et al., 2002).

	Average of NH_3 emissions in % of total N		
	Cattle	Pig	Comments
Tank	8	15	
Lagoon	15	25	Estimated
FYM	25	25	

The potential to decrease N_2O emissions is less certain on account of a great number of competing effects that need to be considered. But, because of the reduction of the surface area, N_2O mainly produced in the surface crust by nitrification should be lower. According to most of the recent studies no significant reduction of CH_4 emissions is expectable if lagoons are replaced by tanks.

Moreover, smaller volumes of slurry may largely increase direct biogenic emissions from stores by the decreased dilution with rain water but reduce GHG emissions from energy use due to lower amounts of manure that have to be transported and applied.

Due to the fact that manure lagoons represent only 3-4 % of total manure storages for German conditions, the replacement of lagoons by tanks will not be considered for the modelling as a mitigation measure. According to IPCC (1997) the percentage of lagoons in Western Europe is 0 % for all farm types whereas for Eastern Europe a share of 8 % for farms with non-dairy cattle is estimated.

For the storage of cattle manure a storage tank with natural crust has to be defined as reference system whereby it must be considered and recommended that filling and emptying liquid manure storage tanks or lagoons should take place from below the surface of the stored manure to conserve the slurry surface crust (underslat flushing).

The formation of a natural crust serves as a biological cover that can reduce NH_3 and CH_4 emissions. The texture of the surface crust thereby depends on the feeding. An increasing share of maize in the ration, for instance, reduces the development of a surface crust (Berg et al., 2002).

De Bode (1990) reported a reduction of NH_3 emissions with a natural crust by 35 %. Koch (1998) measured a NH_3 reduction potential of 30-40 % compared to reference. Döhler et al. (2002) reported a NH_3 reduction by a natural crust for pig slurry of an average of 30 % (20-70 %) and of 70 % (30-80 %) for cattle slurry depending on the development of the surface crust. Also CH_4 emissions can be decreased by the oxidation of produced CH_4 due to the

aerobic conditions of the crust. Studies have shown that a natural crust additionally reduces odours from dairy storages by 75 %. According to Sommer & Petersen (2002) a natural crust may partly cause substantial increase of the N_2O emissions from nitrification processes in the crust.
Aside from the manure surface reduction per unit volume the most practicable technique to reduce NH_3 emissions from stored slurry is to cover slurry tanks on the one hand with low technology covering such as straw, peat, bark, granulates or floating oil or on the other hand with flexible plastic covers or permanent rigid covers such as a solid lid, roof or tent structure. Low technology covering may in particular reduce ammonia emissions but also CH_4 emissions from stored slurries by preventing contact between the slurry and the air. CH_4 emissions from manure storage depend on the manure type and the conditions in the storage. Covers with straw, peat and bark may change the redox status of the slurry surface like in a natural crust. The cover material may be colonised by aerobic micro-organisms that use ambient air as an oxygen source for nitrification of the slurry borne ammonia. A substantial increase in the N_2O emissions was verified in lab experiments by Hüther & Schuchardt (1998) and Roß et al. (1998). The addition of cover material may also result in higher CH_4 emissions due to the input of additional carbon into the system (Hüther, 1999).
The coverage of the manure storage with straw, peat and bark represents a cheap cover option. For covering of dairy manure at least 4 kg straw m^{-2} and of pig slurry at least 7 kg straw m^{-2} (15-25 cm) is recommended. The straw, peat or bark material can be applied to manure storage tanks using a straw chopping/blowing machine. De Bode (1990) reduced NH_3 emissions by up to 60-70 % by the addition of 4-7.5 kg straw m^{-2}. Döhler et al. (2002) reported a NH_3 reduction by straw addition for a pig and cattle slurry of on average 80 % (70-90 %). This is line with a NH_3 reduction of 70-90 % confirmed in lab and practice experiments by Roß et al. (1998) and Wanka et al. (1998). Wanka & Hörnig (1997) and Wanka et al. (1998) additionally reported a reduction of CH_4 and N_2O emissions for practice slurry tanks.
Also granulates like LECA (light expanded clay aggregates) and macrolite balls or other floating material (e.g. perlite) can be used as cover material (Sommer & Hutchings, 1995). In comparison to straw NH_3 mitigation results are on average a little higher with the use of granulates (de Bode, 1990; Miner & Suh, 1997; Hörnig et al., 1998; Hüther & Schuchardt, 1998; Koch, 1998). A reduction of NH_3 emissions by 70-90 % is estimated. Also Döhler et al. (2002) reported a NH_3 reduction by granulates for a pig and cattle slurry of on average 85 % (80-90 %).
A layer of floating oil (e.g. rape seed oil of 0.5 cm) on the surface can also be used to cover stored slurry (Sommer, 1992). At present, there is little expert knowledge about the GHG mitigation efficiency of oil floating but due to the complete exclusion of air a considerable increase in CH_4 emissions is anticipated.
Flexible covers such as plastic sheeting (e.g. swimming vinyl covering) placed on the surface are mainly used for slurry tanks (but also applicable for manure heaps). In general, NH_3 emissions are significantly reduced by 60-99 %. UNECE (1999) reported a NH_3 reduction up to 60 % whereas Döhler et al. (2002) reported a NH_3 reduction by plastic covers for pig and cattle slurry of on average 85 % (80-90 %). According to Jacobson et al. (1999) impermeable floating plastic covers result in about 99 % emission reduction.
Rigid covers and lightweight roofs are permanent covers that are commonly made of concrete, wood, fibreglass, aluminium or plastic membranes. This type of cover is usually more expensive but lasts much longer than other methods (15-20 years). A significant NH_3 reduction of on average 70 % is reported by Klimont (2001) and of 80 % by UNECE (1999). Döhler et al. (2002) reported a reduction of ammonia emissions by a rigid cover for a pig and cattle slurry of on average 90 % (85-95 %). Apart from a significant reduction of NH_3 emissions, rigid covers also reduce manure storage volumes due to the exclusion of rain water

from the store which additionally reduces GHG emissions from energy use and costs for transport and application of less manure amounts. Less diluted slurry (dependent on average rainfall) would have a higher nutrient value per application potentially increasing losses of ammonia, nitrous oxide and nitrate leaching at a later date (after application). Thus, an adequate manure application method is needed (see 3.5).

3.4.1 Definition of mitigation measure

The impact of different cover techniques on farm level GHG mitigation will be modelled for two dairy model farms (DF1, DF3), both bull fattening (BF1, BF2) and both pig fattening (PF1, PF1) model farms. The reference storage system for cattle manure is an uncovered storage with a natural surface crust whereas for fattening pigs no surface crust is anticipated. The average ammonia emission reduction factors for the different cover techniques were taken from Döhler et al. (2002) who also analysed the results of different international studies (Table 29).
N_2O and CH_4 emissions were neglected for modelling since consistent emission factors for the various cover techniques are currently not available.

Table 29: Average mitigation of NH_3 emissions of different storage cover techniques in % compared to an uncovered slurry tank without natural crust (according to Döhler et al., 2002).

Cover technique	Cattle slurry	Pig slurry
Natural crust	70 (30-80)	30 (20-70)
Straw	80 (70-90)	80 (70-90)
Granulates	85 (80-90)	85 (80-90)
Plastic sheeting	85 (80-90)	85 (80-90)
Rigid cover	90 (85-95)	90 (85-95)

3.4.2 Abatement costs

The calculation of the mitigation costs is based on the average cover costs per m^2 according to the literature survey by Döhler et al., 2002 (Table 30). To unify the different literature data (different storage surface to volume ratios; different specifications in m^2, m^3 etc.) the annual cover costs per m^3 were calculated for three different storage tank size categories with a filling height to diameter ratio of approximately 1 : 4 (Table 31).
The straw cover represents the cover technique with the lowest costs and the drivable concrete ceiling with the highest costs. The specific costs of these diverse cover techniques will be implemented in ModelFarm for the different manure amounts of the various model farms considering an average storage duration of six months.

Table 30: Annual costs for different cover techniques in € per m^2 manure storage surface area (according to Eurich-Menden et al., 2002).

Cover technique	Costs in € $m^{-2}a^{-1}$
Straw	1.2
Granulates	2.5
Plastic sheeting	3.5
Tent roof	5.6
Concrete ceiling	6.2
Drivable concrete ceiling	6.4

Table 31: Annual costs for different cover techniques in € per m^3 manure (according to Eurich-Menden et al., 2002).

Cover technique	**Cover costs in € $m^{-3}a^{-1}$**		
	250 m^3 (83 m^2)	**500 m^3 (143 m^2)**	**1500 m^3 (273 m^2)**
Straw	0.2	0.2	0.1
Granulates	0.4	0.4	0.2
Plastic sheeting	0.6	0.5	0.3
Tent roof	0.9	0.8	0.5
Concrete ceiling	1.0	0.9	0.6
Drivable concrete ceiling	1.1	0.9	0.6

3.5 Manure application techniques

Gaseous emissions from land application of slurries and solid manures account for a large proportion of total ammonia emissions from agriculture. Recent estimates of ammonia emissions from agriculture indicate that around 33 % of total emissions may originate from manure spreading (Pain et al., 1998). Controlling emissions from applications of manures to land is important, because these emissions are generally a main share of total manure emissions and application is the last step of manure handling. Without abatement at the end-of-pipe of manure handling, much of the benefit of abating during housing and storage may be lost. Furthermore, it is very important to minimise these losses at this stage because any ammonia saved during livestock housing or manure storage will be lost as nutrient for crop production if it is not controlled by appropriate field application techniques. Reducing ammonia losses from slurries and solid manures means more nitrogen is potentially available for grass and crop uptake and thus a reduced amount of mineral fertilisers in conventional farms is needed. This reduction of mineral fertilisers use may clearly decrease GHG emissions in respect of the high energy use for fertiliser production.

Most of the strategies to reduce NH_3 rely on the balance of NH_3 and NH_4^+ and on the absorption of NH_4^+ negative charged surfaces. According to Horlacher & Marschner (1990) and Wulf et al. (2002) up to 90 % of NH_4^+-N applied with slurry can be lost through NH_3 emissions, substantially reducing the amount of plant-available N. Techniques to mitigate these emissions include using machinery for decreasing the surface area of slurries (i.e. improved application techniques) and burying slurry or solid manures through incorporation into the soil. The effectiveness of improved application techniques relies on reducing the surface area of slurry exposed to the air, increasing the rate of infiltration into the soil so that ammonium-N adsorbs to clay particles, or reducing air flow over the slurry surface by placement beneath a crop of grass canopy (Pain & Jarvis, 1999).

A number of factors must be taken into account to determine the applicability of each application technique. These factors mainly include: soil type and condition (soil depth, stone content, wetness, travelling conditions), topography (slope, size of field, evenness of ground), manure type and composition (slurry or solid manure) (UNECE, 1999).

A huge number of experiments have been carried out to quantify ammonia emissions after manure application. Research has mainly concentrated on slurry application, but a considerable number of experiments have also been carried out with farmyard manure. But in the framework of this analysis only improved slurry application techniques will be investigated with a focus on ammonia emissions after manure application by trailing hose on arable land and by trailing shoe on grassland compared to a reference technique (broadcasting). N_2O emissions will be considered with respect to the reduced NH_3 emissions but also the increased NH_3 deposition. Changes in the way the manure is applied to agricultural soils are not likely to affect emissions of CH_4. According to study results from Chadwick et al. (2000) and Wulf et al. (2001) CH_4 emissions after slurry application can be neglected. Also Clemens et al. (1997), Velthof et al. (1997) and Weslien et al. (1998) found no significant differences between the application techniques. Thus, possible but marginal differences in CH_4 emissions will not be considered.

