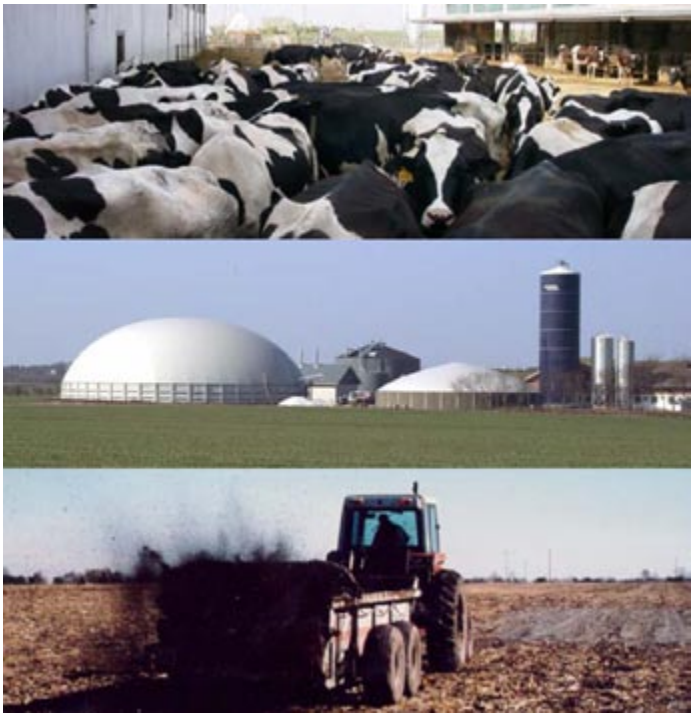
DIAS report

12th Ramiran International conference

Technology for Recycling of Manure and Organic Residues in a Whole-Farm Perspective. Vol. I



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12th Ramiran International conference

Technology for Recycling of Manure and Organic Residues in a Whole-Farm Perspective. Vol. I

Edited by Søren O. Petersen

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About RAMIRAN

The Network on Recycling of Agricultural, Municipal and Industrial Residues in Agriculture (RAMIRAN) is part of ESCORENA - the European System of Cooperative Research Networks in Agriculture. ESCORENA was established by the FAO Regional Office for Europe (REU) in 1974. It is a form of voluntary research cooperation among interested national institutions involved in research in food or agriculture in European countries. Over the years, ESCORENA has expanded its field of activities to include topics and themes of interest to other countries, particularly those from the Near East and Mediterranean area.

The objectives of ESCORENA are to:

- Promote the voluntary exchange of information and experimental data on selected topics.
- Support joint applied research on selected subjects of common interest according to an accepted methodology and an agreed division of tasks and timetable.
- Facilitate voluntary exchange of experts, germplasm and technologies.
- Establish close links between European researchers and institutions working on the same subject to stimulate interaction.
- Accelerate the transfer of European technology advances to, and in cooperation with, developing countries.

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Much of the detailed work of the network is undertaken by the Working Groups. There are currently 7 Working Groups within RAMIRAN including 2 new groups that were established at the last Workshop in Gargnano.

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Preface

Efficient use of agricultural residues and imported waste materials within agriculture is increasingly viewed from a whole-farm perspective. A wide range of management decisions - including feeding, manure collection systems, and treatment for hygienization or energy production - influence the nutrient value and environmental impact of agricultural residues. Field application of manure and urban wastes are affected by societal constraints, such as legislation, tradition, consumer attitudes towards waste recycling, and pollution risks. Hence, the optimal use of manure and organic wastes as a nutrient source and soil conditioner interacts strongly with many other aspects of farming.

The objective behind this 12th International Conference of the Ramiran network is to present and discuss on-farm interactions between manure and waste management practices, and to consider methods to describe and quantify the overall effects of a given strategy or treatment practice. Accordingly, the research presented at the conference and in the proceedings cover a wide range of topics, from feed impact on manure composition to environmental losses in the field, from energy production to odour control, from biochemistry to modelling. We hope that everyone involved in the conference will see this as an opportunity to discover interfaces with other research areas that can strengthen the whole-farm perspective of future research.

The Proceedings of the conference have been compiled in two volumes: Volume I including all oral presentations, as well as an introductory paper prepared by the Scientific Committee, and Volume II including all poster presentations of the conference. We believe that these reports represent an important source of information on the current state-of-the-art with respect to manure and waste management. The papers are brief, but we encourage interested readers to contact the authors for further information.

The Organizing Committee,
Sven G. Sommer, Peter Sørensen, Hanne D. Poulsen and
Søren O. Petersen
August 2006

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Recycling of manure and organic wastes - a whole-farm perspective

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Abstract
As worldwide agricultural production increases, it tends to become concentrated on increasingly larger units. Livestock produce large volumes of manure which, like imported waste materials and crop residues, are a source of valuable plant nutrients and renewable energy, but also a potential threat to the environment and human health. This article discusses briefly the need to assess recycling of organic wastes and manure using a whole-farm approach to avoid a situation where the introduction of new technology and management to regulate one source of pollution will aggravate other environmental impacts downstream in the manure management chain on farms. Some examples of manure N and C turnover are discussed as examples of on-farm interactions.

Introduction

Worldwide agricultural production is increasing dramatically, and it tends to become concentrated on larger production units in order to increase the profitability of the enterprise. Agriculture manages large volumes of animal manure, as well as crop residues and imported wastes. This biomass is both a source of valuable plant nutrients and a threat to the environment.

The whole-farm perspective of agricultural waste treatment and management has been selected as a central theme for the 12th Ramiran conference. The ultimate goal of the work presented in the many contributions is to ensure a rational recycling of nutrients while controlling environmental hazards such as odour, ammonia (NH_3) and greenhouse

gas (GHG) emissions, nutrient leaching, and dissemination of pathogens, heavy metals or organic micro-pollutants in the environment.

Research activities typically focus on an individual production factor or environmental effect, e.g., reducing the N surplus of pig diets or increasing the energy yield from organic waste materials in digesters. But with a strong focus on one factor there is a potential that important side-effects or interactions are overlooked or disregarded because they occur "downstream" in the manure management chain.

The best evaluation of a change in practice is obtained using a holistic approach linking feeding, housing, treatment processes, storage conditions and field application practices. Finding practical methods or models to address the whole-farm perspective, however, is a great challenge. Firstly, agricultural production systems are extremely diverse, and secondly the various indicators of sustainability are not always easy to compare.

Manure and waste management in agriculture

Nutrient and organic matter flows on livestock farms are intimately connected with nutrient cycling associated with crop production (Fig. 1), and this connection of course also applies to pollutants and pathogens.

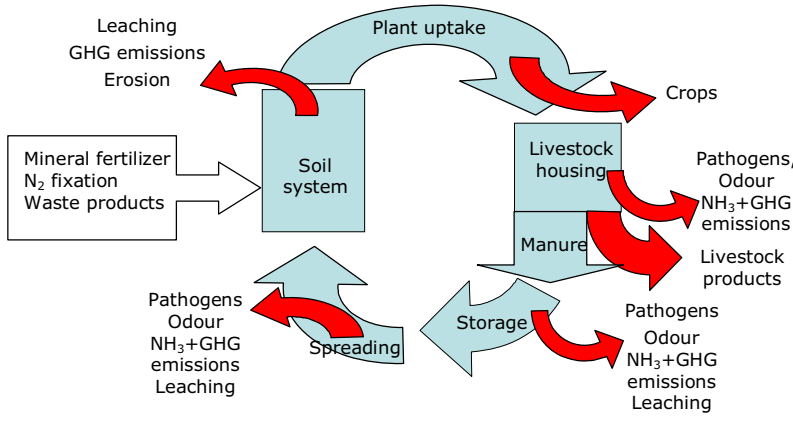

Figure 1. A simplified illustration of carbon and nutrient flows and environmental impacts on a livestock farm.

However, manure and waste management practices vary greatly between different parts of the world. The diversity is illustrated below, but without consideration of economics, nutrient use efficiency, environmental issues or hygienic risks.

In Southeast Asia (i.e., Vietnam) 85% of the livestock is produced by small holders (Tran Thi Dan et al., 2003). Pigs are kept in houses with solid floors, and farmers separate excreta into a liquid and a solid fraction by scraping the solid fraction off the floor by hand. The solid fraction is a commodity sold to farmers producing high value crops like coffee, vegetables, fruits etc. The liquid fraction is channelled to fish ponds, where carp grow on vegetation taking up the added manure nutrients. Sedimented organic matter is emptied from the ponds and used as a fertilizer.

European agriculture handles more than 65% of livestock manure as slurry, that is, a liquid mixture of urine, faeces, water and bedding material (Menzi, 2002). In Scandinavia slurry is typically collected in stores, which are designed to allow for extended storage so that spreading can take place before or during the growing season where a crop can utilize the nutrients. In other countries the slurry storage time is typically shorter and spreading times are often defined by existing storage capacity rather than considerations about nutrient use efficiency. Nutrients are recycled in so far as crops are used for animal feed, or when nutrients are returned to farmland in sewage sludge or other waste products.

The ten new member states of the European Union (EU) face a particular challenge, because subventions and the opening of markets have lead to a rapid intensification of livestock production, with frequent surpluses of nutrients spread on agricultural land. Also, NH_3 and GHG losses, as well as soil pollution with heavy metals, have increased due to the need to comply with the EU legislation banning the land-filling of sewage sludge.

In North America, confined animal production systems (mainly pig production) typically use pit storage of the manure underneath slatted floors. From the pit, excreta are flushed to lagoons where solids are settled and the retention time of the liquid fraction can be several years. The liquid may be discharged via “constructed wetland” treatment systems with intense denitrification, or the liquid may be applied on small spray fields.

Evidently the manure management strategies, pollution risks and needs for import of nutrients in wastes or mineral fertilizers of these systems are extremely different. Still, a set of basic *sustainability indicators* for a common model framework could perhaps be identified which are defined and quantifiable in all systems.

Side effects envisaged through whole-farm analysis

The importance of considering interactions between different parts of a management chain, or between different elements, can be illustrated by previous studies of manure N flows and links with C turnover.

Ammonia emissions from livestock production contribute to soil acidification, threaten N-limited ecosystems, and NH_4^+-based particles in the air represent a health risk for humans. Hutchings et al. (1996) assessed the effect of different mitigation strategies on total emissions from cattle farms with a whole-farm NH_3 model. An interesting conclusion from this study was that establishing a roof on a slurry tank to reduce NH_3 emissions during storage could increase total NH_3 emissions, if no precautions are taken to reduce NH_3 volatilization from the cattle slurry applied in the field. This was due to higher emissions after field application as a result of a higher slurry dry matter content, which in turn resulted from the exclusion of rain water during storage that would otherwise dilute the slurry and facilitate infiltration into the soil.

Ammonia volatilization during manure management and application, as well as N leaching from manure stores, are among the environmental problems which are being addressed in EU member states. National emission ceilings for NH_3 emissions are set in the NEC directive (Directive 2001/81/EC); storage capacity and timing of application are addressed in the Nitrate directive (Directive 91/676/EEC). It is essential that reductions in gaseous and point-source N losses are accompanied by compliance with reduced total N applications rates at manure spreading (such as those specified in Nitrate Vulnerable Zones) otherwise increased N loss from agricultural soil may exacerbate water pollution problems.

It is well known that emissions of methane (CH_4) and nitrous oxide (N_2O) during manure management are influenced by temperature, organic matter composition, nitrogen content and storage time. A recent model therefore linked C and N turnover in a dynamic prediction of CH_4 and N_2O

emissions during handling and use of livestock slurry. The model results indicated that anaerobic digestion, producing CH_4 at the expense of volatile solids, would cause a 90% reduction of CH_4 emissions during the subsequent storage. Also, a >50% reduction of N_2O emissions after spring application of digested as opposed to untreated slurry was predicted (Sommer et al. 2004). The calculations further suggested that daily flushing of slurry from the warmer environment in cattle houses to an outside store would reduce GHG emission by 35% compared to a situation where slurry channels were emptied once a month. Hence, residence time is also an important factor to define.

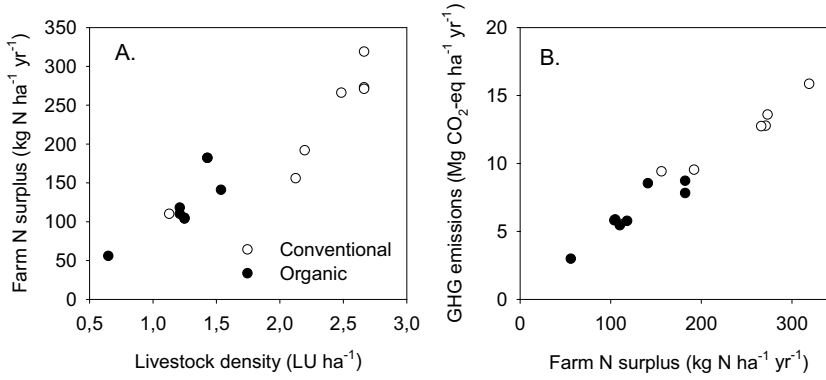

Fig. 2. A strong relationship between livestock density and N surplus (A), and a strong apparent relationship between N surplus and total greenhouse gas emissions (B) has been observed for organic and conventional dairy production (Olesen et al., 2006).