In general, the NH_3 reduction efficiency is approximately 25 % for low efficiency techniques (band spreading) and approximately 60-90 % for high efficiency techniques (injection: open slot 60 %, closed slot 80 %), respectively (Frost, 1994; Mulder & Huijsmans, 1994; Lorenz & Steffens, 1997; Huijsmans et al., 1997; UNECE, 1999; Smith et al., 2000; Klimont, 2001; Döhler et al., 2002).

Ammonia emission reduction by band spreading / trailing hose of 10-50 % on arable land and grassland has been reported by most of current studies. Pig slurry NH_3 emissions were reduced by 30-50 % whereas the emissions of cattle manure were reduced by 10-30 %

(UNECE, 1999; Döhler et al., 2002). Generally, the NH_3 mitigation effect increases if the slurry is applied on crops of higher size (>30 cm) or grassland (Döhler et al., 2002).
Trailing shoe application as band spreading technique which is mainly applicable to grassland showed a NH_3 reduction of 30 % on arable land and of 40 % on grassland (UNECE, 1999). Döhler et al. (2002) reported a reduction of NH_3 emissions on grassland of 30-40 % for cattle slurry and 60 % for pig slurry.
For injection with open slot a reduction of NH_3 emissions is reported by up to 90 % (on average 60 %; UNECE, 1999). Döhler et al. (2002b) reported a NH_3 mitigation of 60 % for pig slurry and 80-90 % for cattle slurry. For injection with closed slot an ammonia abatement efficiency of on average 60-80 % is reported by UNECE (1999).
Table 32 compares the applicability and NH_3 mitigation potential of different application techniques reported by the EU project ALFAM (Ammonia Loss from Field-applied Animal Manure; ALFAM, 2002).

Table 32: Practical considerations in selecting spreading technique for ammonia abatement following field application of manure (ALFAM, 2002).

Abatement technique	Manure type	Land use	Restriction on applicability	Reduction in emission
Trailing hose	Liquid manure	Grassland	Slope, size and shape of field. Non-viscous slurry.	10-20 %
		Arable land	As above. Width of tramlines for growing cereal crops.	30-40 %
Trailing shoe	Liquid manure	Mainly grassland	As above. Optimum grass height is about 10 cm.	40-60 %
Shallow injection	Liquid manure	Mainly grassland	As above. Short (recently cut/grazed grass required), not stony or very compacted soils.	60-70 %
Deep injection	Liquid manure	Arable land	As above. Needs high powered tractor.	70-80 %
Incorporation	All manure types	Arable land including grass leys	Land that is cultivated, preferably ploughed.	20-90 %

3.5.1 Definition of mitigation measure

GHG emissions after manure application by trail hose and trail shoe will be calculated in comparison to broadcasting as standard application technique. The reference for manure application techniques is defined as emissions from untreated slurry spread over the whole soil surface with a discharge nozzle and splash-plate (broadcasting). The slurry is thereby forced under pressure through a nozzle, often onto an inclined plate to increase the sideways spread.
Basically, an improved application mainly influences the ammonia volatilisation as well as the fertiliser replacement values of the applied liquid manures and slurries. As in most of the studies NH_3 emissions were measured for a limited time and possible higher N_2O emissions associated with the application technique were not considered, the effect of improved application techniques will not be calculated by simple adapted emission factors, but by changes of ammonia volatilisation and fertiliser replacement values by the ModelFarm model based on results and calculations from ALFAM (2002), MIDAIR (2004) and KTBL (2004).
In general, ammonia volatilisation represents a surface process defined as a product of the volatilisation per unit area of solution and the area of solution exposed to the atmosphere, whereas the total volatilisation from a particular manure application is the instantaneous volatilisation integrated over the duration of the emission event. The total volatilisation can be considered largely to be the result of the competition for ammonium between the processes

driving volatilisation and those determining its removal to other parts of the plant/soil system (ALFAM, 2002). Following field application, the ammonium in solution dissociates reversibly to ammonia. The balance is determined by the chemical composition of the manure, especially the pH, and the extent of the interaction between the applied manure and the soil. The ammonium at the surface of the solution is in dynamic equilibrium with ammonia gas in the air, the balance being determined by the temperature (ALFAM, 2002).
The analysis of ammonia emission after application with different techniques of the ALFAM project has shown that the results from different European countries are generally comparable. Therefore, the results from ALFAM can be used on the one hand throughout countries and on the other hand to calculate rough estimates of ammonia volatilisation under combinations of conditions, which have not been examined through experiments, thus reducing further extensive measurements.
For the modelling of ammonia volatilisation the following variables were considered: moisture content of soil, air temperature, wind speed, manure type (pig or cattle slurry), dry matter content of manure, total ammoniacal nitrogen (TAN) content of manure, application method, application rate of manure and duration from manure application to incorporation.
The modelling of the ALFAM and ModelFarm model with respect to ammonia volatilisation is based on the Michaelis-Menten-type equation (Equation (8)) which calculates the ammonia losses depending on time (Sommer & Ersbøll, 1994; Søgaard, 2002; Olesen et al., 2004a):

$$N(t) = N_{\max} \frac{t}{t + K_m} \tag{8}$$

where $N(t)$ is the cumulative ammonia volatilisation at time t from the start of the experiment, expressed as a fraction of the total ammoniacal N (TAN) applied, N_{max} is the total loss of ammonia (fraction of TAN applied) as time approaches infinity, and the parameter K_m is the time (hours) when $N(t) = ½\, N_{max}$.
Therefore, the loss rate (loss per time unit; Equation (9)) is defined as the derivative of the function in Equation (8).

$$\frac{dN(t)}{dt} = N_{\max} \frac{K_m}{(t + K_m)^2} \tag{9}$$

According to the ALFAM modelling N_{max} and K_m can be defined as real values:

$$N_{\max} = e^{(a_0 + a_1 x_1 + \ldots + a_m x_m)} \tag{10}$$

$$K_m = e^{(b_0 + b_1 x_1 + \ldots + b_m x_m)} \tag{11}$$

where a_0, …, a_m and b_0, …, b_m are model parameters to be estimated or by rewriting these expressions in Equations (10) and (11) correspond to multiplicative relationships with the exponentials of the explanatory variables as factors:

$$N_{\max} = A_0 \cdot A_1^{x_1} \cdot \ldots \cdot A_m^{x_m} \text{ , where } A_i = e^{a_i}\text{ , } i=0,\ldots,m \tag{12}$$

$$K_m = B_0 \cdot B_1^{x_1} \cdot \ldots \times B_m^{x_m} \text{ , where } B_i = e^{b_i}\text{ , } i=0,\ldots,m \tag{13}$$

Within the ALFAM project N_{max} and K_m were estimated by the use of Equations (12) and (13) and parameter estimates of Table 33 and Table 34.

Table 33: Parameter estimates related to N_{max} and confidence limits for the ALFAM model of ammonia (*cf*. Equation (12)).

Experimental factor	**Interpretation of the corresponding parameter (as a multiplicative factor)**	**Parameter estimate**	**Approximate 95% confidence limits**	
None	Common factor	A_0 = 0.0495	0.0078	0.3153
Moisture content of soil	Wet soil (compared to dry soil)	A_1 = 1.102	1.028	1.181
Air temperature	Increase per °C	A_2 = 1.0223	1.0175	1.0273
Wind speed	Increase per m s^{-1}	A_3 = 1.0417	1.0178	1.0662
Manure type	Pig slurry (compared to cattle slurry)	A_4 = 0.856	0.773	0.947
Dry matter content of manure	Increase per % dry matter	A_5 = 1.108	1.087	1.129
TAN content of manure	Decrease per g N kg^{-1}	A_6 = 0.828	0.786	0.872
Application method	Band spread/trailing hose	A_7 = 0.577	0.496	0.673
	Trailing shoe	A_8 = 0.664	0.261	1.685
	Open slot injection	A_9 = 0.273	0.198	0.377
	Closed slot injection	A_{10} = 0.543	0.327	0.901
	Pressurised injection	A_{11} = 0.028	0.012	0.068
Application rate of manure	Decrease per t ha^{-1} or m^3 ha^{-1}	A_{12} = 0.996	0.993	0.998
Manure incorporation	No incorporation (*versus* shallow cult.)	A_{13} = 11.3	1.8	72.0
Ammonia loss measurement technique	Wind tunnel	A_{14} = 0.528	0.436	0.640
	Micromet (*versus* JTI Equilibrium concentration method)	A_{15} = 0.578	0.470	0.710

Table 34: Parameter estimates related to K_m and confidence limits for the ALFAM model of ammonia (*cf*. Equation (13)).

Experimental factor	**Interpretation of the corresponding parameter (as a multiplicative factor)**	**Parameter estimate**	**Approximate 95% confidence limits**	
None	Common factor	B_0 = 1.038	0.606	0.3153
Moisture content of soil	Wet soil (compared to dry soil)	B_1 = 1.102	0.967	1.181
Air temperature	Decrease per °C	B_2 = 0.960	0.951	1.0273
Wind speed	Decrease per m s^{-1}	B_3 = 0.950	0.913	1.0662
Manure type	Pig slurry (compared to cattle slurry)	B_4 = 3.88	3.18	0.947
Dry matter content of manure	Increase per % dry matter	B_5 = 1.175	1.134	1.129
TAN content of manure	Increase per g N kg^{-1}	B_6 = 1.106	1.004	0.872
Application method	Band spread/trailing hose	B_7 = 1*	-	-
	Trailing shoe	B_8 = 1*	-	-
	Open slot injection	B_9 = 1*	-	-
	Closed slot injection	B_{10} = 1*	-	-
	Pressurised injection	B_{11} = 1*	-	-
Application rate of manure	Increase per t ha^{-1} or m^3 ha^{-1}	B_{12} = 1.0177	1.0127	1.0227
Manure incorporation	No incorporation (*versus* shallow cult.)	B_{13} = 1*	-	-
Ammonia loss measurement technique	Wind tunnel	B_{14} = 1.48	1.04	2.08
	Micromet (*versus* JTI Equilibrium concentration method)	B_{15} = 2.02	1.38	2.94

* Parameter fixed to 1 due to very low level of significance (P > 0.4)

According to Olesen et al. (2004a) N_{max} can also be calculated by the following equation:

$$N_{\max} = N_x \cdot N_{TAN} \tag{14}$$

where N_x is the share of the maximum NH_3 loss of the total applied NH_3-N.
If this dependency is considered, Equation (14) can also be described for N_{vol} as ammonia volatilisation (in kg N ha^{-1}) with t as the duration (hours) from application to incorporation (500 hours if not incorporated) in the following way:

$$N_{Vol} = \frac{N_x \cdot t}{t + K_m} \cdot N_{TAN} \tag{15}$$

with the following definition of N_x and K_m by use of Equations (10) and (11):

$$N_x = e^{(-6.5757 + A_{moist} + 0.0221T + 0.0409u + A_{man} + 0.1024DM - 0.1888TAN + A_{apply} - 0.00433M + A_{inc})} \tag{16}$$

$$K_m = e^{(0.37 + B_{moist} - 0.0409T - 0.0517u + B_{man} + 0.1614DM + 0.1614TAN + 0.0175M)} \tag{17}$$

with:

A_{moist}, B_{moist}:	function of soil moisture (dry soils = 0; wet soils: A_{moist} = 0.0971, B_{moist} = 0.0974)
T:	mean temperature in °C (depending on date of application)
u:	wind speed in m s^{-1} (set to 2 m s^{-1})
A_{man}, B_{man}:	function of manure type (slurry = 0; liquid manure and digested slurry: A_{man} = –0.156, B_{man} = 1.3567)
DM:	dry matter (DM) content of manure in %
TAN:	TAN content of manure in %
A_{apply}:	factor for application technique (broadspread: 3.5691; trail hose: 3.0198; trail shoe: 3.1591; injection: 2.9582)
A_{inct}:	factor for incorporation (time from application to incorporation; shallow incorporation = 0 and if not incorporated = 2.4291)
M:	rate of applied manure in t ha^{-1}

For the modelling in MEACAP the common factor (Table 33) was estimated to be –7.08635 instead of –6.5757 of Olesen et al. (2004a) to avoid NH_3 losses in the calculation of more than 100 % of total NH_4^+-N:

$$N_x = e^{(-7.08635 + A_{moist} + 0.0221T + 0.0409u + A_{man} + 0.1024DM - 0.1888TAN + A_{apply} - 0.00433M + A_{inc})} \tag{18}$$

Within the calculation in ModelFarm N_2O emissions after slurry application will be reduced by the N amount lost by ammonia volatilisation whereas N_2O emissions of 1 % from NH_3 deposition will be considered. As already mentioned CH_4 formation after field application seems not to take place so that possible changes in CH_4 emissions will be neglected.
In addition to the emissions after manure application, the different use of fossil fuels for operating must be also considered when calculating the mitigation efficiency of the different application techniques. The fuel consumption depends on the energy requirements for the field operation (depending on manure amounts per ha, transportation distances etc.), the efficiency of the transmission and tractive efficiency, the fuel efficiency of the power source and the type of fuel used. Fuel amounts of the different application techniques, which were used for the modelling in ModelFarm and that are based on KTBL (2004) are shown in Table 35. Since the energy use of the trailing shoe technique is not available in KTBL (2004) the figures of trailing hose will be taken for modelling (Table 35).

The impact of improved manure application techniques on GHG mitigation will be calculated by ModelFarm for two dairy model farms (DF1, DF3), both bull fattening (BF1, BF2) and both pig fattening (PF1, PF1) model farms.

Table 35: Use of diesel, oil and electricity per m^3 manure for the application techniques broadcasting, trailing hose, trailing shoe and injection according to KTBL (2004).

	Broadcasting	Trailing hose[1)]	Trailing hose[2)]	Trailing shoe[1)]	Trailing shoe[2)]	Injection
Diesel (l m^{-3})	0.2770	0.2930	0.2855	0.2930	0.2855	0.8560
Oil (m^{-3})	0.0030	0.0030	0.0030	0.0030	0.0030	0.0085
Electricity (kWh m^{-3})	—	—	0.0835	—	0.0835	—

[1)] application from storage [2)] application from field with additional manure transport to field

3.5.2 Abatement costs

New manure application techniques that recently have been developed to reduce ammonia emissions cause higher costs than conventional broadcast spreading techniques with splash plate. These costs can be divided into the higher initial capital investment for the more costly new equipment and higher costs for labour, and for the increased use of operating resources for the more draught force that is required for these techniques. In general, the economics mainly depend on the machine costs and the time required for manure application. Within the ALFAM project the machine costs were calculated by the operating costs, depreciation, interest on capital and insurance. The costs of the field application of manure were calculated, taking into account the hourly-based costs of labour and machine for a given piece of work/task. The time necessary for field application of manure depends on several operational variables such as field area and dimensions, working speed, working width, applied manure amount per hectare, distance to manure store and work system (e.g. manure transport to the field in separate tanks). The model CAESAR (Computer simulation of the Ammonia Emission of Slurry application and incorporation on Arable land) developed for the analysis of manure application activities and efficiency (Huijsmans & De Mol, 1999) was used for ALFAM modelling to simulate a range of manure application operations and to calculate their associated costs at the European level. The CAESAR model enables the calculation of the time requirement for the field application of manure by allocating the time spent on specific work components, e.g. spreading, turning, transport and loading.

Table 36 shows the manure application costs that were used for modelling within the ALFAM project taking into account the machine costs and the annual operating time for manure application. These costs are on average 7.2 € per m^3 applied manure and range on European average from 2.8 to 14.5 € m^{-3} dependent on application technique and manure amount applied per year. Table 36 shows that the cost difference between broadcast spreading and improved application techniques mainly decreases with the increasing farm size so that the variation of the size categories is higher than the variation between the application techniques. But on average of all farm sizes, manure application by trailing hose, trailing shoe, shallow injection and arable land injector costs approximately 2 € m^{-3} more than by broadcast spreading.

The calculation method and the costs are similar to the current results of Döhler et al. (2002) that will be used for the modelling in ModelFarm (Table 37). Due to the reduction of machinery costs for some of the application techniques in recent years the cost per m^3 manure are in total lower but show the same dependencies from the annual amount of manure applied at the farm level. In total, the cost assessment by Döhler et al. (2002) seems to be more realistic as the extra costs increase clearly depending on the construction type of the application techniques (broadcasting < trailing hose < trailing shoe < injection, Table 37).

Table 36: Costs of manure application in € per applied m^3 by various techniques for farms with manure production of 500-3000 m^3 per year (ALFAM, 2002).

Manure production [m^3 $year^{-1}$]	Application techniques				
	Broadcast spreading	Trailing hose	Trailing shoe	Shallow injector	Arable land injector
500	8.46	14.04	13.06	14.53	13.41
1000	5.07	7.86	7.58	8.60	8.03
2000	3.38	4.78	4.84	5.63	5.35
3000	2.82	3.75	3.92	4.64	4.45

Table 37: Manure application costs for broadcasting, trailing hose, trailing shoe and injection. Adapted from Döhler et al. (2002).

Application technique	Costs for annual amounts of 500 m^3	Extra costs compared to broadcasting	Costs for annual amounts of 1000 m^3	Extra costs compared to broadcasting	Costs for annual amounts of 3000 m^3	Extra costs compared to broadcasting
	[Euro m^3]					
Broadcasting	5.2	—	3.8	—	2.2	—
Trailing hose	6.9	1.7	5.5	1.6	3.0	0.7
Trailing shoe	9.3	4.1	7.5	3.6	3.8	1.6
Injection	10.9	5.8	8.9	5.1	4.5	2.2

3.6 Slow- and controlled-release fertilisers and fertilisers with urease or nitrification inhibitors

In recent years, it has been the aim of science and also of the fertiliser industry to develop special types of fertilisers avoiding or at least reducing N losses through immobilisation, volatilisation, nitrification, denitrification and leaching, in addition to the production of conventional nitrogen-containing fertiliser types (ammonium sulphate, ammonium nitrate, calcium ammonium nitrate, ammonium sulphate nitrate, urea, DAP, and NP and NPK fertilisers) (Joly, 1993; Trenkel, 1997).

These special types of fertilisers can be divided into:

- slow-release and controlled-release coated/encapsulated fertilisers, and
- fertilisers with nitrification and urease inhibitors or stabilised fertilisers.

Shoji & Gandeza (1992) judged that an ideal fertiliser should have at least the following three characteristics:

- it should only need one single application throughout the entire growing season to supply the necessary amount of nutrients for optimum plant growth,
- it should have a high maximum percentage recovery in order to achieve a higher return to the production input, and
- it should have minimum detrimental effects on soil, water and atmospheric environments.

Slow- and particularly controlled-release as well as stabilised fertilisers can meet these requirements for an ideal fertiliser to a considerable extent.

The delay of initial availability, or extended time of continued availability of slow- and controlled-release fertilisers, might occur through a variety of mechanisms. These include controlled water solubility of the material (by semipermeable coatings, occlusion, or by inherent water insolubility of polymers, natural nitrogenous organics, protein materials, or other chemical forms), by slow hydrolysis of water-soluble low molecular weight compounds, or by other unknown means (AAPFCO, 1995). For example, polyolefin-coated fertilisers are a type of controlled-release fertiliser where fertiliser granules are covered with a thermoplastic resin. The release of the N fertiliser is temperature dependent and is not controlled by hydraulic reactions or microbial attack of the coating.

The use of controlled-release fertilisers may improve N-use efficiency by matching nutrient release with crop demand, reducing NO_3^--leaching and N_2O losses. Many of the results in the literature indicate that controlled-release fertilisers are useful for the reduction of N_2O emission from fertilised soils. Since this type of fertiliser is more expensive than conventional fertilisers, it is used more widely in horticulture compared to in agriculture. This is in contrast to the use of the less expensive fertilisers with inhibitors (urease and nitrification inhibitors) which are used in the horticultural sector but are also common in agricultural practice. Therefore, the modelling will focus on the mitigation effect of fertilisers with inhibitors.

Urease inhibitors prevent or depress over a certain period of time the transformation of amide-N in urea to ammonium hydroxide and ammonium. They do so by slowing down the rate at which urea hydrolyses in the soil, thus avoiding or reducing volatilisation losses of ammonia to the air (as well as further leaching losses of nitrate or emissions of nitrous oxide by nitrification and/or denitrification). Urease inhibitors therefore inhibit for a certain period of time the enzymatic hydrolysis of urea, which depends on the enzyme urease (Farm Chemicals Handbook '96, 1996). Thus, the efficiency of urea and nitrogen fertilisers containing urea (e.g. urea ammonium nitrate solution UAN) can be increased. As the use of urease inhibitors is mainly reserved to fertilisers with urea and as the share of fertilisers with urea is still low (15 % in Germany 2004) and although the share of urea fertilisers of total N fertilisers increases, the modelling will focus on the mitigation effect of fertilisers with nitrification inhibitors.

Nitrification inhibitors are compounds that delay bacterial oxidation of the ammonium-ion (NH_4^+) by depressing over a certain period of time the activities of *Nitrosomonas* bacteria in the soil. They are responsible for the transformation of ammonium into nitrite (NO_2^-) which is further changed into nitrate (NO_3^-) by *Nitrobacter* and *Nitrosolobus* bacteria. The objective of using nitrification inhibitors is, therefore, to control leaching of nitrate by keeping nitrogen in the ammonia form longer, to prevent denitrification of nitrate-N and N_2O emissions from nitrification and denitrification and thus to increase the efficiency of nitrogen applied (Granli & Bockman, 1994; Mosier et al., 1996; Trenkel, 1997; Weiske et al., 2001).
Several studies have shown that the use of different nitrification inhibitors can reduce N_2O emissions from mineral fertilisers considerably (Table 38).

Table 38: Inhibition of nitrous oxide emissions after use of different nitrification inhibitors (according to Michel & Wozniak, 1998 and Weiske, 2001).