The last example considers a whole-farm model of C and N flows that was used to analyze dairy production under organic and conventional conditions in different parts of Europe (Olesen et al., 2006) and to evaluate GHG mitigation strategies (Weiske et al., 2006). The model quantified internal flows, as well as imports and exports, and it estimated GHG emissions, crop yields and milk production levels. Whereas the strong relationship between livestock density and N surplus (Fig. 2A) was not unexpected, it was interesting that the relationship appeared to be the same for extensive (organic) and intensive production systems. Even more surprising was the apparent relationship between N surplus and total GHG emissions (Fig. 2B), a relationship that was also observed by Schils et al. (2005) using a different model.

Conclusion

The examples of the previous section indicate that the consequences of introducing new technology or changing management may be difficult to predict without the support of a whole-farm model with quantitative descriptions of nutrient transformations and emissions at the different stages of a production cycle.

A model framework containing a set of basic, inter-linked sustainability indicators could be used to evaluate overall effects of new technology or management at an early stage. Ideally, this could ensure that research and development of one particular aspect of manure and waste management does not increase health risks or environmental problems elsewhere on the farm.

The many contributions for the conference have been allocated to one of seven main themes (see below). We hope that, by bringing together topics such as feeding strategies, manure and waste treatment and handling, energy production, hygienization, monitoring of nutrient flows and modelling, and with the participation of researchers representing a wide range of agricultural systems, we can strengthen the whole-farm perspective and practical relevance of the discussions.

Main themes of the 12th Ramiran conference

Energy production - biofuel & biogas, incineration
Livestock production, ammonia and greenhouse gases
Treatment technologies for organic effluents
Measuring and monitoring nutrient flows and emissions
Merging models for predicting emissions and nutrient leachings
Feeding of livestock - manure composition
Technology needs in the developing world

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A farm level approach for mitigating greenhouse gas emissions from ruminant livestock systems

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Introduction

Ruminant livestock systems contribute significantly to global warming through the emission of nitrous oxide (N_2O) and methane (CH_4). In the European Union (EU-15) the total emission of these greenhouse gases (GHG) was 456 Tg CO_2-equivalents in the reference year 1990, which was 10% of the total GHG emission (EEA, 2004). However, the contribution of agriculture to the total emissions varies considerably among member states (Table 1).

Between 1990 and 2002, the agricultural-related emissions of N_2O and CH_4 were reduced by 9%. This reduction did not result from specific GHG policies, but was driven mainly by reduced cattle populations and less nitrogen (N) fertiliser inputs. Projections for the year 2010 show an additional 3% reduction for the agricultural sector. The agricultural sector is way ahead of other sectors in the reduction of GHG emissions. Due to the potential for the implementation of cost-effective measures, there is even scope for further reductions so that the agricultural sector could share a larger part of the burden.

In contrast to industry, the emissions from agriculture are not confined to relatively few large sources, but are diffusely spread across Europe. On each individual holding the farm manager is responsible for the actions taken to achieve the farmers' goals. The objectives can be general, like income or continuity, or specific, depending on personal drive, conviction and style (Oenema et al., 2004). Consequently, farms are very different and thus require an individual approach when mitigation options are developed. To date, mitigation options focused mainly on a single gas and are often treated as isolated activities, independent of the farming system. The objective of this paper is to propose a framework for a farm

level approach, integrating GHG emissions and other environmentally relevant emissions, like nitrate leaching and ammonia volatilisation.

Table 1. GHG emissions in CO_2-equivalents (Tg) from agriculture in 2002 in the Netherlands (NL), Denmark (DK), United Kingdom (UK), France (F) and the European Union (EU-15).

	NL	DK	UK	F	EU-15
Methane					
- Enteric fermentation	6.4	2.8	16.9	28.9	135
- Manure management	1.7	1.0	2.1	14.1	72
Nitrous oxide					
- Manure management	0.2	0.6	1.3	2.9	18
- Agricultural soils	6.6	5.8	26.4	52.0	193
Total	15	10	47	98	416
Agriculture's share in total GHG (%)	7	15	7	18	10

General framework

A whole farm approach for ruminant livestock systems requires at least the definition of two essential farm compartments (Figure 1). The utilisation of home-grown roughage by animals and the return of excreta to the soil-crop system is a unique feature for ruminant livestock systems.

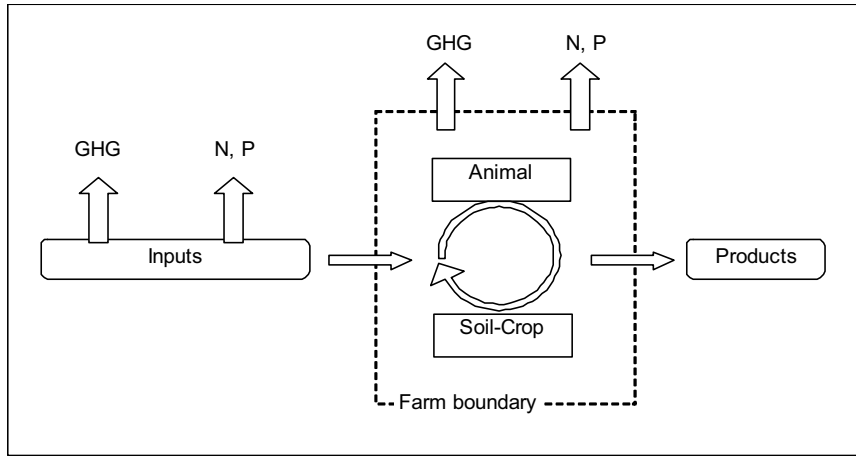

Figure 1. Basic elements of a whole farm approach.

This distinguishes them from intensive pig or poultry production systems where compound feeds are imported and animals and manure are exported, and from arable systems where fertilisers and manures are imported and crops are exported.

Inputs from ruminant livestock systems comprise those of biogenic origin like manures or biological N fixation by legumes, but also industrially manufactured inputs like feeds and fertilisers. The outputs are generally milk and meat products. Emissions occur at several stages within the nutrient cycle. The level of detail depends on the objective of the research. Choices to be made are related to:

- Number of farm components. A typical cycle used in N studies comprises components feed-animal-manure-soil-crop, but further sub-divisions are possible.
- System boundaries. In whole farm approaches the system within the farm gate is the minimum that should be studied. However there can be emissions occurring before inputs arrive at the farm, or after products leave the farm. Therefore it can be justified to include pre- and post-chain effects into the whole farm approach.
- Simulation methodology. Whole farm models are usually a diverse mix of empirical and mechanistic modelling, with more or less reliance on one of them. With respect to GHG, emissions can be calculated with emission factors, comparable to the IPCC methodology, or simulated with mechanistic (sub)models.
- Aspects to be studied. In this paper we focus on GHG emissions, mainly in relation to N cycling. However, this can be extended endlessly with aspects such as phosphorus, energy use, heavy metals, landscape, animal welfare and milk quality. Inclusion of financial evaluation of mitigation options is a must when it comes to potential implementation by farmers.

The whole farm approach should not be seen as a replacement of the IPCC methodology. The choice depends on the objective. The IPCC method is used to prepare transparent and consistent inventories for national emission reporting. Calculations on farm scale are useful to explore mitigation options for individual farms. However, for fulfilling national reduction targets there is a need to ensure that such mitigation options are also reflected in the national emission inventories.

Current GHG models

The development of whole farm approaches has been taken up recently by several research groups in Europe (Table 2).

- *DairyWise* is an existing empirical model used for technical and financial simulation of dairy farms (Van Alem and Van Scheppingen, 1993). It is used in research, consultancy, teaching and policy development. The core of DairyWise is a grass growth model and a herd model. The model simulates N, P and K flows, including nitrate (NO_3) leaching and ammonia (NH_3) emissions. Recently, a GHG module has been added in which CH_4, N_2O and CO_2 emissions are calculated with refined emission factors (Schils et al., 2006a) . Pre-chain emissions are only calculated for energy use and the associated CO_2 emissions. Furthermore, DairyWise generates a detailed overview of farm costs and income.
- *FarmGHG* is a model of carbon (C) and N flows on dairy farms (Olesen et al., 2006). The model is designed to calculate all direct and indirect gaseous emissions from dairy farms. The flow of all products through the internal chains and through imports and exports from the farm are modelled. FarmGHG allows different methodologies for emission estimates. They include the tier 1 and 2 IPCC methods, but also a default FarmGHG methodology.
- *$SIMS_{DAIRY}$* is a new modelling framework (Del Prado et al., 2006) which integrates existing models for N, CH_4 and P, matrices to score farm sustainability attributes and an economic model. SIMSDAIRY is very sensitive not only to management but also weather, topography and soil characteristics and is capable to optimise farm management practices to

Table 2. General characteristics of farm models.

	DairyWise	FarmGHG	$SIMS_{DAIRY}$	FarmSim
Model type	Empirical	Empirical	Semi-Mechanistic	Semi-Mechanistic
CH_4 and N_2O	x	x	x	x
CO_2	x	x		x
NH_3 and NO_3	x	x	x	x
P	x		x	
Pre-chain	x	x		
Economics	x		x	
Miscellaneous			x	

meet user multi-weighted criteria and to explore the possible impact of application of mitigation options on (i) pollutants such as: N_2O, CH_4, NH_3, NO_x, NO_3 and P; (ii) economic profitability; (iii) milk quality; (iv) biodiversity; (v) landscape; (vi) soil quality and (vii) animal welfare.
- *FarmSim* has been designed to describe the above and below ground C and N fluxes in cattle farms, and calculate the net balance of GHG emissions (Saletes et al., 2004). The model is structured in 9 modules, requiring detailed data inputs on the farm structure, the herd, grazing, housing, manures, fertilisation, crops, feeding, waste production and storage. Emissions of N_2O, CH_4 and CO_2 from grassland, including the grazing animal, are calculated by the mechanistic PASIM model. The other emissions, e.g. from housing and manure storage, are calculated in a spreadsheet module of FARMSIM according to the IPCC methodology.

Evaluation of mitigation options
Recent research has delivered a wide range of mitigation options, generally focusing on a single gas, and usually considered as isolated activities. However, it is farmers who decide on implementation of mitigation options and judge the effectiveness in the context of a whole farm system. The integrated evaluation at the whole farm scale ensures that crucial interactions between C and N cycles are taken into account, and possible trade-offs with other emissions are indicated. Therefore, the whole farm approach is a powerful tool to evaluate mitigation options in the appropriate context, at the level of the decision maker.

GHG mitigation options can then be assessed against several criteria. Currently, mitigation options are usually evaluated against other EU or country-specific environmental goals such as the reduction of N and P losses. Reduced grazing, for example, primarily aimed at a lower N_2O emission, also reduces NO_3 leaching, but may increase CH_4 and NH_3 emissions.

Considering manure management, mitigation options may include improved application techniques, frequent removal from the house to an outside storage, cooling of the slurry channel, solid covers on slurry tanks, air-tight covers on solid manure storages and anaerobic digestion. Among the options mentioned, digestion of animal manures appears to have the largest potential for reducing net GHG emission (Sommer et al., 2004; Weiske et al., 2006), because it targets several greenhouse gases. Co-digestion is often promoted to stimulate the digestion process. If these

substrates contain N, co-digestion may have an adverse effect on N utilisation at the farm level (Schröder and Uenk, 2006). The whole farm approach can also be extended to other particular issues. For instance increasing the genetic merit of the dairy herd is regarded as one of the most effective CH_4 mitigation options as CH_4 losses per litre of milk is greatly reduced. However, this positive effect on CH_4 is generally associated with welfare and milk quality trade-offs, as high genetic merit cows have an increased risk of suffering health disorders and reduced protein and butterfat content in milk. Health disorders in turn may carry a price in terms of higher replacement rates, and thus higher GHG emissions from the required additional young stock. In addition to research, the whole farm approach is also useful for the communication of mitigation options to farmers, especially if the model also evaluates additional costs and benefits.

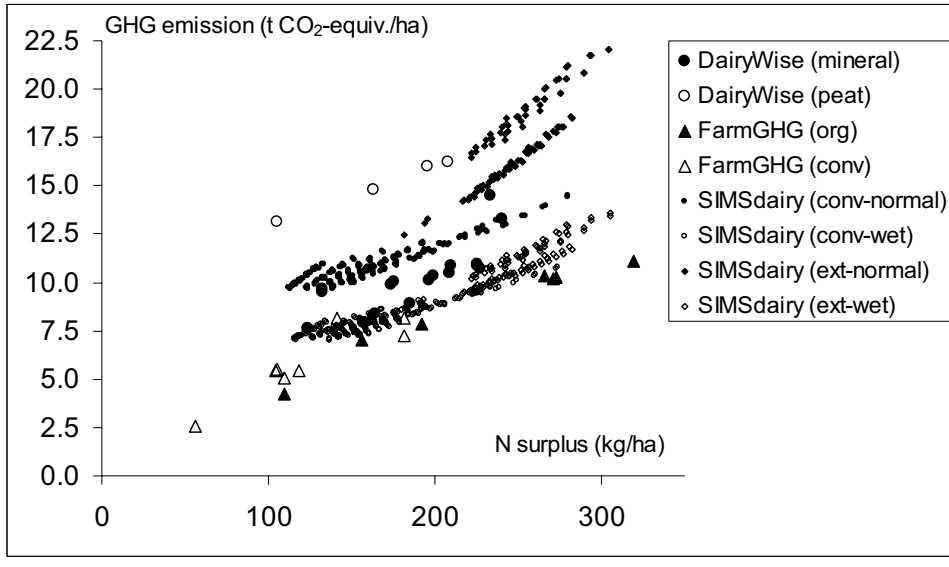

Figure 2. Relationship between GHG emission (N_2O and CH_4) and N surplus. DairyWise: conventional NL farms on mineral and peat soils. FarmGHG: organic and conventional farms in several regions of Europe. $SIMS_{DAIRY}$: farms in UK with either conventional grazing or extended grazing, and either under normal (720 mm/year; 11-12 ^{o}C during growing season) or wet conditions (1493 mm/year; 10-11 ^{o}C during growing season).