Nitrification inhibitor	Fertiliser	Crop	N_2O reduction in %	Reference
Nitrapyrin	ammonium sulphate	—	93 %	Bremner & Blackmer, 1978
Nitrapyrin	urea	—	96 %	Bremner & Blackmer, 1978
Nitrapyrin	urea	maize	60 %	Bronson et al., 1992
Calcium carbide	urea	maize	55 %	Bronson et al., 1992
DCD	liquid manure	grassland	50-88 %	de Klein & van Logtestijn, 1994
DCD	ammonium sulphate	grassland	40-92 % [1)]	Skiba et al., 1993
DCD	urea	spring barley	82 %	Delgado & Mosier, 1996
POCU [2)]	urea	spring barley	~71 %	Delgado & Mosier, 1996
DCS [3)]	ammonium sulphate	grassland	62 %	Skiba et al., 1993
DCS [3)]	ammonium sulphate	carrot	28 %	Minami, 1994
DMPP [4)]	ammonium sulphate nitrate	spring barley, maize, winter wheat	51 %	Weiske et al., 2001

[1)] Measurement of 2-8 days after application, potassium nitrate under dry conditions
[2)] POCU = polyolefin coated urea
[3)] DCS = N-(2,5-Dichlorophenyl) succinic acid monoamide
[4)] DMPP = 3,4-dimethylpyrazole phosphate

3.6.1 Definiton of mitigation measure

Most of the fertiliser N applied to soils in the form of NH_4^+ or NH_4^+-producing compounds is usually oxidised quite rapidly to NO_3^- by nitrifying microorganisms. Through the use of specific nitrification inhibitors the eco-efficiency of N may be increased by decreasing NO_3^- losses caused through leaching and denitrification. Recently, nitrification rather than denitrification has been recognised as the major source of N_2O from arable soils (Bremner, 1997; Smith et al., 1997; Mosier, 1998), and interest has increased in new, relatively stable nitrification inhibitors that are highly effective in blocking nitritation, some at extremely low concentrations, when applied to soils in conjunction with N fertilisers. This promising strategy for crop production has been developed based on the specific inhibition of nitritation by the nitrification inhibitors dicyandiamide (DCD), 1-H-1,2,4 triazole (TZ), 3-methyl pyrazole (3MP) and 3,4-dimethylpyrazole phosphate (DMPP) being licensed in Europe. The combination of N fertiliser with DMPP is available on the market as ENTEC granules (Pasda et al., 2001; Zerulla et al., 2001).

The addition of DMPP to ammonium sulphate nitrate (ASN) fertiliser resulted in 51 % lower N_2O emissions during the vegetation season, when comparing plots which received fertiliser with and without nitrification inhibitor (Weiske et al., 2001; Table 38). Generally, the calculation of N_2O emissions in ModelFarm are based on the IPCC emission factor (1.25 % of the N goes into N_2O). This emission factor will be reduced (to 0.6375 %) to reflect 51 % lower N_2O emissions after nitrification inhibitor addition. A treatment of manure with nitrification inhibitors will be neglected although recent studies confirm a similar N_2O mitigation effect if manure is applied together with DMPP (Merino et al., 2005).
Furthermore, the energy use and GHG emissions caused by the production of the nitrification inhibitor will be calculated considering a similar energy use and GHG emissions compared to those of active ingredients for pesticide production (Table 39). ENTEC granules contain 11 g active ingredient (nitrification inhibitor) per kg ASN which will be modelled with respect to the mineral fertiliser amounts applied to the different crops within the model farms.
Since the nitrogen supply of fertilisers with nitrification inhibitors compared to common application schemes is better synchronised with the crop demand, at least one fertiliser application operation (depending on the application scheme) can be saved. Thus, for the modelling of the field operations within the different model farms one fertiliser application will be disregarded compared to the reference scheme of split application if more than one fertiliser application per crop is estimated.
Possible higher yields due to the more efficient use of nitrification inhibitors or a reduced use of N fertiliser matching nitrogen supply for a preestablished target yield will not be considered since reliable data are currently not available.
The effect of the application of fertilisers with nitrification inhibitors on farm level GHG mitigation will be modelled for one dairy (DF1), one bull fattening (BF2) and both pig fattening (PF1, PF1) model farms having a high share of arable land and hence a higher mitigation potential.

Table 39: Primary energy use in MJ kg^{-1} active ingredient and associated GHG emissions in g kg^{-1} active ingredient of the plant protection agent production (according to Kaltschmitt & Reinhardt, 1997).

Primary energy use	263	MJ kg^{-1} active ingredient
CO_2	4921	g kg^{-1} active ingredient
CH_4	0.18	g kg^{-1} active ingredient
N_2O	1.50	g kg^{-1} active ingredient
NH_3	0.16	g kg^{-1} active ingredient
NO_X	6.92	g kg^{-1} active ingredient

3.6.2 Abatement costs

The costs of fertilisers with nitrification inhibitors are higher compared to usual fertilisers. While the reference fertiliser ASN costs under German conditions on average 195 € t^{-1} N (depending on region and season) the ASN fertiliser with nitrification inhibitor costs approximately 10 % more (213 € t^{-1} N).
Due to the fact that less field operations for fertiliser application are needed, lower energy and labour costs will be considered by the modelling.

3.7 Increase of grazing in comparison to animal housing

Traditionally, grazing was an integrated part of dairy production systems, where animals spend the summer months on pastures. Depending on the management system, dairy cows spend the night either in animal housing or on pasture. Currently, many dairy systems keep their livestock in animal houses all year round. A clear European trend to more grazing or more animal housing management is not predictable. On the one hand, the development of new housing systems with more flexible working routines, and the trend towards increasing herd sizes, may restrict the number of cattle on pastures in the future. For the Netherlands, for instance, it is currently estimated that the proportions of cattle on day-and-night grazing, day-only grazing and zero-grazing are 45, 45 and 10 % (Schils et al., 2002). On the other hand, recent studies indicate that the increasing higher requirements for compliance with animal welfare environmental regulations are causing many dairy operators to rethink their animal housing system. Therefore, there are increasing numbers of reports of dairy operators changing from animal housing to grazing-based production systems.

The animal husbandry system and the type of grazing system best suited to a given farm depends on the farmer's goals and resources. Generally, grazing systems can be divided into continuous and rotational stocking. Rotational grazing allows flexibility in management and provides a better opportunity to use livestock to manage pastures' grasses, legumes and weeds. Rotational stocking is also best suited where the manager wants to increase animal production per pastured area or reduce operating costs by harvesting forage with livestock instead of machinery. The number of paddocks used in a rotational stocking system varies with management goals and personal preferences. Intensive systems are best suited for high-producing animals since this management provides a uniform, high quality feed supply. This usually can also result in a more uniform level of milk production.

With respect to GHG emissions, farm emissions are strongly influenced by N fertilisation on arable land and by animal excreta on grazed grasslands (Fowler et al., 1997). Here, especially, the different NH_3 and N_2O emissions of both grazing and animal housing systems have to be considered for modelling on farm level.

On the one hand, pastures with unevenly distributed dung and urine patches as well as more or less compacted soil sites significantly contribute to a spatial and temporal variability of N_2O fluxes from soils. On the other hand, several studies reported that NH_3 emissions per animal are lower for grazing animals than for those in housing where the excreta are collected, stored and applied to land (Pain & Jarvis, 1999). This is mainly caused by the fact that urine excreted during grazing often infiltrates into the soil before substantial NH_3 emissions can occur. Because of these relatively low losses from grazing compared to losses from the housed phase and manure application, Pain & Jarvis (1999) suggested to extend the grazing season so that the amount of excreta produced indoors would be reduced.

In general, grazing animals contribute to slightly more than 10 % of the global N_2O budget (Oenema et al., 1997). Emissions are partly caused by the fact that the distribution of N returns via grazing animals are more heterogeneous than if applied as manure, and more exposed to leaching losses because of extremely high point levels. In this regard, patches are important sites for N loss via NH_3 volatilisation (Jarvis et al., 1989; de Klein & Ledgard, 2001; Ledgard et al., 2001), via nitrate leaching (Ryden et al., 1984; de Klein & Ledgard, 2001) and via denitrification and thus N_2O emissions (Ryden et al., 1986). According to Oenema et al. (1997) grazing animals affect the emission of N_2O by 1) the return of N in urine patches, 2) the return of N in dung patches, and 3) treading and trampling.

Mosier et al. (1998) reported that N_2O emissions from livestock are much higher when animals are in the meadows than when they are in animal housing systems. The FASSET model has been used to estimate the emissions from grassland used for either grazing or cutting. To simplify calculations a pure ryegrass was simulated and fertilised with mineral nitrogen only. The calculations were repeated for 30 different climate-years. The average annual cycle of nitrous oxide emissions and differences between cutting and grazing are shown in Figure 3. Clayton et al. (1994) found that N_2O emissions from a grazed pasture were three times higher than from an ungrazed equivalent. Also Velthof et al. (1998) argued that grazing-derived emissions are sometimes larger than N fertiliser-derived emissions. Therefore, according to these authors N_2O emissions from animal waste management can be reduced by restricting grazing. This will result in a shift from high N_2O emissions during grazing to lower emissions from anaerobic waste management systems. When grazing is restricted, the cattle will be stalled for a longer time and more urine and dung will be collected and stored as slurry. Here, various technical measures are available to control and reduce emissions. The slurry will then be applied as fertiliser to grassland (e.g. by the use of improved application techniques (see 3.5)) and, as a result, less N fertiliser will be required. Consequently, N_2O emissions are larger for dung and urine patches in grassland than for slurry which has been properly applied to soil. Therefore, total leaching-derived and N fertiliser-derived N_2O emissions will also be lower when grazing is restricted. Thus, restricted grazing may be an option to mitigate N_2O emissions from intensively managed grasslands rather than an extension of grazing.

Hence, a final assessment of the mitigation effect of the extension of grazing instead of animal housing is only possible if all direct and indirect GHG emissions and operations are modelled on farm level.

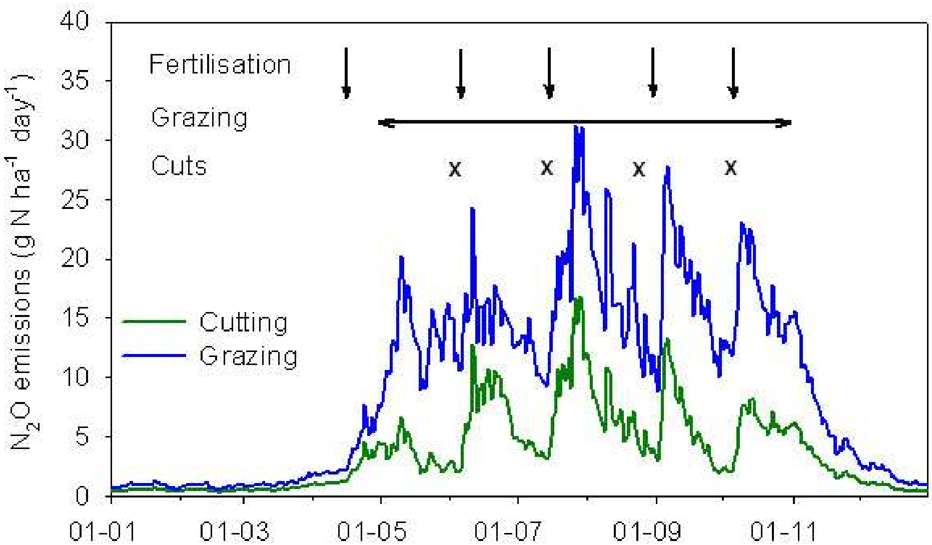


Figure 3: Simulated mean daily nitrous oxide emissions from ryegrass on a loamy sand soil fertilised with 200 kg N ha^{-1} a^{-1} and used for either cutting or grazing. The dates of fertilisation are shown with arrows. The line shows the grazing period, and the crosses show dates of cuts.

3.7.1 Definition of mitigation measure

Instead of a whole day animal housing, dairy cows will be driven between milking events on the pasture for grazing during May to October (153 days). This change in management results in wide modifications of the standard model farms that have to be defined for modelling.

One aspect that may increase the mitigation effect of an increased area for grazing is based on the higher oxidation of atmospheric methane of grassland soil compared to arable land. Boeckx & van Cleemput (2001) summarised available results of European measurements to CH_4 uptake i.e. negative emission factors. Average values are –2.5 kg ha^{-1} a^{-1} for grassland

and –1.5 kg ha^{-1} a^{-1} for arable land. These emission factors are used for the modelling but there will be no difference when increasing the time of grazing as the area for grassland or pastures will basically not change. But if the pasture area increases, these differences in methane oxidation have to be implemented.

In addition, there are several operation differences that have to be taken into account when considering an extension of grazing. If grazing is implemented, less field operations such as grass harvest, silage baling, transport and storage, manure applications as well as operations for the feed supply will take place. Since grazing is defined for the period of May to October, not the whole scale but 80 % of field operations will be reduced as in spring and autumn grass silage will also be harvested. Manure application operations will be reduced according to the reduced amount of manure excreted in animal housing. This reduced number of operations is also connected with savings of diesel and accordingly of CO_2 emissions which will be calculated by ModelFarm. In contrast, there are additional operations for the grazing system like the daily cattle drive as well as the construction and maintenance of fences that cause minor GHG emission changes but increase abatement costs that have to be considered for the calculation of the cost-efficiency of this technical measure (see 3.3.2).

Furthermore, as already mentioned, differences in NH_3 and N_2O emissions have to be modelled for the defined dairy farms. According to EMEP (2003), a dairy cow that excretes 100 kg N per year leads to an emission of 3.21 kg NH_3-N per year. It is assumed that 40 % of 100 kg = 40 kg is excreted, while the cows are on the pasture. This means that a fraction of 3.21/40 = 8.0 % of the amount of N that is excreted on the pasture is volatilised as NH_3. Thus, for the modelling in ModelFarm it is also assumed that 40 % of N is excreted by dairy cows on pastures so that similar to EMEP (2003) an emission factor of 8 % NH_3-N of total excreted N is used.

In particular, emissions of nitrous oxide from excreta have been investigated in a large number of studies. Excreta, in the form of urine patches deposited while animals are grazing, provide high concentrations of readily available N on small areas so that grazed pastures can be significant sources of N_2O production. However, the regulation of N_2O emissions from urine-affected soil is not well understood. According to Oenema et al. (1998), emissions from grazed grasslands typically range from 10 to 20 kg N_2O-N ha^{-1} a^{-1}. Galbally (1992) estimated the annual emission of N_2O from improved pastures to be 4 kg N ha^{-1} a^{-1}. These results confirm that the variability of emission factors observed is extremely high. Therefore, IPCC (1996) and Mosier et al. (1998) have proposed emission factors which distinguish between excreta that are collected and stored as liquid or solid manure on the farm and are later applied to arable crops (EF = 0.0125) and excreta that are deposited during grazing (EF = 0.02). These emission factors will be used for modelling. In addition to the direct N_2O emission factor, the indirect N_2O emission factor is 1 % for N lost by ammonia volatilisation and 2.5 % for N lost by nitrate leaching (IPCC, 1996).

In general, the emission reduction that probably can be achieved by increasing the proportion of the year spent grazing will depend on the reference system, the time animals graze, the fertiliser level of the pasture etc. Hence, the comparison of grazing instead of animal housing will be modelled with the standard reference farms and additionally with reference farms where technical GHG mitigation measures such as a higher manure removal frequency from animal housing into a covered outdoor manure storage and an improved application techniques is already implemented.

3.7.2 Abatement costs

According to some studies (e.g. Waßmuth, 2002), grazing systems (especially intensive rotational grazing) accrue less revenues than animals kept in houses.

If a half-day grazing is implemented, machinery, energy and labour costs for less field operations such as grass harvest, silage baling, transport and storage, manure application as well as operations for feed supply are incurred. Instead, higher labour costs have to be considered for the daily cattle drive, more time spent on pastures checking and moving livestock, and the additional effort required to make water available to the animals. The additional labour costs will be calculated according to KTBL (2004) with respect to the different herd sizes of the model farms (Table 40). For the pastures also more fences have to be built and maintained. According to KTBL (2004) the additional fence costs were calculated for 3 ha pastures to be 76.5 € $ha^{-1}a^{-1}$. Additional labour costs for fence maintenance will be modelled according to Table 41.

Table 40: Additional labour minutes for half day pasture (according to KTBL, 2004).

Animals per process step			
10	20	40	80
[labour minutes cow^{-1} d^{-1}]			
4.4	3.6	1.25	0.75

Table 41: Additional labour hours for fence maintenance (according to KTBL, 2004).

Animal number per farm			
10	20	40	80
[labour hours ha^{-1} a^{-1}]			
6.6	4.6	3.7	3.6

3.8 Anaerobic digestion

Anaerobic digestion is the bacterial fermentation of organic material under controlled conditions in a closed vessel. The fermentation process produces biogas which typically consists of up to 65 % methane (50-65 %) and about 35-45 % CO_2. The rate of biogas and methane generation is dependent on the rate of anaerobic digestion. Environmental factors affecting the rate of anaerobic digestion include temperature, pH, carbon to nitrogen and water to solid ratios, nutrient composition particle size, retention time and quality of manure and/or co-digestible material.

The produced methane from anaerobic digestion can be recovered and used as energy by adapting manure management and treatment practices to facilitate methane collection. Anaerobic digestion plants can be small scale, located on a farm, or large centralised plants can be used (Meeks & Bates, 1999). Especially in the case of the latter, other organic wastes or energy plants may also be taken as substrates to increase methane yields and to ensure a consistent supply of organic substrates all year round. Both farm-scale and centralised plants can be used to produce vehicle fuel, heat and/or electricity, which operators of biogas plants may utilise, or especially in the case of centralised plants may be sold (Weiske et al., 2006). The biogas plants also produce a digestate, which potentially can be sold as a soil conditioner. Hence, the biogas production from animal manure, residues and on-farm produced or imported energy crops is a very promising option to generate renewable energy and simultaneously to reduce GHG emissions directly (manure management) or in the way that the energy produced can offset CO_2 emissions from fossil fuels.

Biogas production is typically carried out in wet fermentation processes (substrates are suspended so that they can pass a pump). In addition, dry fermentation plants are getting more popular but are still prototypes. Thus, model calculation will only be carried out for the more common biogas production with manure as basic substrate.

In digestion plants it is possible to use only manure as organic substrate whereas co-digestion is the simultaneous digestion of a homogenous mixture of two or more substrates such as residues from animal husbandry and plant production, directly produced energy plants or imported residues from the food industry. On-farm produced or imported energy plants can be fed directly, or after ensilage, as a co-substrate to digesters. The co-digestion of plant material and solid waste can provide an improved nutrient balance and therefore better digester performance and higher biogas yields and therewith may result in a higher reduction of GHG emissions by substitution of fossil fuels. The additional biogas collection can also bring farmers a higher income due to the increased methane yields and the improved efficiency of a biogas plant.

In Germany, 95 % of modern biogas plants use co-substrates (Weiland et al., 2004). Cattle slurry is the basic substrate for $^2/_3$ of the biogas plants and pig slurry for 15 %. The remaining farms use poultry slurry or manure mixtures (BMVEL, 2002; Weiland et al., 2004). About 70 % of biogas plants use maize silage and approximately 50 % grass silage as co-substrate. Thus, the model calculations will focus on farms with dairy and beef cattle production and biogas plants that use maize and grass silage as the main co-substrate.

With respect to the GHG mitigation potential it is assumed that a reduction of CH_4 emissions of 50 % is achievable for both farm scale and centralised plants in cool climates, for manures that would otherwise be stored as liquid slurry, and hence have relatively high methane emissions. For warmer climates, where the methane emissions from such manure storage systems are estimated to be more than three times higher (IPCC, 1997), a reduction potential of 75 % is assumed. Anaerobic digestion may also prevent N_2O and NH_3 emissions into the atmosphere if an appropriate application technique is used.

Thus, apart from the substitution of fossil fuels by the produced biogas a central aspect for the evaluation of the implementation of biogas production is that anaerobic digestion of organic material mainly influences the emissions of manures (digested compared to non-digested) during storage and after manure application. Here, the CH_4, N_2O and NH_3 mitigation potential essentially depends on the amount and composition of the input material and an adequate comparison to an appropriate reference farming system.

Methane and ammonia emissions dominate the GHG emissions during manure storage and originate from the slurry itself. Factors influencing the emissions are the physical and chemical properties of the slurry such as the content of easily degradable carbon, NH_4^+ content, redox conditions, dry matter content and viscosity, but also pH. Ammonia is mainly influenced and emitted due to the pH controlled equilibrium of NH_4^+ and NH_3. Methane is formed by methanogenic bacteria in the slurry during storage and is also influenced by different environmental factors as described above. In contrast, N_2O is mainly formed by nitrification when the slurry surface dries up during storage. Emissions are further affected by environmental conditions e.g. wind speed, temperature and the degree of slurry exposure to the atmosphere.

However, during the process of fermentation, substrate parameters such as DM, ODM and the NH_3/NH_4^+ ratio undergo changes that may affect the potential to emit GHG (Table 42; Clemens et al., 2004). Moreover, the manure is stored in hermetically sealed fermenters which prevent nearly all emissions and should at least be stored after the fermentation process in covered storages if it is not used as additional digester which also reduces GHG emissions considerably.

Table 42: Properties of digested and undigested cattle manure and mixtures of cattle manure (according to Clemens et al., 2004).

Parameter		**Cattle manure undigested**	**Cattle manure digested 29 d**	**Cattle manure digested 50 d**
DM	**[g kg^{-1}]**	30.4	22.9	23.1
ODM	**[g kg^{-1}]**	22.0	14.5	14.3
COD	**[g kg^{-1}]**	37.3	21.7	19.7
N-Kj	**[g kg^{-1}]**	1.99	2.06	2.28
NH_4-N	**[g kg^{-1}]**	1.04	1.41	1.51

Recent study results show that anaerobic digestion seems to be an effective mitigation option for methane and greenhouse gas emissions from slurry stores. Schumacher (1999) and Wulf et al. (2003) show that the mitigation effect for cattle slurry is substantially higher compared to pig slurry (Table 43, Figure 4). The study results of Wulf et al. (2003) show that anaerobic digestion reduces CH_4 emissions but enhances NH_3 emissions (Figure 4). Straw cover reduces NH_3 emissions but enhances (in particular for digested pig slurry) CH_4 emissions (Schumacher, 1999; Wulf et al., 2003). Amon et al. (2004) show that the production of NH_3 and N_2O but in particular of CH_4 is strongly related to slurry temperature and thus, under warm summer conditions, considerably more greenhouse gases were emitted than under cool winter conditions (Table 44).

Table 43: GHG emissions from slurry according to Schumacher (1999; in Clemens et al., 2000).