Nitrogen surplus

In recent years, N policies have had a substantial effect on dairy farming. In the Netherlands, farms have improved the N utilisation through improved manure management, less fertiliser and feed import, reduced grazing and less young stock per cow. Unintended, the improved N management also reduced the GHG emissions (Schils et al., 2006b). Modelling exercises for a range of farm types across Europe confirm the positive relationship between N surplus and GHG emissions (Figure 2).

The range of N surpluses was generated by differences in stocking rates in combination with several specific farm management practices. Each kg of N surplus corresponds with a GHG emission of approximately 30 to 70 kg CO_2-equivalents. Although there is a variation between farm types and conditions, and of course between the models used, there is certainly scope to use N surplus as a proxy for GHG emissions. Similar relationships are found with FarmSim (data not shown). However, FarmSim also includes a carbon sink for grassland soils, changing the total farm scale picture.

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Nutrient losses from manure management

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Abstract
Manure management systems are conducive to nutrient losses, but the magnitude of the loss highly depends on the nutrient element, the manure management system and the environmental conditions. This paper briefly reviews the diversity in manure management systems in practice and the nutrient losses from these systems. Losses decrease in the order: N >> S > K, Na, Cl, B > P, Ca, Mg, Fe, Mn, Cu, Zn, Mo, Co, Se, Ni. Gaseous losses are in the range of 10 to 50% for N and of 2 - 10% for S. Leaching losses are usually << 10% of the nutrients initially present in the manure. Calculations with MITERRA-Europe indicate that N losses from manure management systems in EU-25 are almost 3 Tg. Decreasing nutrient losses requires analyses of the feed – animal – manure – crop production chain, and farm-specific technological and management measures.

Introduction

Animals retain only 5-50% of the nutrient elements in the feed, depending on animal species, the nutrient element and the nutrient content in the animal feed. The major fraction (roughly 50-95%) is excreted via dung and urine, and animal manure therefore is a valuable source of nutrients. However, nutrients in animal manure are conducive to dissipating into the wider environment, depending on the nutrient element, the animal manure management system and environmental conditions (Tamminga, 2003; Sommer et al., 2006).

The management of animal manure has gained importance over the years, because nutrient losses from animal manure greatly influence the agronomic and environmental performances of animal farming systems. The emphasis is often on nitrogen (N), because of its key roles in both, animal and crop productivity and environmental impacts (Rotz, 2004). Yet, the value of manure also depends on the availability of other nutrients, and neglecting these can be counterproductive (Zingora et al., 2006). This paper briefly reviews nutrient losses from manure management systems.

Manure management systems

There is a wide variety of systems (Table 1) and environmental conditions, and as a consequence, nutrient losses vary greatly. On a global scale, 50% (range 40-60%) of animal excrements are voided in pastures and left there unmanaged. The nutrients in the excrements are taken up by the grass, stored in the topsoil or lost to the wider environment. In parts of Asia and Africa, the feces are collected, dried in cakes and subsequently burned for cooking and heating purposes. The ashes are usually returned to gardens and crop land, but the amount and plant-availability of some nutrients have decreased (especially for N and P, respectively).

Table 1. Animal manure storage and management systems in the world; their characteristics, relative importance and estimated total N loss (After Oenema and Tamminga, 2006).

Manure management systems	Characteristics	Relative impor-tance, %	N loss, %
Pasture/ range	Dung and urine from grazing animals in pastures and rangeland is allowed to lie and is not handled	40-60	10-50
Burned for fuel	Dung is collected and dried in cakes and burned for heating or cooking	5-10	20-90
Dry lots	In dry climate, dung from confined animals in unpaved feedlots is removed periodically and applied to land elsewhere	5-10	10-60
Solid storage	Dung (often with litter) from confined animals is collected and stored in bulk for months before application to land	5-10	10-60
Composting	Dung (often with litter) from confined animals is collected and composted via managed aeration (often for phyto-sanitary reasons) before application	<5	20-80
Daily spread	Dung (and urine) from confined animals is collected and applied to land regularly (daily)	5-10	5-50
Liquid/slurry	Urine (with or without dung) from confined animals is collected and stored in concrete/lined tanks for months until application to land	5-10	5-30
Pit storage	Urine (with or without dung) from confined animals (pigs) is stored in a pit beneath the confinement for months until application to land	5-10	10-30
Anaerobic lagoons	Urine (with or without dung) from confined animals is flushed with water to lagoons and stored until treatment and/or application to land via irrigation	<5	10-50
Anaerobic digesters	Dung or mixtures of dung and urine are digested to produce methane (CH_4) for energy, while the digester effluent is often applied to land	<5	0-10
Anaerobic / aerobic treatment	Animal excrements are treated (an)aerobically to decrease the amount of suspended solids, organic C and N before discharge to surface waters	<5	20-90

Confined animals void their urine and feces in stables, barns, sheds and corals, where it is stored for some time, before treatment and/or application to land (Table 1). About 50% (range 40-60%) of the global amount of manure from domesticated animals is voided in stables, barns, sheds and corals, but this estimate is rather uncertain. There is a wide variety in animal manure storage and management systems and there are many intermediate forms. The systems differ in the types of manure collected (urine and/or dung), litter amendments, oxygenation, length of storage period, and bottom sealing and top coverage (Menzi, 2002).

In land-based animal systems, most of the animal manure is ultimately returned to the land that produced the animal feed. In specialized animal production systems, the manure is disposed of elsewhere, as the land-base is missing. At best, the manure is applied to the land of nearby farmers, but often it is processed, composted, treated, discharged or dumped, and with considerable losses of nutrients to the environment.

Nutrient loss pathways from manure management systems

Losses from animal manure management systems roughly decrease in the order: N >> S > K, Na, Cl, B > P, Ca, Mg, Fe, Mn, Cu, Zn, Mo, Co, Se, Ni. This order is related to the reactivity, speciation, fugacity and mobility of the nutrients. Major N loss pathways are NH_3 volatilization, (de)nitrification with gaseous emissions of NO, N_2O, N_2, and leaching of nitrate (Figure 1).

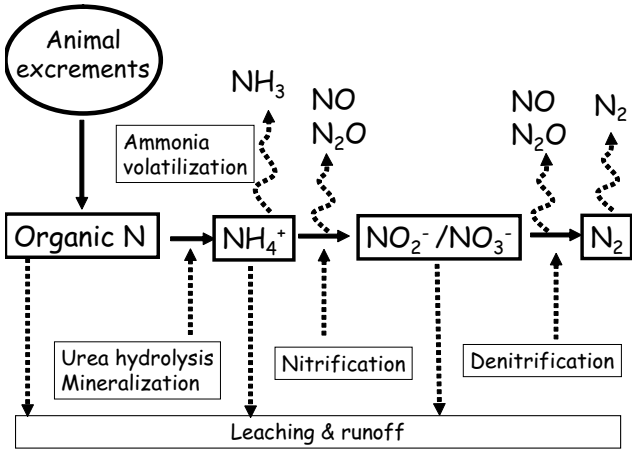

Figure 1. Sequence of N transformation processes, and the loss pathways of N from animal excrements.

Sulfur (S) is also involved in reduction-oxidation reactions, yielding S species with a range in fugacity and solubility. Main S loss pathways from manure are the volatilization of sulfides (H_2S) and sulfur oxides (SO_2) and the leaching of sulfate (SO_4). Potassium (K), sodium (Na), chloride (Cl) and boron (B) have low fugacity and high solubility in water, and main loss pathway is via leaching. High concentrations of for example K, Cl and NO_3 in wells near farm houses are often seen as evidence of leaching losses from manure heaps. Finally, the elements P, Ca, Mg, Fe, Mn, Cu, Zn, Mo, Co, Se and Ni have low mobility because of low fugacity and solubility. However, some (Fe, Mn, Se) are redox-active, and some elements form complexes with dissolved organic carbon and inorganic anions (sulphate, chloride) and thereby increase there mobility. Areas with high livestock density and in particular the topsoil underneath manure heaps are usually enriched with these elements.

Estimating and decreasing N losses from manure management

Here, we focus the discussion on N because N has the largest loss from manure management. In general, there is lack of accurate information about actual manure management in practice, and as consequence estimates of N losses at (inter)national scale are uncertain. We have estimated N losses from manure management in EU-25, using the model MITERRA-Europe. Ammonia emissions are the main loss pathway, followed by (de)nitrification and leaching plus runoff. Mean NH_3 loss at NUTS-2 level in EU-25 is shown in Figure 2. The model results show (nearly) linear relationships between livestock density, N surpluses, and NH_3, CH_4, N_2O emissions (see also Petersen et al., this issue), but the slope of these relationships differs between EU Member States, due to differences in N excretion, manure management system, and mitigation measures (Velthof et al. in prep.).

Decreasing N losses from manure involves farm specific analyses of the feed – animal – animal produce – animal manure – crop production chains. The weakest part of this chain determines the best option. Often, significant improvements can be made by improvements at animal level and animal manure level. Improvements at animal level require genetic improvement of the herd, a better description of feed, and higher quality feed. Anaerobic digestion of manure during storage has the advantage of producing methane (CH_4) to be used as biofuel, and less emissions of odours, NH_3, N_2O and CH_4 during storage, and less emissions of N_2O

following application to land. Improving the utilization of animal manure as source of nutrients has the potential of replacing fertilizer nutrients.

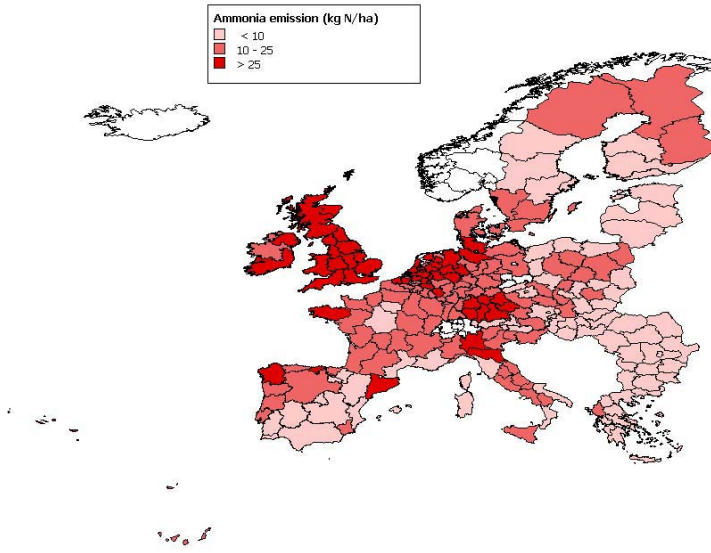

Figure 2. Ammonia emissions from animal manure management in EU-25, calculated at NUTS-2 level (Velthof et al, in prep.). White color is 'no data'

Conclusions

Animal manure management in practice is diverse. On a global scale, only about 50% of the manure excreted by domesticated animals is collected in confinements and manageable. Depending on the management and environmental conditions 10-50% of the N and 2-10% of the S is lost through gaseous emissions. In addition, leaching losses may occur, when the manure system is not sealed at the bottom and/or covered on top. Nutrient losses can be greatly decreased and nutrient use efficiency at farm level greatly increased following the implementation of a series of technological and management measures.

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Available on request.

Manure as a key resource to sustainability of smallholder farming systems in Africa: An introduction to the *NUANCES* framework

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Preamble

Although manure is seen as a problematic waste in many intensive agricultural systems in developed countries, it is a key resource to sustain productivity of the majority of smallholder farming systems in Africa. Spatial patterns of resource use are consistent across different farming systems. Livestock are the central means of concentration of nutrients within farming systems, resulting in inequitable redistribution of nutrients from common to cultivated lands and poorer households to farms of richer households. Productivity gains are achieved by concentration from common lands, or concentration to infields, at the long-term expense of declining productivity in remote fields and common lands.