	CO_2-equivalents [g m^{-2} h^{-1}]			
	Slurry	**Slurry / Straw**	**Digested**	**Digested / Straw**
Pig slurry	1.32	2.08	1.18	3.49
Cattle slurry	2.10	1.77	0.43	0.58

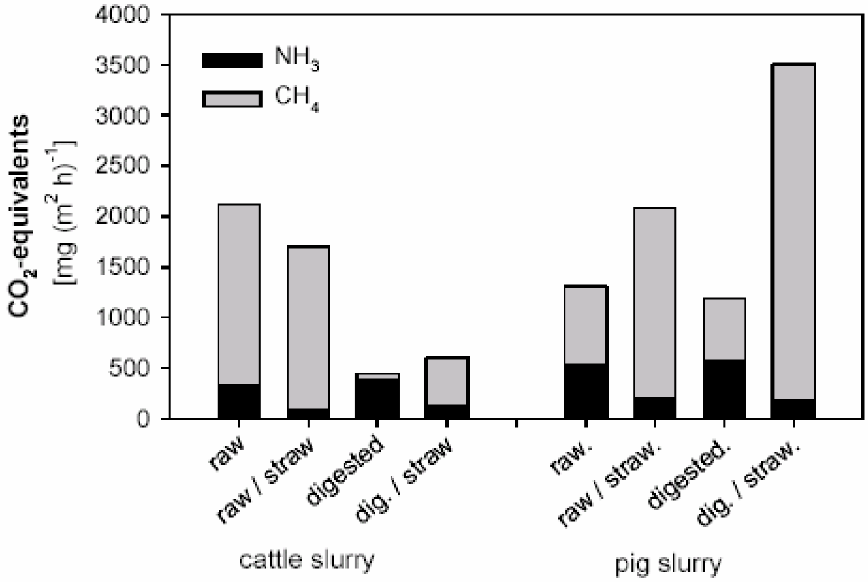


Figure 4: Emissions of NH_3 and CH_4 during storage of raw and digested slurries (calculated in CO_2-equivalents; according to Wulf et al., 2003)

Table 44: Cumulated CH_4, NH_3, N_2O and greenhouse gas emissions measured in the winter and in a summer experiment (Amon et al., 2004).

Treatment	winter experiment				summer experiment			
	CH_4 [g m^{-3}]	NH_3 [g m^{-3}]	N_2O [g m^{-3}]	GHG [kg CO_2eq. m^{-3}]	CH_4 [g m^{-3}]	NH_3 [g m^{-3}]	N_2O [g m^{-3}]	GHG [kg CO_2eq. m^{-3}]
Untreated slurry with natural surface crust	164.3	72.5	44.0	17.10	3591.2	110.5	48.7	90.52
Untreated slurry with natural surface crust and wooden cover	142.0	52.2	38.2	14.83	2999.0	60.0	58.6	81.13
Anaerobically digested slurry without any cover	111.3	62.0	40.1	14.76	1154.2	222.5	72.4	46.70
Anaerobically digested slurry with a layer of chopped straw	114.5	49.6	39.9	14.79	1191.9	125.7	75.7	48.51
Anaerobically digested slurry with a layer of chopped straw and a wooden cover	81.1	48.7	40.7	14.31	1021.4	78.1	61.4	40.50

To conserve the GHG mitigation by anaerobic digestion for the whole production chain an improved manure application technique is needed, otherwise much of the benefit of abating during manure storage in the digesters may be lost.

Fermented substrates differ from slurry in some of their chemical and physical parameters (see also Table 42) that might also influence GHG emissions after application. Nitrogen from this organic pool is transferred to inorganic nitrogen during this process so that the share of NH_4^+-N from nitrogen increases (Wulf et al., 2002a). Due to the higher NH_4^+-N content of fermented slurry the likelihood is given that NH_3 emissions increase after application compared to untreated slurry. In addition, constituents that can be oxidised by chemical or biological processes as well as dry matter content are reduced. Thus, due to the fermentation the consistency of the manure is changing (it turns into a thin fluid) so that the rate of slurry infiltration into soil can increase.

Digested slurry also has a higher pH which may additionally increase the risk for NH_3 emissions. Therefore, the digestate must be applied with improved application techniques, otherwise NH_3 emissions are likely to increase even if digested slurry infiltrates more rapidly into the soil.

Rubaek et al. (1996) reported similar or even lower NH_3 emissions loss from agricultural systems from fermented substrates compared to untreated slurry, whereas Kuhn (1998) postulates an increase of NH_3 emissions through slurry fermentation. Petersen (1999) showed in field experiments that anaerobically digested slurry induced lower N_2O emissions compared to undigested slurry. Clemens & Huschka (2001) reported the same from lab experiments.

Wulf et al. (2002b) showed that the influence of (co-)fermentation on N_2O and CH_4 emissions was only small and of short duration, whereas the application technique had a much stronger effect (Figure 5). In total, their measurements showed that GHG emissions after field application from anaerobically treated substrates are similar to those from untreated slurry (Figure 5). The experiments showed that indirect N_2O production from emitted NH_3 might contribute a great proportion to GHG emissions from organic fertilisation. Therefore, NH_3 measurements should in future always be included in experiments designed to evaluate emissions of greenhouse gases. For spreading co-fermented slurry on grassland, trail shoe application seemed to be the best way of minimising trace gas emissions (Wulf et al., 2002b). On arable land, trail hose application with immediate harrowing seems to be recommendable, as in addition to the mentioned sources of greenhouse gases, injection of slurry causes higher fuel consumption with negative effects on GHG budgets. In total, GHG emissions after field application from anaerobically treated substrates are similar to those from untreated slurry (Figure 5). Also Clemens et al. (2004) reported that fermentation of the slurry did not affect overall GHG emissions after application. Thus, these emissions need not to be included into the calculation if an improved application technique is used for manure application.

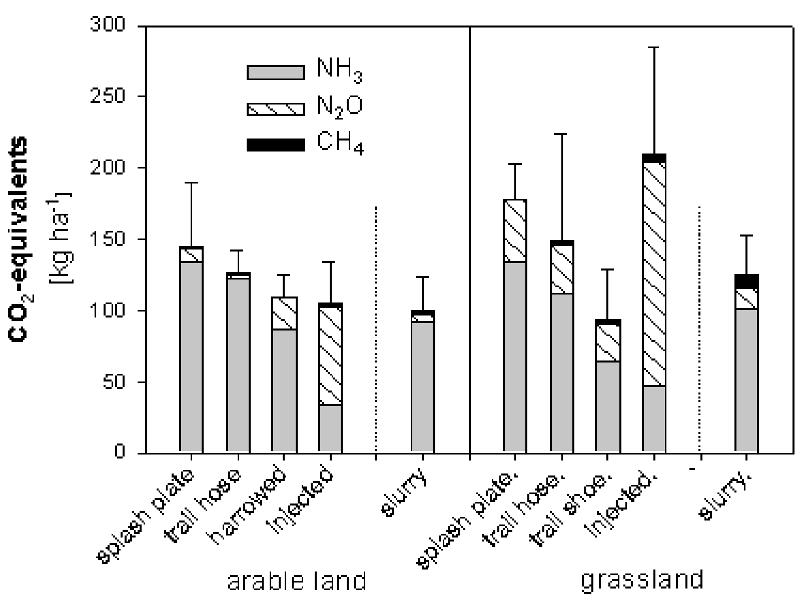


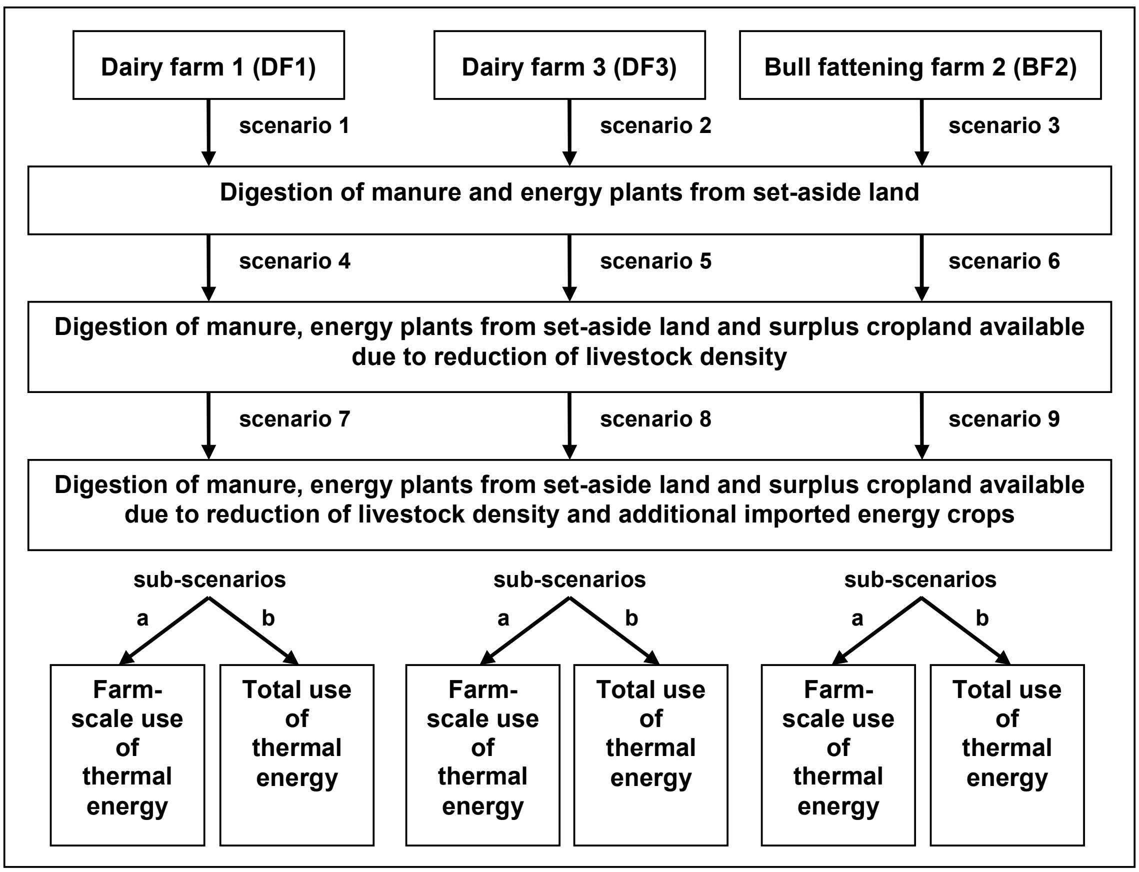
Figure 5: Global warming potential of different application techniques for co-fermented slurry and trail hose applied unfermented slurry expressed as CO_2-equivalents calculated from the emission of NH_3, N_2O and CH_4 (Wulf et al., 2001, Wulf et al., 2002b).

3.8.1 Definition of mitigation measure

For the biogas production as technical and management-based mitigation option, different scenarios will be calculated for two dairy farms (DF1, DF3) and one farm with bull fattening (BF2; Figure 6). Due to the fact that the standard farms of the cluster analysis are too small to operate anaerobic digestion cost-efficiently, it is assumed that a multiple of the defined farms conduct one collective biogas plant. The adequate number of farms thereby depends on the amounts of organic material that is available for anaerobic digestion. Thus, the input amounts for the biogas plant will be determined and described in detail when the definition of the model farms is finalised.

For the modelling of scenarios 1-3 (Figure 6), only the manure collected from the animal stores plus additional energy plants (maize and grass silage) grown on set-aside land are calculated as input material for the digestion process. Furthermore, in scenarios 4-6 the livestock density of the farms will be reduced by 20 % to use the surplus cropland and grassland for additional substrate production. For the calculation of the scenarios 7-9 with an additional import of co-substrates it must be considered that the produced digestate amounts do not exceed the Nitrates Directive (Dir. 91/676/EEC) limits of 170 kg N per hectare arable land and 210 kg N per hectare grassland applied each year.

Figure 6: Biogas production scenarios for dairy and bull fattening farms.



The import of substrates as co-digestates reflects the possibility that crops are directly produced as energy crops for anaerobic digestion. Therefore, the additional production of substrates that are imported into the model farm represents a direct extension of the farm, so that the GHG emissions associated with the co-substrate production have to be included in the model calculations. The energy use and GHG emissions of the imported energy plants will be calculated similar to the on-farm produced amounts by ModelFarm whereas the use of fossil

fuels and the associated GHG emissions for transportation of imported substrates have to be considered separately.
According to the different combinations of the digested substrates, the duration of the fermentation was estimated to be 30 days with manure only and 35 days with additional input of organic matter for co-digestion. The estimated duration of the fermentation process and changing input amounts of organic material for anaerobic digestion give rise to different sizes of digestion plants that will be calculated by the model under mesophilic digestion conditions. The necessary adjustments affect the volume of the digester and manure storage as well as the size of the combined heat and power unit (CHP). The calculation of the produced biogas will be carried out by ModelFarm by the use of the methane production factors (B_o-factors) of different organic substrates summarised in Table 45. The size of the pilot injection gas engine as combustion engine will also be modelled by ModelFarm depending on the biogas yield by anaerobic digestion. For the combustion of the produced methane the use of additional fossil fuel amounts needed for the process of the pilot injection gas engines will be calculated by the model too. For the biogas production, a share of 10 % from post digestion is estimated. The resulting efficiencies of the needed fossil fuel for the biogas combustion and the produced amounts of electric power and thermal energy will also be calculated by ModelFarm.

Table 45: Methane production factors (B_o-factors) for manure and organic matter for methane production during anaerobic digestion.

Substrate	B_o [m^3 CH_4 kg^{-1} VS]
Manure from heifers (6-25 months)	0.182
Manure from cows and bulls	0.200
Clover-grass silage / red clover	0.260
Maize silage	0.340
Whole crop cereal silage	0.350

In general, the efficiency of biogas production as GHG mitigation measure does not only depend on the amount and quality of organic matter used for co-digestion, but is also considerably dependent on how much of the thermal energy produced is exploited. Not all farms are able to use all produced heat for adjacent houses, hotels, manufactories etc. The usage also depends on climatic (the produced thermal energy cannot be used for the whole year in all cases) and other conditions. Furthermore, plants to use produced heat for cooling are still in the development stage and are presently connected with high costs. However, for each scenario two sub-scenarios (a + b, Figure 6) will be calculated to show the efficiency on the one hand, for the case in which the thermal energy produced is only used to heat the digester (which depends on the input amounts, fermentation process etc.) and farm houses (which is equivalent to an average heat use of 4000 l fossil fuel a^{-1}) and, on the other hand, if all the thermal energy produced is utilised on the farm and by external users.
Since for the scenarios with biogas production the manure storage subsequent to the main digester is also defined to be operated as closed post digester, most of the typically emitted greenhouse gases of an open storage are eliminated. The hermetically sealed storage of manure prevents all N_2O and NH_3 emissions and reduces CH_4 emissions significantly. Methane losses are only caused by the permeation of the cover plastic foil and leakage of pipe connections and valves. These CH_4 emissions are calculated according to ELTRA (2003) and Olesen et al. (2004a) who estimated that 1.8 % of the produced CH_4 is lost in the biogas production (Table 46). Furthermore, the loss of CH_4 during the combustion process is defined to be 0.06 g CH_4 per kg biogas if a pilot injection gas engine is used (Edelmann et al., 2001).
ModelFarm will also be used to estimate the effect of anaerobic digestion on the substitution of mineral fertiliser. The application amounts of mineral fertiliser have to be adapted because of the increased fraction of plant available N (more NH_4^+-N) within the digestate and

additionally as a result of the increased digestate amount due to the additional on-farm produced and imported organic material. The substitution of mineral fertiliser by manure reduces the energy use and GHG emissions of mineral fertiliser production and partially of fertiliser application but increases the emissions from manure transports and application.
The NH_3 emissions will be calculated by equation (15) according to the ALFAM (2002) project for trail hose as standard application technique (see chapter 3.5.1) considering the higher NH_4^+-N content of the digested manure. The direct effect of digested compared to non-digested manure composition on nitrous oxide emissions after application will be neglected. The N_2O emissions will be calculated by the standard calculation (1.25 ± 1 %; IPCC, 2000) related to total applied N and considering the changes due to different substrate composition and digestate amounts. CH_4 emissions after manure application will be neglected (see chapter 3.5.1).

Table 46: N_2O, CH_4 and NH_3 emissions of non-digested manure in open storage tanks in comparison to the closed storage of digested manure and organic matter.

	Non-digested (open storage)	Digested (closed storage)
N_2O	1 % of total N (IPCC, 2000)	0
CH_4	39 % of maximum CH_4 production (IPCC, 2000)	1.8 % of total produced CH_4 (ELTRA, 2003; Olesen et al., 2004a), 0.06 g CH_4 per kg biogas if a pilot injection gas engine is used (Edelmann et al., 2001)
NH_3	2.4 % of total N (Olesen et al., 2004a)	0

3.8.2 Abatement costs

In general, the cost-effectiveness of anaerobic digestion and also the cost-efficiency of biogas production as mitigation measure depends on the efficiency of the digestion process itself, on the achievable benefit from the produced and utilised electricity and thermal energy and finally, directly and indirectly on subsidies that can contribute to the profitability of the biogas production.
However, the overall economics of energy crops co-digestion depends crucially on crop yield, raw material production costs, biogas yields and on the energy utilisation degree (Weiske et al., 2006). The costs for the substrate production and digestate application by trail hose (see Table 37 in chapter 3.5.2) as well as for the construction and operation of the different biogas plants (digester, CHP etc.) will be calculated on a yearly basis by ModelFarm. The costs of the imported co-substrates were calculated provided that these silages are produced as energy plants on adjacent farms (Table 47). The costs for transportation of the substrates will also be calculated separately.

Table 47: Full costs of the imported substrates maize, grass and whole crop silage.

Substrate	**Substrate costs [€ t^{-1} FM]**
Maize silage	29.4 (28 % DM)
Grass silage	34.6
Whole crop silage	27.1

Furthermore, for the calculation of the cost-efficiency of biogas production it must be considered that in different European countries operators of biogas plants receive support in the form of capital grants, low costs loans and tax incentives. For example in Germany, Austria, Spain and Italy, premium electricity prices are available. Thus, as the cost-efficiency of the biogas production is also substantially dependent on subsidies, the modelling will be carried out for both cases 1) without any subsidies to calculate the cost-efficiency of biogas production as mitigation measure and 2) with subsidies according to German conditions to evaluate the cost-effectiveness of biogas production for operators. In addition, the sub-scenarios a + b (Figure 6) consider either the farm-scale use of thermal energy of 40 MWh a^{-1} (which is equivalent to an average heat use of 4000 l fossil fuel a^{-1}) or that total produced thermal energy is utilised. To calculate the absolute costs of GHG mitigation by biogas production without any subsidies, the commonly realised energy prices were estimated to be 6 €ct per kWh electricity and 3 €ct per kWh thermal energy. A share of on average 8 % of the produced electricity will be used for the operation needs of the biogas plant and will be calculated dependent on the input material and amount and therewith the size and efficiency of the digestion plant. An additional share of 263 kWh $cattle^{-1}$ a^{-1} (Clausen, 2000) will also be considered according to the stocking rate of the defined model farms. For the case in which a payment of approximately 15-17 €ct kWh_{el}^{-1} for the planned mode and dimension of biogas digestion plants is guaranteed, the operator of the biogas plant will not use their own produced electricity. In this case the farmer will sell all utilised electric power and buy the electricity for a common tariff of 14 €ct kWh_{el}^{-1}.

3.9 Organic farming

In recent years, it has been the aim of many studies to evaluate organic and conventional agricultural production systems with respect to energy and nutrient efficiency in animal and plant production and the farm greenhouse gas emissions (e.g. Dalgaard et al., 1998; Cederberg & Mattsson, 2000; FAL, 2000; Olesen et al., 2006). Until now, a concluding assessment of the environmental impact of conventional and organic practical farms, however, particularly with regard to GHG emissions does not exist as research results are mutually contradictory on the whole. This is mainly caused by differing system boundaries (e.g. without prechain emissions) and by the fact that some of the studies have not always considered all relevant flows of the calculated farms. Therefore, a combined assessment of all key issues of efficiency and sustainability is essential in order to determine the contribution of organic and conventional production systems to sustainable nutrient cycles and GHG emissions.

In general, extensive agricultural systems are characterised by low productivity per unit area, low inputs of energy or technology, but high labour input, and are frequently used for grazing and shifting cultivation. Conversely, intensive agricultural systems are characterised by high productivity per unit area, high inputs of energy or technology but a lower labour input.

This qualitative comparison indicates that the evaluation of organic and conventional production systems depends on whether GHG emissions are calculated per unit area or per product unit (e.g. kg milk, meat, wheat).

A study by FAL (2000) showed that the primary energy input of conventional crop production is clearly higher than in organic farming both on the land area and the product basis, despite lower crop yields in organic production systems. The primary energy input in the conventional compared to the organic animal production systems was 53 % higher per kg of pig meat produced and 85 % higher per kg of milk. Hence, the farm input related to CO_2, CH_4 and N_2O emissions were generally higher both per hectare farmland and per kg product in conventional farming compared with organic production.

The results can, to a great extent, be explained by differences in feeding strategies and the use of synthetic fertilisers in conventional production systems (Refsgaard et al., 1998; Cederberg & Mattsson, 2000). This is confirmed by a life cycle assessment by Cederberg & Mattsson (2000) for Swedish organic and conventional dairy farming systems. Due to the feeding strategy with a larger share of roughage fodder it was estimated that methane emissions are 10-15 % higher from cows in organic farming compared with conventional production (Figure 7). According to Cederberg & Mattsson (2000) the potential negative effect on farm level on global warming is more apparent in a conventional system due to the larger part of CO_2 emissions from the use of fossil fuels and the N_2O emissions caused by higher fertiliser rates.

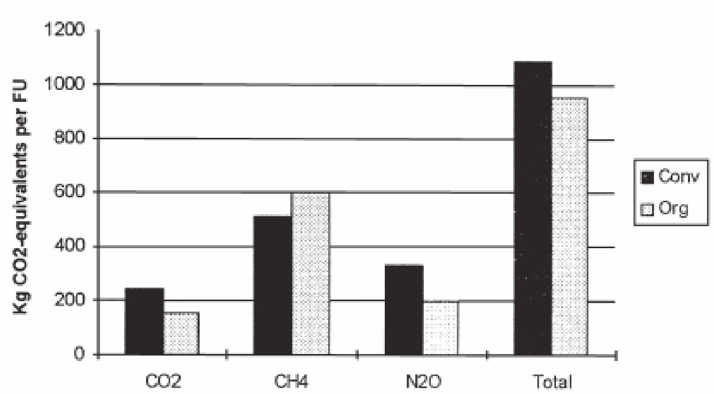


Figure 7: The potential contribution of Swedish organic and conventional dairy farms to global warming (Cederberg & Mattsson, 2000).