Development of principles for enhancing efficient allocation of scarce resources must therefore be seen within the complex dynamics of interacting temporal and spatial scales. Nutrient management for crop production should focus on efficient use within complete rotations and across different fields within the farm, rather than on the requirements of individual crops. The livelihoods of farming families depend on complex interactions between competing demands for investment of cash and labour within and beyond farm boundaries. They are particularly sensitive to opportunities for off-farm earnings through markets for produce and employment in urban centres, which form the major sources for investment in agriculture. Indeed, a frequent investment goal of farmers is schooling of future generations to allow an escape from agriculture, rather than investment in the farm. Combinations of socio-economic and agro-ecological conditions can provide windows of opportunity in both time and space that favour investment in particular forms of management. A research framework is proposed which represents a farm

livelihood systems as a set of interacting components. This can be used to explore the short and long-term trade-offs of introducing new technologies, and to evaluate effects of policy on farms of differing resource endowment.

Introduction

African farming systems are highly heterogeneous: both in terms of the wide variability in resource endowment of farmers and the management of the individual fields within a farm. Farmers preferentially allocate manure, mineral fertilizers and labour to in-fields, resulting in strong gradients of soil fertility decline with increasing distance from the homestead as this provides the highest returns (Tittonell et al., 2005a; Tittonell et al., 2005b; Zingore et al., 2006). Manure is a regarded by farmers as a major resource provided by cattle – largely because much of the land is characterised by poor productivity that results from continuous cultivation on soils that are often inherently poor in nutrients.

The *NUANCES (Nutrient Use in ANimal and Cropping systems – Efficiency and Scales)* framework

We are developing an integrated analytical framework with the aim of embedding analyses of the potential for different potential soil improving technologies within the wider livelihood strategies of farmers (see http://www.africanuances.nl/). Few studies have compared the potential of all the different options for soil fertility improvement or the ways that they can best be combined at farm scale. The scheme in Figure 1 illustrates how diverse, complex smallholder farming systems can be understood as a limited set of interacting components. The components that are used to represent a farm livelihood within NUANCES are analysed using simple models of the sub-systems.

Our overall aim is to increase our understanding of the tactical and strategic decisions farmers make in allocating resources and the underlying trade-offs, where immediate needs of the family may often override the possibilities of investing in the longer-term sustainability of the farm. By synthesizing knowledge we can analyse trade-offs between implementation of various soil fertility technologies for smallholder farmers in mixed crop/livestock systems in Africa. The emphasis is on efficiency of targeting and use of nutrients and legume-based soil improving technologies, with outputs evaluated in terms of costs, benefits

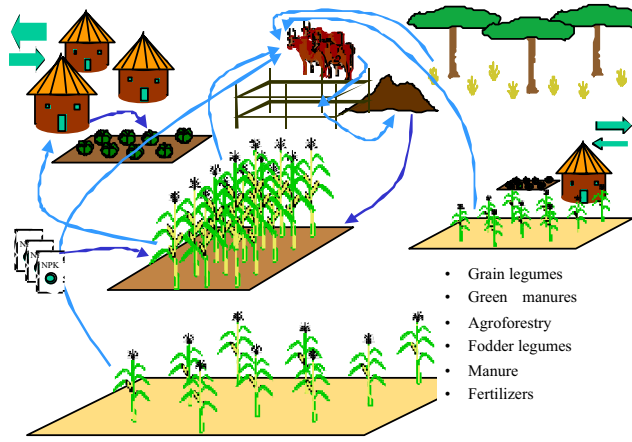

Figure 1. A representation of the key components of the farming system typical to smallholder farming systems in sub-Saharan Africa, that forms the core of the NUANCES framework. See text for further explanation.

and compromises in productivity, economics and environmental services. The potential for using integrated crop-livestock simulation models in scenario analysis was reviewed by Thornton and Herrero (2001) who warned of the risk for being drowned by complexity. Our approach is to

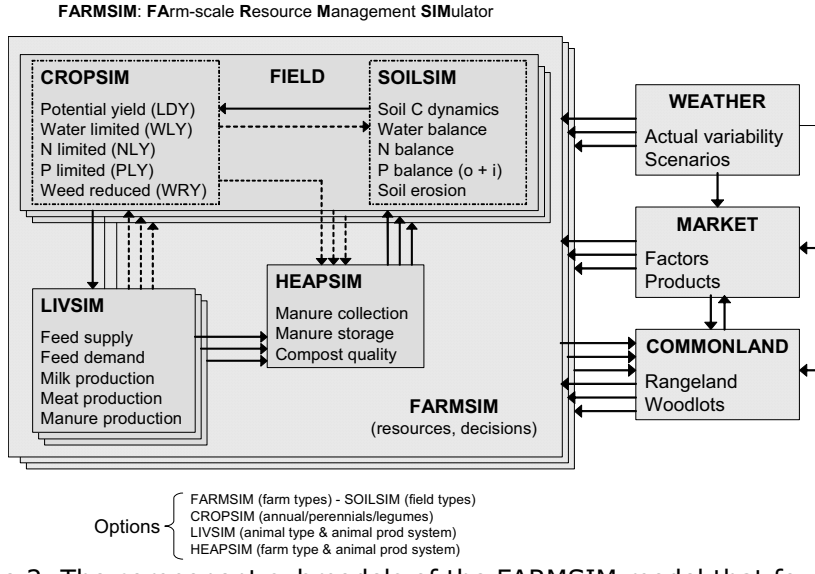

Figure 2. The component submodels of the FARMSIM model that forms the core of the NUANCES framework (from Tittonell et al., 2006).

use simple component subsystems to avoid being overwhelmed by detail, but to include all relevant components to allow analysis of realistic scenarios (Figure 2). Fields are represented by the FIELD model that contains linked crop and soil models. Livestock feeding, productivity of milk, meat and manure production (per animal and including herd dynamics) are represented by LIVSIM and manure management by HEAPSIM.

Consideration of both socio-economic and agro-ecological conditions allows identification of the windows of opportunity in both time and space that will favour particular forms of management. Thus, the attractiveness of technologies grows, and wanes, as intensity of land use and links to urban markets for both produce and employment develop (de Ridder et al., 2004). For a given combination of agro-ecological and socio-economic conditions, a multitude of different combinations and trajectories of response by farmers may be equally productive. Farmers who have ready access to mineral fertilizers have less interest in labour-demanding soil improving technologies. Equally, poor households that are often labour-constrained are unlikely to be able to invest in labour-demanding technologies due to the need to use their labour to generate income. Technology development specifically for poor farmers needs to target labour-saving approaches: in Zimbabwe management to increase the abundance of leguminous weeds in farmers' fallows shows promise in raising base yields of maize, marginally in absolute terms, but significantly in terms of food provision for poor households (Mapfumo et al., 2005).

Fundamental questions for analysis of resource dynamics and potential for modification of complex farming systems relate to the degree of simplification of process and the site-specific knowledge that is necessary to integrate and move from one scale to the next. Understanding which factors are the most important in determining site-specific response to changes in management is a central issue.

Efficiencies of N use through livestock (LIVSIM and HEAPSIM)

Rufino et al. (2006) conceptualised African farming systems in four sub-subsystems through which nutrient transfer takes place: 1. Livestock: animals partition dietary intake into growth and milk production, faeces and urine; 2. Manure collection and handling: housing and management determine what proportion of the animal excreta may be collected; 3.

Manure storage: manure can be composted with or without addition of plant materials; 4. Soil and crop conversion: a proportion of the N in organic materials applied to soil becomes available, part of which is taken up by plants, of which a further proportion is partitioned into grain N (Figure 3).

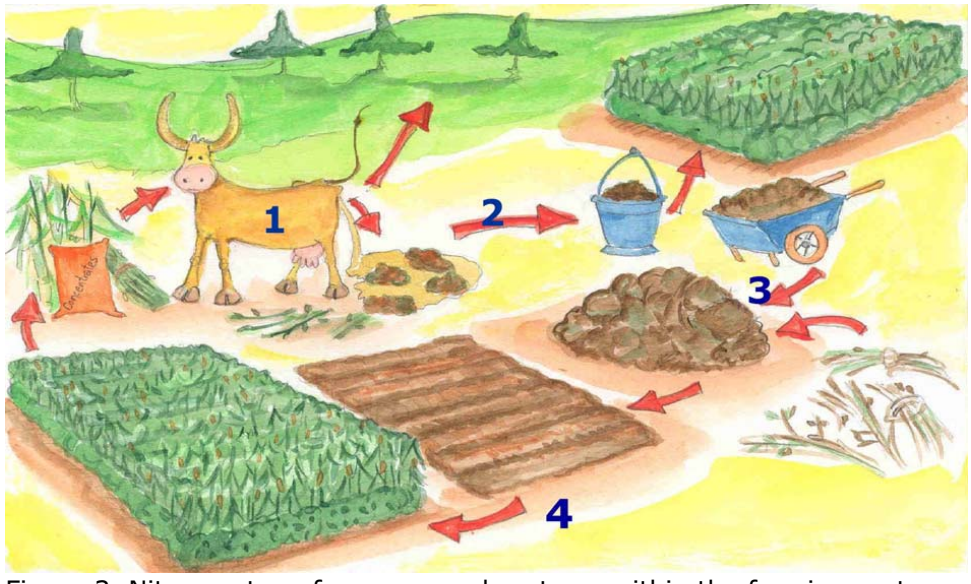

Figure 3. Nitrogen transfer among subsystems within the farming system (see text for detailed explanation. From Rufino et al., 2006).

An exhaustive literature review showed that partial efficiencies have been much more commonly calculated for the first and last steps than for manure handling and storage. Partial N cycling efficiencies were calculated for every sub-system as the ratio of nutrient output to nutrient input. Estimates of partial N cycling efficiency for each sub-system ranged from 46 – 121 % (Livestock), 6 – 99 % (Manure handling), 30 – 87 % (Manure storage) and 3 – 76 % (Soil and crop conversion). Overall N cycling efficiency is the product of the partial efficiencies at each of the steps through which N passes. Direct application of plant materials to soil results in more efficient cycling of N, with fewer losses than from materials fed to livestock. However, livestock provide many other benefits highly valued by farmers, and animal manures can contain large amounts of available N which increases the immediate crop response. Making most efficient use of animal manures depends critically on improving manure handling and storage, and on synchrony of N mineralization with crop uptake. Manure is an excellent soil amendment as it contains multiple nutrients and can

overcome deficiencies of P, S, Ca, Mg, Zn etc which are widespread in the sandy soils that cover a large part of the African land mass.

Conclusions

Manure is a key resource to the sustainability of farming in Africa and is in short-supply! Measures to improve manure handling and storage are generally easier to design and implement than measures to improve crop recovery of N, and should receive much greater attention if overall system N cycling efficiency is to be improved.

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Impact of nutrition on nitrogen, phosphorus and trace elements in pig manure and emissions in the air

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Abstract
In order to reduce N, P and trace elements in pig manure, researches toward a better agreement between supply and requirement were undertaken, and ways to improve the availability of these elements in feedstuffs were investigated. Substantial reduction in N excreted by pigs can be achieved by phase feeding combined with a better adjustment of dietary amino acid balance. Feeding pigs with low N diets also allows a reduction of ammonia emission and to some extend the production of malodorous compounds. Phase feeding is also effective in reducing P excretion. However, low digestibility of P in feeds remains the main problem although it is partly alleviated by the supplementation of pig diets with microbial phytase. In the same way as for N and P, lowering Cu and Zn dietary supply is obviously an efficient way to reduce their excretion. In a whole farm perspective, improving the efficiency of nutrient utilisation by the animals is an efficient way to reduce the import of nutrients from outside the farm and decrease the risk for the environment.

Introduction

For a sustainable pork production, emission of pollutants from pig farms and use of non-renewable resources should be decreased as far as possible. Through the last decades, ways to reduce environmental impact of N, P and trace elements in pig production were investigated, because of the potential negative impact of these elements on ground and surface water, and on air quality. Besides, the use of diminishing phosphate world reserves should be limited. Among other, the nutritional approach has received great attention from researchers. The nutritional approach relies on improvements in our knowledge of pig's requirement in order to achieve a better agreement between supply and requirement. The improvement of nutrient availability in feedstuffs is the second way to be investigated. The aim of this paper is to give a review of the nutritional possibilities to reduce N, P and trace elements excreted by pigs, and to take knowledge of the means that could be or are already implemented in practice.

Reduction of N excretion in pig manure

Efficiency of protein utilisation by pigs depends on both the composition of the diet and the physiological status or growing stage of the animals. In growing-finishing pigs fed a cereal-soybean meal diet, about 32% of N intake is retained. Nitrogen excreted in the faeces amounts to 17% of intake and corresponds mainly to the undigested protein fraction and the endogenous losses. Digested proteins are absorbed as amino acids which are used for protein synthesis. Obligatory losses of amino acids occur in connection with protein metabolism (turnover) and renewal of skin and hair. The remaining amino acids are catabolised and excreted mainly as urea. With conventional diets, this last fraction is often the most important. Average efficiency of N retention is the lowest in sows (23%), intermediate in growing pigs (34%), and the highest in weaners (48%) (Dourmad et al., 1999). Two complementary approaches can be used to improve the efficiency of N utilisation by pigs and, consequently, to reduce N excretion.

The first approach is to ensure adequate protein and amino acid supplies over time according to the growth potential of the animals or their physiological status. In reproducing sows, N excretion was reduced by 20 to 25% when different diets were used for pregnancy and for lactation, compared to the use of a single diet. In fattening pigs, Latimier and Dourmad (1993) measured about 10% reduction of N in the slurry when different diets were used during the growing and the finishing periods, compared to feeding the same diet over the whole period.