These results confirm that the efficiency of nutrient cycles, and especially with respect to nitrogen, give an indirect indication of the GHG emission balance of a farm but is not sufficient enough for a life cycle assessment alone. For a satisfactory life cycle assessment all input and output parameters and nutrient flows have to be included.

Model calculations by Olesen et al. (2006) also showed that in the case of the inclusion of all relevant C and N flows and GHG emissions at the farm level (including indirect N_2O emissions associated with N losses) and all pre-chain emissions from imports of products in the model, the simulated total GHG emissions of organic and conventional dairy farming systems could be closely related to either the farm N surplus (as the difference between imported and exported N) or the farm N efficiency. The N surplus calculated for seven conventional and eight organic model farms in EU15 was found to increase with increasing livestock density on both conventional and organic dairy production systems (Figure 8a; $R^2 = 0.87$). The simulated N efficiencies, calculated as the ratio of exported over imported N, generally varied between 16 and 26 %, and there was a tendency for a decline with increasing livestock density (Figure 8b; $R^2 = 0.17$).

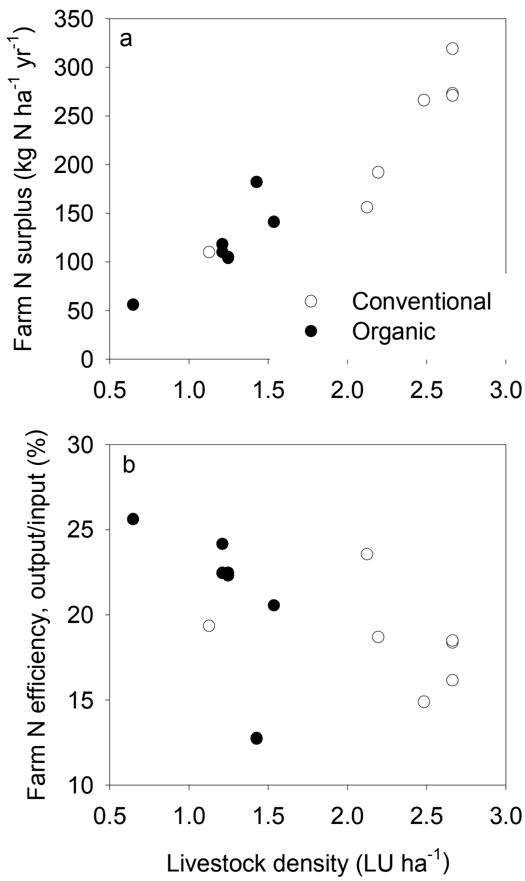


Figure 8: Farm N surplus calculated as imported minus exported N (a) and N use efficiency calculated as exported over imported N (b) depending on livestock density (Olesen et al., 2006).

The GHG emissions increased with increasing N surplus, and on an area basis there was no difference in the linear relationship between conventional and organic farms (Figure 9a). The slope of the regression line indicates an increase in GHG emissions of 0.76 Mg CO_2-eq. kg^{-1} N in N surplus ($R^2 = 0.96$). In addition, there was a tendency for higher emissions per unit farm area from conventional compared with organic farms for similar farm N efficiencies

(Figure 9b). When calculating the emissions on the basis of milk production, organic farms tended to have higher emissions than conventional farms at similar farm N surplus and N efficiency (Figure 9c, d). The farms also exported meat and some plant products (e.g. cereals and potatoes), and Figure 9e and f shows the GHG emissions per unit of energy in the exported product. Here, there was a linear decline in emissions per energy unit with increasing farm N efficiency (Figure 9f; $R^2 = 0.88$), and the slope corresponded to –16.5 kg CO_2-eq. MJ^{-1} per % increase in efficiency.

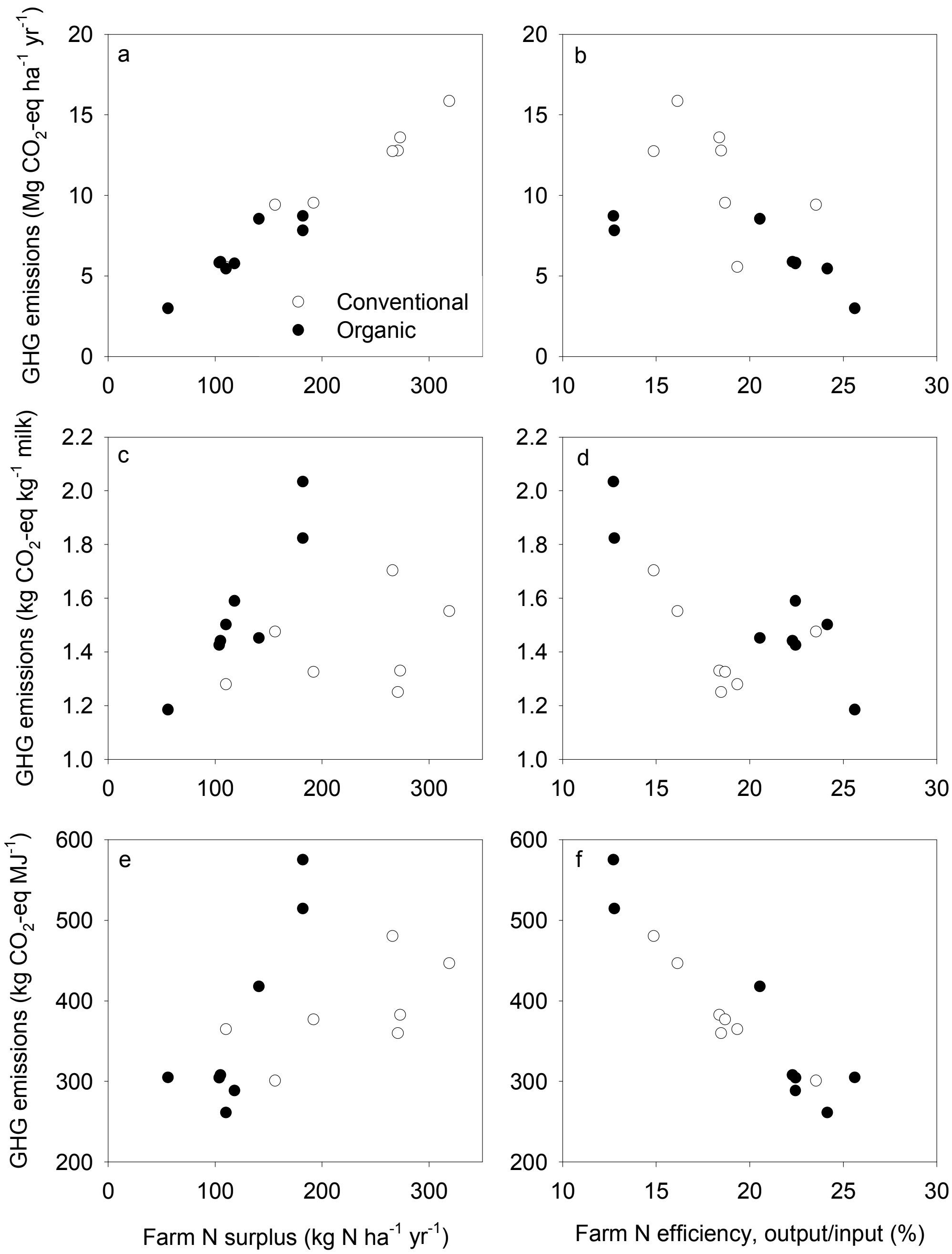


Figure 9: Farm greenhouse gas emissions depending on farm N surplus (a, c and e) and on farm N efficiency (b, d and f). The emissions are shown as emissions per farmed area (a and b), emissions per kg milk produced (c and d), and emissions per MJ of metabolic energy in the exported milk, meat and plant products (Olesen et al., 2006).

A finding of these model calculations is that farm-level GHG emissions are clearly related to the N surplus of the production, irrespective of whether farm management is organic or conventional. The N surplus reflects the livestock density and thus the intensity of the production system. This could be a reason for extensification as an effective management-based GHG mitigation measure to reduce the emissions per area, which would also be in compliance with the recent CAP reforms that seek to de-couple subsidies and production volume. However, extensification implies a reduction in productivity and a general extensification of the European agricultural land would therefore reduce the agricultural production (Neufeldt et al., 2006). For the European society a better indicator is therefore the emissions per unit of product. Here, the results indicate that farm N use efficiency is a good indicator of GHG emissions per unit milk or meat production. The relationship was strongly significant for both organic and conventional farms, but with slightly higher GHG emissions from organic farms at similar farm N efficiency. This can also be explained by the higher estimated CH_4 emissions from enteric fermentation due to a higher proportion of forage crops in the diet. Hence, it was the conclusion of these model calculations that organic dairy farms might not have an advantage from a GHG emissions perspective at all.

3.9.1 Definition of mitigation measure

The conversion to organic farming offers potential to reduce GHG emissions on farm level since less input is used per hectare than in conventional farming. In contrast, European mean yields are 20-45 % lower on organic farms than on conventional farms primarily due to the reduced levels of plant available nutrients so that losses per product unit from plant and subsequently from animal production can be higher due to the lower N use efficiency (Olesen et al., 2006). Thus, the impact of the conversion from conventional to organic farming as management-based mitigation measure will be calculated per unit area as well as per unit product.

The implication of a full conversion to organic production will be carried out for two dairy farms (DF1 and DF2) as a conversion of a milk producing farm appears to be more realistic and relevant than for a pig or bull fattening farm.

For a direct comparison of the different production systems, the size of the conventional and organic farms and the ratio of grassland to arable land will be kept constant. Set-aside land of conventional farms with non-food rape seed production will be taken in for food production of organic farms. In contrast to the conventional model farms, the organic production systems will be defined to be 100 % self-sufficient with respect to feed, meaning that the stocking rate depends solely on the feed that can be produced off the farm land. According to the cluster analysis made by FAL the organic model farms will be defined with respect to representative crop and milk yields. The organic farms are assumed to be operated according to Council Regulations No 2091/91 and No 1804/1999 having an adapted crop rotation with a higher share of catch crops and legumes as N fixing crops. Field operations will also be adjusted according to the conventions of organic production practice. The grass and crop yields will affect the milk yield of the cows and thereby the livestock density of the model farms. The adapted crop rotations and feed plans will be determined and described in detail when the definition of the model farms is finalised.

3.9.2 Abatement costs

In accordance with the changes of the defined dairy model farms as a result of the conversion to organic farming, the production costs as well as the revenues of the agricultural products have to be adapted. The changed production costs mainly concern the costs of field operations and the corresponding labour costs due to the adapted crop rotations and management system. These costs will be calculated by ModelFarm based on data by KTBL (2002, 2004) and Mittelfränkische Landwirtschaftsverwaltung (2004). The revenues for the relevant cash crops

(if available), milk and meat from organic production are based on Bioland (2005) and ZMP (2005) and are listed in Table 48.
Changes with respect to CAP instruments (set-aside, quota regimes, support payments) will be discussed within WP6 and 7.

Table 48: Revenues for different cash crops and product from animal production of organic production systems (Bioland, 2005; ZMP, 2005).

Product	Revenues
Wheat	320 € t^{-1} FM
Spelt	480 € t^{-1} FM
Triticale	180 € t^{-1} FM
Peas	220 € t^{-1} FM
Potatoes	200 € t^{-1} FM
Milk	0.327 € kg^{-1} milk
Meat (cattle)	1.90 € kg^{-1} meat
Meat (heifer)	4.52 € kg^{-1} meat

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