The second approach is to improve dietary amino acids balance and consequently reduce protein content of the diet. This can be obtained through the combination of different protein sources and/or the utilisation of industrial amino acids. In fattening pigs, Dourmad et al. (1993) measured a 35% reduction of N excretion after improvement of the dietary amino acid profile, whereas feed intake, average daily gain, feed efficiency and carcass composition were not affected.

It must be pointed out that the development of such feeding techniques for reducing N excretion by the pigs requires a good knowledge of amino acid availability in the feedstuffs, and of changes in amino acids requirements according to growing stage or physiological status. This is now within reach with the use of modelling techniques for predicting the

requirements, and with a better knowledge of variations of amino acid availability in feedstuffs.

Reduction of gaseous losses

By alternatives in feeding practices it is possible to influence urea concentration in the urine and the pH of slurry, which will affect ammonia release. When pigs are fed low protein diets, urine urea concentration and pH decrease (Canh et al, 1998; Portejoie et al, 2004). When water is available ad libitum, feeding low protein diets also results in a reduced amount of urine produced due to decreased water consumption (Portejoie et al., 2004). These changes in slurry characteristics result in lower ammonia losses during collection, storage and following application of slurry (Canh et al., 1998; Hayes et al., 2004; Portejoie et al., 2004). For instance, in the study of Portejoie et al. (2004) ammonia emissions during slurry management was decreased by 63% when dietary protein was decreased from 20 to 12%.

The electrolytic balance (EB), calculated as ($Na^+ + K^+ - Cl^-$), is often used by nutritionists to evaluate the acidogenicity of the diet. When dietary CP content is reduced, EB also decreases. This explains the effect of CP on urinary pH. However, as shown by Canh et al. (1998), more drastic changes in urinary pH and ammonia volatilisation can be obtained by adding Ca-salts: $CaSO_4$ or $CaCl_2$ instead of $CaCO_3$. The addition of Ca-benzoate (Canh et al., 1998) or benzoic acid was also effective in reducing slurry pH and ammonia volatilisation, these products being metabolized to hippuric acid which is rapidly excreted in urine. A similar effect was observed with adipic acid (van Kempen, 2001) which is partially excreted in urine.

Urea N excretion can also be reduced by including fibrous feedstuffs in the diet. When more fermentable non-starch polysaccharides (NSP) are included in the diet, some of the N excretion is shifted from urine to bacterial protein in faeces (Canh et al., 1998), while total N excretion is not affected. Moreover, in the study of Canh et al. (1998) slurry pH was decreased with the use of fermentable NSP by volatile fatty acid (VFA) formation in the hindgut of the pig and in the slurry.

Effect of feeding on odours

Odours are generated by volatile compounds that pigs excrete with manure or that are released during manure storage (de Lange et al.,

1999). Few studies have evaluated the direct effect of diet manipulation on odour production, mainly because it is extremely difficult to objectively assess odours. As previously shown, protein nutrition affects ammonia production, but ammonia level is not well correlated with odour strength. Hayes et al. (2004) showed a significant reduction of both ammonia and odour emissions when protein content was reduced. Hobbs et al. (1996) reported that the concentration in the air of nine out of ten odorous compounds was significantly reduced when low protein diets were fed to the pigs. The manipulation of gut fermentation could also be a way for altering the production of odorous compounds, such as skatole (de Lange et al., 1999). Using a different methodology for assessing "pleasantness", "irritation" and "intensity" scores of odours, Moeser et al. (2003) were able to differentiate significantly diets differing in their composition. The diets that yielded manure with the worst odour were high in sulphur (garlic or feather meal). In contrast, the purified diet mainly based on starch and casein presented the lowest score (most pleasant).

Reduction of P in pig manure

In growing-finishing pigs fed a cereal-soybean meal diet, about 45% of P intake is absorbed, of which around 30% is retained, the remaining being excreted via urine (Poulsen et al., 1999). Totally, 70% of P ingested is excreted by pigs. In order to reduce P losses, P supplied to pigs should be adjusted to their requirement, and strategies to improve P availability should be implemented. This approach relies on an accurate knowledge of P availability and P requirement according to the physiological status of pigs.

A first approach to improve P digestibility is to use highly digestible mineral P supplements. However, most strategies implemented to reduce P excretion refer to improvements in phytic P utilisation by pigs. In many countries, microbial phytase is currently introduced in diets for pigs because of its well documented positive effect on P digestibility. Total P supply may be decreased, resulting in a reduction by almost 40% of P excreted (Jongbloed and Lenis, 1992).

Methodologies to reduce trace elements in pig manure

Because they are used as growth promoter or because large safety margins are applied, Cu and Zn are often oversupplied in pig diets. Consequently, manure is highly concentrated in these elements, which accumulate in soil and may cause at medium or long term toxicity to

plants and micro-organisms (Jondreville et al., 2003). Moreover, when a treatment is applied to the slurry, Cu and Zn concentrate in the solid fraction and their concentration often exceed the maximal values for the valorisation of these products as organic fertilisers. The only way to decrease concentrations of trace elements in manure is to limit their incorporation in the diet.

The incorporation of 150 to 250 ppm Cu in pig diets has been employed for a long time because of its growth promoting effect. This practice is currently authorized in EU for pigs up to 12 weeks, with diets containing a maximum of 170 ppm Cu. After 12 weeks of age Cu is no more used as growth factor in EU, and the maximal allowed level of incorporation is 25 ppm. Compared to the former regulation (175 ppm up to 16 weeks of age and 100 ppm thereafter), this resulted in a drastic reduction of Cu in manure, by almost 60%.

Supplementing weaned piglets diets with 1500 to 3000 ppm Zn as ZnO was also reported to stimulate their growth (Poulsen, 1995). In fact, in 2003 (EC, 1334/2003) the maximal level on Zn incorporation in pig diets was reduced to 150 ppm, compared to 250 ppm before. This level is much closer to the published requirement which vary between 100 and 50 ppm according to growing stage and authors. Compared to a situation in which weaning pigs are fed diet with 2500 ppm Zn from 8 to 15 kg BW and 250 ppm thereafter, the present EU regulation (150 ppm Zn) resulted in 53% reduction of excretion.

Nevertheless, although the situation has been drastically improved by these new regulations, Cu and Zn still largely exceed the export by crops when slurry is spread on the basis of 170 kg N/ha. Further reduction in excretion should be possible in the future, but this will require improving our knowledge on animal's requirements and dietary nutrient availability.

Conclusion and perspectives

Improving the efficiency of nutrient utilisation by the animals is an efficient way for reducing the excretion in the slurry. In a whole farm perspective this is an efficient way to reduce the import of nutrients, especially N, P and trace elements, from outside the farm. Moreover, because nutrition also affects the chemical composition of the effluents, ammonia emissions from livestock housing and during storage and spreading of manure are also reduced.

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The contribution of separation technologies to the management of livestock manure

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Separation processes have a distinct role in the management of livestock slurries but it is important to recognise their limitations. Equipment generally falls into those based on screening (which can produce a fibrous and seemingly dry product) or settling which often results in a sludge. Although physical separation can remove up to 80% of the total solid content this will include a relatively small part of the soluble nutrient and reactive organic matter; this is particularly so where separation is based on screens. Total clarification of an effluent is possible but its polluting strength is still not greatly reduced. It is important to minimise the water content and volume of the concentrate stream especially for the production of organic products in subsequent processes.

Introduction

Management techniques at a farm for livestock manure may be implemented (a) to make the farm operation more efficient (improved handling), (b) to reduce the various pollution risks from manure, (c) to reduce nuisance factors such as offensive odour, (d) to respond to hygienic concerns and (e), to draw some value from the solid and liquid wastes produced at the farm. This paper focuses on separation processes that can be used within this context. This is not an arbitrary division though as, in many instances, the separation process falls comfortably into the farming system. These techniques are often relatively cheap and simple and require little attention. However, it is important that the true value of such systems is appreciated and that unrealistic expectations are avoided. This paper will set out the scope of such systems marking out what can be achieved and that which requires additional steps such as biological treatment.

The main separation technologies

Separation processes can be grouped under three principle headings according to their principal role as set out in Figure 1. These are, screening, settling and refining; for each there is a range of potential equipment varying in cost and performance. Screening processes imply the passage of the slurry through a screen, the solid matter being

retained. The broad principle is thus that of filtration. However, it is not necessarily equivalent to a simple sieving process in that finer particles than the hole size can be retained which is in part due to the fibrous nature of much of the matter removed. Hole size can vary from 1 to 5mm with the inevitable loss of capacity as finer screens are used to retain a higher proportion of the suspended matter. In some cases, finer screens can also lead to a wetter solid product and operational problems. To combat this limitation, more intensive machines have been developed such as screw presses or sieve centrifuges but these also tend to be more costly and offer a relatively low throughput. Various publications can be consulted for details on the many equipment designs available (eg, Burton and Turner, 2003).

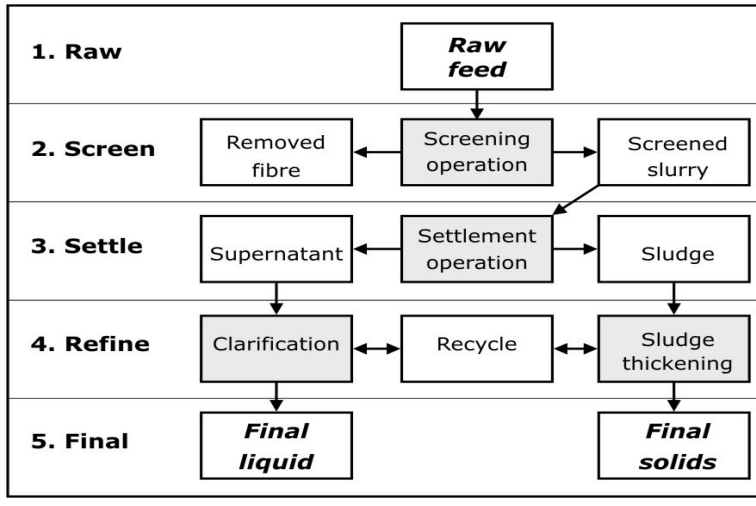

Figure 1. A summary of the main treatment options based on separation. Biological treatments may be included between phase 2 and 3 (eg: aeration) and between phases 4 and 5 (eg: composting).

Settling or sedimentation processes often follow a biological treatment for the very good reason that many natural but degradable surfactants exist in raw slurry that inhibit flocculation and settling (Martinez et al, 1995). The process relies on the higher density of suspended particles over water but for animal slurries, this difference is small. A preliminary screening operation can remove the lighter fibrous material and much of what remains can be settled, even if slowly, in many cases. Adding flocculants or raising the temperature can accelerate the process but the greatest

benefit will result from increasing the forces involved and decanter centrifuges can be expected to produce the best separation.
Separation processes rarely remove both a high proportion of the suspended matter and produce a sludge phase high in dry matter; one objective is usually at the expense of the other. As such, a second step may be included to complete the process. For the sludge phase, this amounts to thickening or drying options, the removed water being sent back to the feed stream. For the liquid phase, the option of a filter press may be considered if the amount of suspended matter is small but these tend to work better if a filter-aid substrate is added; periodic cleaning will be needed. Alternatively, for very dilute effluents, membrane separation with cross flow has been considered but entrained solid and concentrated streams need to be recycled or removed. There also remains issues on high costs, low capacity and the need for frequent cleaning.

Table 1: typical analysis of a livestock slurry (fattening pig in this example) illustrating the main components that characterise the effluent (Williams and Evans, 1981 and Martinez et al, 1995).

Typical analysis (% of total solids)		**Insoluble proportion of component**
Total solids (TS)	100	60 - 80 % depending on manure
Volatile solids	76 - 82	Slightly less soluble than TS
Total suspended solids	82 - 87	Theoretically 100%
Volatile suspended solids	71 - 75	Theoretically 100%
Biological oxygen demand	31 - 35	60 - 80 %
Kjeldahl nitrogen	6.4 - 7.7	30 - 50 % - mostly organic fraction
Organic nitrogen	3.4 - 5.4	Over 80 %
Ammoniacal nitrogen	2.3 – 3.3	Below 10%
Phosphorous (as P)	1.2 – 2.5	20 – 80 % greatly depending on pH
Volatile fatty acids	3.4 – 4.6	Below 10%
Potassium (as K)	1.8 – 3.5	Entirely soluble
Copper	0.1 - 0.2	Above 90% - depends on pH
Zinc	0.1 - 0.2	Above 90% - depends on pH

The scope of separation technologies

Ultimately, the scope of any separation system depends on the solubility of the fraction of concern (or at least its readiness to flocculate).
Typically, this sort of analysis comes down to a series of categories that collectively, characterise the effluent, including nitrogen (subdivided into ammoniacal and organic forms), phosphorous (which can also be divided into mineral and organic forms) and organic matter given as that

degradable biologically in 5 days (BOD_5) or that defined as volatile (either in the suspended or total solid fraction). In addition there are "heavy" metals (eg, copper, zinc), and various salt ions (eg, potassium, chlorine, sodium, calcium). Other special groups include volatile fatty acids (chains of up to 5 carbon atoms but mostly ethanoic or propanoic acids) that are both an indicator of the production of offensive odour (Williams, 1984, Sneath, 1985) and of biogas production, and also total suspended solids not forgetting the total solids content. Table 1 gives some mean values for pig slurry along with some indication of insolubility and thus the potential for removal by physical separation. These figures will vary with manure type but the broad principle remains, that certain parts of the slurry are largely soluble and thus unlikely to be affected by any separation process. This especially applies to the common salts, ammonia, volatile fatty acids and much of the organic matter contributing to the BOD_5 value. For phosphorous and most of the metals (except sodium and potassium) the pH will have a strong influence and the addition of lime in particular will induce a high degree of precipitation. Even if a component is rendered insoluble by the addition of a precipitant, its removal by screening alone may be incomplete as fine particles tend to pass through; a settling processes would thus be implied.

Evaluating the performance of separation options

A crucial consideration with any separation process is that it is also a method of splitting a stream. Thus the apparent removal of 25% of a component from the principal stream into a concentrate representing 25% of the original volume equate to no separation! With this in mind, Martinez et al (1995) endeavoured to define separation efficiency in more meaningful terms by the function $S/F(X_s/X_f-1) \times 100\%$ where S and F are sludge and feed flow rates and X is the related concentration. The ratio $S/F(X_s/X_f)$ thus represents the classical efficiency; subtracting S/F from this leaves the effective separation beyond simple partition. This is a particular useful approach when objectively comparing separator performance in terms of a specific component where there is the production of relatively large volume of the concentrate stream.

Applications of separation in the whole farm model

That the clarification of effluent streams allows easier handling and a reduced impact is a sound enough reason for using separation. However, a biological step will need to be included if ammonia or soluble organic matter is an issue. If the main concern is health risks from pathogens,

then separation will make no difference and storage, biological, thermal or chemical treatment options must be considered. The production of organic products is a second common reason for separation technologies at the farm. These include the feed material for compost schemes (especially from the fibre from screening process) or dried products from sludge production. Monitoring the production of solids from a separator can give an indication of strength thus enabling some degree of process control in a subsequent biological treatment (Burton and Sneath, 1995). Sedimentation processes themselves can be used to both enable the production of a clarified wastewater for cleaning or flushing duties or for the purpose of pre-concentrating a dilute stream ahead of anaerobic digestion.

Conclusion

Separation technologies are the right option if the main purpose is either (a) the improvement of manure handling, (b) the removal of specific insoluble components of the effluent including organic matter, and some of the phosphorous, organic nitrogen, copper and zinc or (c) the preparation of a concentrate to produce a organic fertiliser product. Alone, separation has little effect on pathogens, offensive odour or soluble components including ammoniacal nitrogen. It is noted that by virtue of removing part of the feed stream to a concentrate, a proportion of the soluble components will thus be removed but that this does not necessarily represent an effective separation. Soluble salts including potassium are largely unaffected by any treatment process; although they can by removed by membrane technology, this is not yet a practical option for farm systems.

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Interactions between biomass energy technologies and nutrient and carbon balances at the farm level

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Introduction

Biomass energy is by far the largest renewable energy source in the world (IEA Renewable information (www.iea.org)). Biomass utilisation is closely linked to management and sustainability issues of forestry and agriculture. Carbon is extracted from forests and agriculture to bioenergy facilities, from where it is partly or fully emitted as CO_2 and thus no longer available for sustaining soil organic matter content. Nutrients are extracted as well and, depending of the conversion technology, they may be recycled to farmland or lost as gaseous emissions. Thus, we must be able to describe these effects, and to suggest strategies to alleviate adverse effects on farm sustainability and on the environment. By choosing intelligent combinations of cropping systems and energy conversion technologies, win-win solutions may be achieved.

This paper illustrates, via three cases, some agricultural impacts of choice of biomass technology and describes an intriguing possibility for recycling municipal or industrial wastes through the bioenergy chain.

Biomass energy technologies

A large fraction of current biomass utilisation is simple firewood utilisation in both third world and industrialised economies. However, a range of more or less high technology conversion techniques are now being developed with the aim to improve energy efficiency and introduce biomass energy into the power and transport sectors. Some technologies utilise solid lignocellulosic fuels for thermochemical conversion, while others operate on biological conversion of wet feedstocks. Direct combustion or thermochemical gasification convert all carbon into energy, while biological conversion to ethanol or biogas cannot convert the most stable lignocellulosic biomass components.

So far, biogas systems have recycled these stable carbon components to farmland and thus contributed to sustaining the soil carbon pool. However, current research (Mladenovska *et al.,* 2006) and commercial

trends address subsequent energy conversion of the stable components in order to maximise energy yield and profit. The first case illustrates what this will mean to farm carbon and nutrient balance.

Different manure energy conversion scenarios and their long-term effects on soil carbon and nitrate leaching

A large biogas project involving more than 200 farmers and aiming at converting around 450,000 tonnes of biomass annually is planned for at Maabjerg in Denmark (www.maabjerg-bioenergy.dk). A prerequisite for obtaining good economy in the project has been a permission to combust the remaining fibre fraction after biogas treatment and separation, and this now seems to be in place after the passing of three new bills through the Danish Parliament. However, what will this mean to soil carbon and nitrate leaching at the involved farms? This is illustrated in Table 1 by simple calculations on the different components returned to agriculture with and without biogas treatment and fibre combustion. The calculations are linear interpolations based on simulated scenarios of a crop rotation over a time-span of 50 years by the dynamic farm-model FASSET (Berntsen *et al.*, 2003). Four different combinations of soil type and climate regime were simulated, and the results shown are overall means.

Table 1. Soil carbon changes and nitrate leaching in a crop rotation on a pig farm applying 121 kg N/ha in raw liquid manure or differently treated manure. Effects accumulated over 50 years of a single year with manure application are shown.

	Raw manure	*Biogas treated manure*	*Biogas treated, adjusted application*[1]	*Biogas treated, fibres combusted, adjusted application*[1]
Soil carbon changes *(kg C ha^{-1} y^{-1})*	78	-7	-7[2]	-80
Nitrate leaching *(kg N ha^{-1} y^{-1})*	57	60	54	52

1: mineral fertiliser application reduced equivalently to the increased manure ammonia content

2: a small difference due to adjusted application is likely, but could not be quantified with the applied interpolation method

It appears that biogas treatment could increase nitrate leaching due to a higher content of mineral N than in the raw manure. However, by adjusting the supplemental mineral fertilisation so that total mineral N application is not increased, nitrate leaching will actually decrease. The greatest reduction in nitrate leaching was calculated for sandy soils with high precipitation (not shown). By combusting the fibre fraction a further reduction of nitrate leaching is achieved, but this is accompanied by a significant reduction in soil carbon.

These effects must be included in an overall environmental evaluation of biogas technologies, and they underline the conclusions from Börjesson & Berglund (2006) that a whole fuel-chain evaluation may vary greatly with the choice of raw materials, conversion efficiency and end use technology, and that also indirect impacts on e.g. agricultural systems must be considered.

In order to obtain the maximal energy yield and reduced nitrate leaching resulting from combustion of the fibre fraction from biogas, alternative strategies for sustaining soil carbon content at the farms delivering manure for the energy plant may be sought. One such strategy could be to increase the amount of grass and catch crops in the crop rotation, which would even further reduce nitrate leaching (Simmelsgaard 1998). The grass and catch crops may be harvested for biogas production (Lehtomäki, 2006) and further increase energy production.

Biogas plants for recycling of crop residues and optimisation of farm nutrient balance

Crop residues, such as sugar beet or potato tops, are usually left in the field to decompose. In organic farming grass-clover is often used as green manure to collect nitrogen for the crop rotation. However, these practices are not always very efficient in conserving and recycling nitrogen. The abundance of organic matter and nitrogen in these residues, often decomposing under more or less anaerobic conditions, are conditions that support denitrification. If not denitrified, nitrogen from such easily decomposable residues is prone to leaching during the winter, before the next crop is ready to take up the mineralised nitrogen. An improved practice may be to collect the residues for energy production in biogas plants, which for Sweden has been calculated to have an energy potential of 11 TWh (Svensson *et al.,* 2005).

In a Danish study on energy aspects of organic farming, the effects on yields and nitrate leaching at the farm level of collecting green manure in a biogas plant and subsequently redistributing the nutrients onto the crop rotation were analysed by dynamic modelling with the FASSET farm model (Dalgaard *et al*., 2004). The organic crop rotation on the model farm comprised 20% of the area grown with grass-clover. Two scenarios were analysed where, respectively, half and all the grass-clover was harvested for energy production. Six combinations of soil types and soil organic matter content were analysed.

The collection of green manure for energy production improved modelled yields in the organic crop rotation and at the same time reduced nitrate leaching (Table 2). Albeit differences are small, the effect is interesting since simultaneous improvements in yield and environmental impact are not easily obtained. The highest yield response was calculated for sandy soils with a low organic matter content, while the highest reduction in nitrate leaching was calculated for sandy soils with a high organic matter content (not shown).

Table 2. Modelled dry matter yields and nitrate leaching of organic crop rotations with 10 or 20% of the area used for biogas production (from Dalgaard et al., 2004).

	Base scenario	*Biogas-10%*	*Biogas-20%*
Yield of grain and peas (t ha^{-1})	3.2	3.4	3.5
Nitrate leaching (kg N ha^{-1})	40	38	37

Energy balance calculations in a LCA framework showed that, when utilising 10% of the organic crop rotation for nutrient and energy production, the farm was converted from a net energy consuming to a net energy producing entity. Such practice would be in good accordance with the organic principles of "using, as far as possible, renewable resources in production and processing systems and avoid pollution and waste" (IFOAM, 2002) which, however, so far are not met by organic farming with respect to the use of fossil energy.

Accordingly, a number of positive whole-farm effects can be achieved from biogas energy production from crop residues and green manures. An open question is if such a practise will influence significantly soil carbon contents. The bottlenecks for introducing such practices are to find

appropriate technologies and to achieve a positive economic return (Svensson *et al*., 2005). The economic return is highly influenced by local policies on renewable energy. Accordingly, there is a significant increase in biogas production in e.g. Germany and Austria (Amon et al., this issue), where high prices for biogas electricity are resolved on, while in Denmark the establishment of biogas plants has stopped due to economically unfavourable conditions.

Energy crops for double-loop recycling of municipal wastewater and sludge

In Sweden several examples exist of recycling municipal wastewater and sludge in commercial willow plantations, and approx. 10% of the biosolids from Swedish sewage plants are utilised in willow. At the city of Enköping, decanted water from dewatering of sewage sludge is distributed through 350 km drip irrigation pipes in an 80 ha willow plantation (www.enae.se). The water contains approx. 25% of the total N load of the wastewater treatment plant, which is applied to the willow plantation at a rate of approx. 250 kg N/ha accompanied by approx. 7 kg P/ha. This practice saves investments and management costs at the wastewater plant and increases growth of the willow crop due to the unlimited water and nitrogen availability. The willow is combusted at the local combined heat and power (CHP) plant, and the nutrients in the bottom ash are recycled to willow plantations.

In this way nutrients from wastewater are recycled to agriculture in a double-loop system via energy crops that are not as sensitive to risks of contamination with pathogens or chemical substances as are food or feed crops. Heavy metals, especially cadmium, are taken up quite efficiently by willow (Dimitriou *et al*., 2006) and are concentrated in the fly ash fraction at the CHP plant, from where they may be extracted or the fly ash may be deposited. However, nitrogen taken up in the woody part of willow and combusted at the CHP is lost to the atmosphere as free nitrogen. If, alternatively, the biomass produced on the basis of wastewater and sludge was converted in biogas plants, also the nitrogen may subsequently be redistributed to agricultural land for food production. Willow, if not carefully pre-treated, is not suited for biogas production, but also grass crops may be irrigated with wastewater and thus facilitate double-loop recycling of nitrogen into agriculture (Geber, 2000).

Soil carbon content is likely to be sustained or even increased, when perennial energy crops are grown (Hansen, 1993; Hansen *et al.,* 2004). The lack of soil tillage is expected to reduce soil mineralization, and even though the whole crops are harvested for energy, substantial organic matter losses from roots and leaves feed the soil carbon pool. However, there is a need for more thorough analysis of these aspects.

Recycling of organic waste products to perennial energy crops can take place with very low nitrate losses to the aquatic environment (Geber, 2000; Jørgensen, 2005) ensuring that recycling does not conflict with the increasing environmental demands, e.g. as stipulated in the European Water Framework Directive. However, factors of concern that are under investigation or should be investigated are: is there a risk of pathogen dispersal when distributing wastewater onto energy crops (Carlander, 2000)? And, is nitrous oxide formed in the denitrification processes that undoubtedly occur at such high levels of water and nitrogen application? These open questions, along with lack of clear-cut organisational models for the concept, are amongst the barriers that hamper a more widespread distribution of a potential win-win solution (Borjesson & Berndes, 2006).

Conclusions

Biomass energy technologies affect both carbon and nutrient balances at the farm level. This can be positively exploited to reduce nitrate leaching along with increasing energy production from biomass. Also, bioenergy crops can be used for recycling of wastewater with low losses to the environment. Accordingly, more political goals can be pursued within the same projects, helping to fulfil both biomass energy goals and e.g. the European Water Framework Directive. The intriguing question is whether simultaneously soil carbon contents can be sustained ensuring high overall greenhouse gas abatement from bioenergy systems and fulfilment of the Kyoto Protocol. It is likely that this is possible as well, if farm crop rotations are carefully manipulated. Inclusion of more perennial crops and/or catch crops will positively affect soil organic matter content.

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Biosecurity and arable use of manure and biowaste

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Abstract
The potential negative environmental impact of manure and biological waste can be minimised at a profit by recycling plant nutrients in the food chain. Current large-scale livestock production, epizootic diseases and increasing globalisation increase the need for biosecurity. Arable use of manure and biowaste can inadvertently spread infectious diseases; opinion differs concerning the risk levels. To obtain general acceptance for this use, a hygienically safe end-product is needed. Composting, anaerobic digestion and ammonia treatment are options further discussed. Suitable treatment methods must combine biosecurity aspects with environmental, economic and nutrient recycling aspects to create a beneficial whole-farm approach.

Health risks when using manure and biowaste as a fertiliser
Pathogens. Enormous numbers of species and subtypes of bacteria, viruses and parasites are found in manure and in biological waste (BW) from households, food industry, restaurants, toilets etc. Some are disease transmitters (pathogens) and some pass between humans and animals (zoonoses), i.e. salmonella and EHEC (enterohaemorrhagic *E. coli*). Pathogens may cause epizootic diseases e.g. classical swine fever. In developing countries, infectious diseases of both animals and man are more frequent than elsewhere, causing a heavy load of pathogens in manure and BW. Parasitic diseases are of special interest here.

Unwanted organic material such as hormones and antibiotics may exist in manure and BW. Antibiotic-resistant bacteria may end up in the environment where their resistance genes can spread to better-adapted indigenous bacteria, increasing the resistance reservoir (Kühn et al. 2005). Unwanted organics in soil can affect plant growth and in some cases be taken up by plants. The higher density of microorganisms in soil than in water results in higher degradation of unwanted organics in the soil, so the main effect observed of organics is on aquatic life, e.g. reproductive disorders in fish (Sumpter and Johnsson 2005).

Dissemination of pathogens

On-farm spread may occur via storage, transport and use of manure. Further spread may occur from manured land via surface runoff, leakage to groundwater, dust particles and harvested crops. Animals kept outdoors on frozen land in winter e.g. horses or organic livestock, increase surface runoff. Grazing animals can transmit pathogens directly to other animals and to the environment.

To-farm spread may occur as described above from neighbouring farms or via vector animal such as birds, rodents or insects. Infections may also be introduced via incoming live animals, feedstuff, equipment, manure and BW, etc. Humans may spread pathogens, i.e. if toilet waste is added to slurry tanks. In high-density livestock areas, excess manure may have to be transported to other regions, a practice involving a considerable increase in biosecurity risk as diseases not indigenous in a region may be introduced. In addition, use of BW from society creates new routes of disease transmission between animals, humans and the environment.

Treatment of manure and BW

Manure treatment differs according to tradition and local conditions. In general, larger farms have more opportunities. Farms located close to urban areas may be forced to treat manure, mainly to decrease the smell. In Europe, more than 65% of livestock manure is handled as slurry, a mixture of urine, faeces, water and bedding (Menzi 2002). Slurry is relatively convenient to handle, but sanitation during storage is not sufficient (Himathongkham et al. 1999). Some pathogens can persist in slurry for a long time. Some bacteria (Salmonella, EHEC) can multiply significantly if conditions are favourable, e.g. as regards nutrient availability, so it is important that no material is added during storage. Levels of indicator organisms vary over time, indicating that pathogen levels may follow a similar pattern. A sufficiently long period of storage without adding fresh material is generally impossible, since storage capacity is usually limited. Traditional outdoor manure storage results in unwanted emissions of ammonia and methane.

If available farmland is not already heavily loaded with manure, BW from society may be recycled as a fertiliser, a solution that can profit both the farmer and society. Three different treatment methods for producing

hygienically safe end products are described below, methods which offer opportunities to co-treat manure with BW.

EC legislation (1774/2002) strictly regulates the treatment of BW if it includes animal by-products (ABP) or manure. Manure for sale has to be sterilised, however, several exceptions exist. Category 3 ABP (i.e. low-risk slaughterhouse waste) may be recycled, if separate pasteurisation at 70°C for 60 min. is combined with other treatment. Additional legislation 208/2006 to be implemented in 2007 permits alternative treatments to pasteurisation, once individual member states have validated that such treatments have an equivalent hygiene effect. However, validation of alternative treatment methods in a scientifically based, generally accepted way is currently impossible.

The effectiveness of a sanitation treatment depends on its temperature, duration, pH, volatile fatty acids, oxygen availability and other factors. Treatment goals can also vary depending on the origin of the manure and BW and the potential use of the end-product. Pasteurisation at 70°C for 60 min gives a sufficient reduction in pathogens (Sahlström et al. unpubl.; Mitscherlich and Marth 1984). However, the reduction in heat-resistant viruses is limited, and spore-forming bacteria and prions are not reduced at all. The treatment effect has to be continuously monitored by checking the process (i.e. temperature, pH-value, treatment time) and the end-product (i.e. indicator organisms or pathogens). Types of checks performed and frequency and number of analyses of the end-product may vary for different kinds of BW and processes. In large volumes of material, sample distribution and sampling technique are essential in obtaining an accurate hygiene assessment, but such sampling procedures have not yet been standardised. As an extra safety precaution, the use of a particular end-product on farmland may be restricted, or there may be a quarantine period between spread of the end-product and crop harvest/grazing.

Composting. In the UK, France and Eastern Europe, more than 50% of manure is handled in solid form (Menzi 2002), making composting convenient. Slurry may also be successfully composted by forced aeration, or following liquid-solid separation using mechanical methods and/or polymer flocculation. Composting can give acceptable hygiene quality in the end-product if most of the material achieves sufficiently high temperature (Kjellberg Christensen et al. 2002), which requires the compost to be repeatedly turned and thoroughly mixed. Incorporation of

structural material may be needed, plus an insulation layer above and below the compost. More technically advanced practices, such as preheating of incoming air, can also increase the temperature. Stabilisation of the treated material minimises the risk of pathogen re-growth. The main environmental concern is that most of the ammonia released during degradation of organic material will be lost as an acidifying or eutrophying emission during composting. The high target treatment temperature increases this effect. Stabilisation decreases the risk of methane emission from the end-product. Controlled reactor composting offers possibilities to minimise gaseous emissions via condensing or bio-filter treatment of the outgoing gas.

Anaerobic digestion at farm-scale has a long history in Asia, but in Europe fuel shortages during the World War II and thereafter was the driving force (Köttner 1999). Manure has been used as the main substrate. The interest in farm-scale biogas plants (BGP) is growing in some EC countries, e.g. Denmark and Germany (Al Seadi and Holm-Nielsen 1999; Köttner 1999) and also in several developing countries. Co-digesting with BW is one of the main ideas for large-scale centralised BGPs, but is also practised by some farm-scale BGPs. Large-scale BGPs are increasing in numbers in many countries (15 in Sweden at present). If a pasteurisation step is not used, BGPs have to rely on sanitation in the digestion chamber (Sahlström 2002). Most large-scale BGPs use a continuous process, which is less reliable regarding sanitation. Thermophilic digestion (50-58°C) of sewage sludge in a large-scale continuous process reduces indicator bacteria and salmonella sufficiently, while mesophilic digestion (30-38°C) is unreliable (Sahlström et al. 2004). Pasteurisation of substrate in a separate batch-wise step prior to digestion is a reliable treatment, but recontamination of digested residues can occur post digestion (Bagge et al. 2005). By using a process adapted for high ammonia content (8 g L^{-1}) at a pH close to 8, it is possible to have a sanitising mesophilic process (unpublished data). Maintaining high ammonia levels requires restricted feeding with a high protein diet. Evaluation of biogas use shows a low pathogen risk (Vinnerås et al. 2006).

Anaerobic digestion is a complex system with environmental benefits, as valuable energy in the form of biogas is produced. However, depending on the system design, large amounts of methane can be emitted during the process and road transport to/from large-scale BGPs may be considerable.

Ammonia treatment both stabilises and sanitises manure and BW. The sanitation effect is achieved at considerably lower pH (9-10) than regular treatment with bases (Allievi et al. 1994). Increased temperature and pH shift the ionic ammonia to uncharged ammonia, which gives the microbicidal effect. Sanitation requires a closed treatment system, e.g. roofed slurry tanks, otherwise the ammonia is lost as gaseous emissions. Ammonia is added either as aqueous ammonia solution or as granulated urea. The treatment is efficient for inactivation of bacteria, parasites and some viruses. The reduction of single-stranded RNA viruses such as enteroviruses is effective (Ward 1978), but double-stranded viruses (e.g. rotavirus) are relatively resistant to ammonia, as to most other treatments. Recommended treatment of manure is either 0.5% NH_3 (aq.) for one week or 2% urea for two weeks at temperatures above 10°C, or for one month at temperatures below 10°C (Ottoson et al. unpubl.). The environmental and economic cost of ammonia treatment is low, as the ammonia used can be recycled as a fertiliser.

Survival of pathogens after land application

Both the survival and growth potential of pathogens vary considerably between different species and subtypes (Mitscherlich and Marth 1984). Natural inactivation factors also vary considerably due to climate, season, vegetation, soil type, etc. (Cools et al. 2001; Nicholson et al. 2004). Method of application to land is important too, as ploughing-in or injection reduce pathogen spread and animal exposure, but persistence may be prolonged within soil compared to at the surface. The reliability of natural inactivation factors on plant surfaces, in soil and in feed and foodstuffs should not be overestimated. Pathogens may persist in the environment for very long periods, several decades for spore-forming bacteria (Mitscherlich and Marth 1984). Survival of pathogens in soil, grass and silage for close to 2 months has been shown under laboratory conditions (Johansson et al. 2005). Wild animals may acquire a pathogen and then act as a disease reservoir without displaying clinical symptoms, e.g. wild boars in Eastern Europe have transmitted swine fever back to domestic pigs. On the other hand, pathogen pollution may cause infection of immunologically naive wildlife and markedly reduce whole populations.

Conclusions

The potential health risks associated with plant nutrient recycling in the food-chain must not be ignored. More effective manure and BW

management can prevent ecosystem contamination and dissemination of pathogens, while the use of artificial fertilisers may be reduced.

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The present status of rapid methods for on-farm analysis of manure composition with emphasis on N and P: What is available and what is lacking?

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Biomass materials represent both potential sources of crop nutrients, energy or environmental pollutants, the potential of which varies with its nature and use; the only way to maximize the benefits and minimize deleterious effects is to know its composition. Due to their heterogeneous nature, and frequent presence of volatile constituents, optimal utilization requires rapid, accurate and timely analysis; something difficult to achieve off site; thus, the increased interest in on-farm testing. Presently, the US emphasis is on minimizing excess nutrients from manures, specifically N and P. However, on-farm testing methods could be easily adapted to other materials including: farm products, soil composition, animal health, household wastes or industrial biomass materials. A term often used for on-farm tests is "Quick Tests" (QT). The analytes of interest for manures are: NH_3- and organic (Org) -N, the mineralization potential (MP) of the Org-N, and plant available- and total-P. For C-sequestration, household wastes, many farm products and biomass materials, fiber, protein and moisture would be of interest.

For traditional QT, the most methods [Agros and Quantofix meters (reaction of bleach with ammonia), electrical conductivity (EC), hydrometer and Reflectometer] are available for NH_3-N (Van Kessel and Reeves, 2000; Van Kessel et al., 1999). The Reflectometer is a small portable spectrometer which reads test strips (NH_3-N, soluble-P). QT measurement of NH_3-N were found to be quite good using several methods, while those for Org-N were not, for 99 dairy manures from the NE US (Table 1). Similar results were obtained for poultry manures except that EC did not perform well for near surface samples due to the use of $NaHSO_4$ in the poultry houses to control NH_3 (Reeves et al, 2002). The Quantofix and Agros meters both rely on the reaction of Cl and NH_3 to produce N_2. While the Quantofix uses liquid, the Agros uses solid $Ca(ClO)_2$ which reacts to a greater degree with Org-N. Similar results for NH_3- or Org-N should be expected for other biomasses with these QT.

No P QT have shown results comparable to those for NH_3 (Lugo-Ospina et al, 2005). Even the Reflectometer did not match laboratory results based on the same chemistry. The best QT results (R^2 of .66) were for a combination of Reflectometer and pH. While some advocate using EC for P, we have not found this to be useful for poultry or dairy manures. The results with the hydrometer for N, and EC for P, raise the question of the value/danger of surrogate calibrations. Hydrometers are sold that are calibrated for N. This is based on assumed relationships between specific gravity and N content (Van Kessel et al, 1999) which is possible only with consistent rations, etc. The manures (Table 1) were obtained from 107 farms in 5 different states and thus were quite diverse (bedding, handling systems, etc.) resulting in no such relationships. With EC, the same principle holds, as H^+ or OH^- are the principle determinates of EC, thus the poor results for NH_3 for poultry manures after treatment with $NaHSO_4$.

Table 1. Summary of results for determination of ammonia and total N by traditional quick tests (99 dairy manure samples from NE USA).

	Ammonia		Total-N	
Quick Test	R^2	RMSD*	R^2	RMSD
Agros Meter	0.785	0.034	0.779	0.068
Hydrometer	0.175	0.066	0.395	0.11
Con. Meter	0.856	0.028	0.699	0.079
Con. Pen	0.840	0.029	0.697	0.080
Quantofix	0.939	0.018	0.581	0.093
Reflect.	0.900	0.023	0.688	0.081

*RMSD = Root Mean Squared Deviation = (Sum Squared Residuals / 99)$^{1/2}$.

While traditional QT methods rely on chemical reactions or surrogate relationships to determine manure N or P, spectroscopic methods; from visible (400-700 nm), through near-NIR and NIR (700-1100 and 1100 to 2500 nm, resp.), to the mid-IR (2500-25,000 nm); are based on the spectral signatures of components of interest and do not require the use of chemical reactions or extractions once calibrations relating the spectral signature to the component of interest have been developed. They have become a primary means to analyze the composition of agricultural products. Until recently, this work has primarily involved either the visible/shortwave NIR (400 to 1100 nm) alone or in combination with the NIR (1100 to 2500 nm) and been used to analyze animal feeds. With the increase in interest in compositional determinations of manures, biomass, soil, etc., the use of NIR spectroscopy (NIRS) has logically followed. More recently mid-IR in the form of diffuse reflectance mid-IR Fourier transform spectroscopy (DRIFTS) for analysis has gained interest. Regardless of the spectral region, various methods (e.g., stepwise regression and others)

are used to develop calibrations relating the spectral information to analyte levels. For on-site use, one of the primary advantages of NIRS or DRIFTS is that, once calibrations are developed, one can simultaneously determine multiple analytes for any sample, and the same instrument can be used for many different types of samples.

NIRS using a large sample transport accurately determined NH_3- and Org-N in dairy manure (Table 2), however even with such a large sample, heterogeneity resulted in variation, but the results were still more accurate than obtained by the traditional QT, esp. for Org-N (Reeves and Van Kessel, 2000a). With a fiber-optic NIR unit wavelengths up to at least 2300 nm were needed for the accurate determination of NH_3-N, and even then were not as good as in Table 2 (Reeves and Van Kessel 2000b). This is important because many portable NIR spectrometers use fiber-optic probes, and many non-fiber-optic NIR instruments do not scan beyond 1800 nm. NIRS results for poultry manure were similar to those for dairy (Reeves, 2001).

Table 2. Results using 3 reps in developing calibrations (n=99) for dairy manures and large sample transport.

	1-out X validation		Calibration	
Assay	R^2 range	RMSD range	R^2 range	RMSD range
NH_3-N	0.908 - 0.927	0.020 - 0.022	0.956 - 0.958	0.015 - 0.015
Moisture	0.874 - 0.892	1.44 - 1.56	0.922 - 0.938	1.09 - 1.22
Total C	0.886 - 0.901	0.57 - 0.61	0.939 - 0.960	0.376 - 0.44
Total N	0.902 - 0.911	0.044 - 0.045	0.944 - 0.964	0.027 - 0.034

RMSD = Root Mean Squared Deviation = (Sum Squared Residuals / 99)$^{1/2}$.

NIRS and DRIFTS results for P with either dairy or poultry manure, dried or non-dried, using a variety of P measures including, total, soluble and phytate have failed to produce satisfactory results with R^2 generally well below .5. While forms of P may be spectrally featureless in the NIR, this is not true for the mid-IR. Examination of DRIFTS spectra of various metal phytates and inorganic phosphates indicates that one problem may be that each different salt has a slightly different spectrum. A possible confounding factor is that the traditional extraction methods used to determine P in manures may only determine varying percentages of any specific form of P present depending on the metal salts present and sample matrix. Interestingly, it has been reported that NIRS can accurately determine P and K in swine manure but K salts have no NIR spectrum (Malley and Vandenbyllaardt, 1999). The most likely basis for

this success is surrogate correlations with organic components. While potentially useful, great care must be exercised as sample changes not related to the analyte can result in erroneous results.

Efforts to find a NIRS or DRIFTS method to estimate the MP of the Org-N have not been successful. The determination of mineralizable-N requires incubations with soil for various periods of time or for a set period of time. Experiments have demonstrated that different components of manure, as expected, mineralize at different rates and that the composition of manure varies greatly beyond simply its NH_3 or Org-N (Van Kessel et 2000). NIRS or DRIFTS may see the entire components and not discern the difference between the fractional %'s mineralized in a growing season versus a year.

While several traditional QT and NIRS accurately determine NH_3-N and can be easily used on-farm, only EC and NIRS can be easily adapted for in-line testing. NIRS, also has the advantage of being able to determine many analytes in many materials whereas other QT determine a single analyte. One caveat with fiber measurements in manures, and perhaps other biomass materials, is that many dairy manures have been found to have very high ash contents (8-52%, mean 21% for 99 dairy manures), the nature of which and effects on traditional fiber value determination is unknown (Reeves and Van Kessel 2002). Until recently, spectroscopic analyses were performed in the laboratory, due to the cost, size and fragile nature of the spectrometers. Recently, field portable spectrometers have become available such as the SOC-400 (mid-IR) and ASD AgriSpec (NIR). However, these instruments are still too expensive for routine purchase by each farmer with prices generally in the $50,000 US range. While it may be difficult to greatly reduce the price of portable instruments based on traditional technologies, there are newer technologies, such as the spectrometer on the chip concept (AXSUN Technologies, www.asxum.com), micro-moveable mirrors (MMM) and others which may greatly reduce the cost in the future if demand increases.

Due to costs, time requirements**,** environmental regulations, etc. it would appear that the future is on-farm testing. On-farm testing for: forage and grain quality (fiber, protein, moisture), manure composition (inorganic- and Org-N and P, available P, minerals), milk composition (fat and protein) and mastitis, and soil C for sequestration payments would seem to be desirable. It is easy to see the impracticality if a unique QT is used

for each determination. Therefore, spectroscopic methods such as NIRS or DRIFTS are the way of the future. Finally, while results with soils have shown DRIFTS to be more accurate and robust than NIRS, NIRS is presently more adaptable for on-farm analysis. With the rapid changes occurring in technology it would appear that a portable NIR instrument capable of measuring most of the analytes of interest for on-farm analysis will be available at an acceptable price in the near future.

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Reducing ammonia volatilisation from pig slurry through the reduction of dietary crude protein and the incorporation of benzoic acid

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Several animal feed additives are authorised in the European Union as acidity regulators; some of them have potential benefits beyond the digestive effects, including the reduction of ammonia emissions from the effluents subsequently produced. Benzoic acid (C_6H_5COOH) is only provisionally authorised from May 2003, and for a period of four years. After being absorbed in the small intestine and combined with glycine, benzoic acid is transformed into hippuric acid ($C_6H_5CONHCH_2COOH$) and then eliminated through the kidneys, as a consequence decreasing the pH of urine. Reduction of dietary protein content is also known to reduce urinary pH and ammonia volatilization. The aim of this study was to evaluate the combined effects of reduced protein supply and supplementation with benzoic acid on ammonia volatilization.

Sixteen fattening pigs of about 35 kg were fed with 4 experimental diets, which differed in their crude protein content (13 or 18%), and in the addition of benzoic acid (0 or 1%). The duration of the feeding experiment was 28 days, in witch 10 days were spent for the adaptation of animals. All the excreta from the 4 pigs in each dietary treatment were collected over two consecutive 9-day periods. During the first period the weight and composition of urine and faeces, separately collected, were determined, including urinary pH. During the second period, after been weighed, the excreta were mixed to constitute a slurry representative of each diet. Slurries were stored at 4°C for a few days before the test of ammonia volatilisation. Two samples (five kilograms each) of slurry from the 4 diets were placed in glass cells on a laboratory pilot scale system designed to measure ammonia emission; this equipment allowed the simultaneous measurement of ammonia emission from 12 cells maximum. Changes in slurry composition were determined by mass balance calculations, and ammonia emission was measured during two storage trials of 95 and 238 hours duration. Ammonia was trapped by bubbling blowing air in acid.

Potential ammonia volatilisation after spreading was measured using a wind tunnel technique only for the slurries from the two 13% crude protein diets, with or without benzoic acid supplementation (3 replicates of five days each).

Main fattening results are presented in Table 1. The weight gain was higher for the 18% crude protein diet; the incorporation of benzoic acid did not influence the performance of animals.

Table 1. Fattening results of growing pigs fed the four diets.

Diet	1	2	3	4
Crude Protein (%)	13	18	13	18
Benzoic Acid (%)	0	0	1	1
Weight at start (kg)	42.8	44.1	43.2	43.0
End weight (kg)	60.5	63.3	61.7	62.4
Weight gain ($g.kg^{-1}$)	982	1069	1023	1075
Feed Conversion ($kg.kg^{-1}$)	2.17	2.00	2.08	1.98

During the collecting period, the quantities of excreta were not clearly influenced by the diet, even though high protein level feed gave higher volumes of slurry. Urinary pH was higher for the high protein diet (9.25 vs. 8.66, $P < 0.002$), and was lower for animals fed with diet with 1% of benzoic acid (8.78 vs. 9.19, $P < 0,002$). The effect of benzoic acid on urinary pH was higher for the diet with 13% crude protein content.

Slurry pH at the end of the collecting period was lower for low protein diets (7.83 vs. 8.70) and for diets containing benzoic acid (8.03 vs. 8.50). The lowest pH was 7.48 for slurry from the diet with low protein and 1% benzoic acid, and the highest pH was 8.82 for slurry from diet with high protein and 0% benzoic acid. Table 2 shows the pH value of the slurries at the start and at the end of the two ammonia emission trials of short storage. Initial pH values varied between the two trials for same slurry even so stored at 4°C during one week.

The evolutions of nitrogen (Nk: Kjeldahl Nitrogen) for all the storage experiments are presented in Table 3. Nitrogen losses were more or less twice as high for the 10 days monitoring trial (238 hours) compared to the 4 days monitoring trial.

Table 2. Initial and final pH values of slurries from the 4 diets during the 2 trials of short storage.

Benzoic Acid	Crude Protein (%) 18			Crude Protein (%) 13		
	Diet N°	Initial pH	Final pH	Diet N°	Initial pH	Final pH
0 %	R2			R1		
1st Trial		8.81	8.48		8.37	7.85
2nd Trial		8.73	7.58		8.44	7.53
1 %	R4			R3		
1st Trial		8.58	8.05		7.57	7.25
2nd Trial		8.73	7.94		7.62	6.77

Table 3. Evolution of N stocks during the trials.

	Initial Nk content (g N)	Losses of Nk in the cell (g N)(*1*)	Total N-NH_3 trapped (g N)(*2*)	Yield (%) of trapping (*1/2*)
R1 trial 1	18.37	1.55	1.03	67
R1 trial 2	21.50	3.55	2.42	68
R2 trial 1	25.74	1.95	1.86	95
R2 trial 2	27.73	4.83	3.90	81
R3 trial 1	17.12	0.79	0.65	83
R3 trial 2	16.23	1.92	1.52	79
R4 trial 1	25.77	1.59	1.34	84
R4 trial 2	25.16	3.52	3.31	94

The analysis of the emission rate calculated from 8 trapping sequences in the 2nd trial showed that concentrations of ammonia in air were close to constant after 7 days and were around 20, 26, 42 and 49 mg NH_3-N.m^3 air, respectively, for diets R3, R1, R4, and R2. Nevertheless, the cumulative curves of NH_3-N emission at 10 days did not present tendencies of diminishing rates. Relative to the R2 diet, a reduction of NH_3 emissions during storage of 60% could be achieved by reducing the feed protein level to 13% and adding 1% of benzoic acid, as in diet R3.
Only stored slurries from R3 and R4 used in 2nd trial were field spread with subsequent monitoring of ammonia emission. No NH_3 emission could be measured during each of 3 five days replicate periods.

Diet influence on ammonia emissions in lactating dairy cows

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Animal husbandry is well recognised as the major contributor of ammonia to the atmosphere, which leads to NH_3 deposition elsewhere. In Europe, livestock production is the dominant source (70 to 90% of total emissions) followed by application of mineral N-fertilizers (up to 20% of total). In the area of study, intensive dairy cattle production is responsible for most environmental problems related to farming activities. In dairy cattle, feeding strategies are one of the measures to achieve mitigation of NH_3 emission.

There are previous works on the effect of changing the forage-to-concentrate ratio on N intake and excretion, but few studies include environmental effects of the excreta produced. This study collected urine and faeces from two different forage-to-concentrate ratios of one cross-over experiment. The effects of these diets on N intake, milk urea, N and urinary urea-N excretion, and on ammonia volatilization from the slurry applied to soil were studied.

Description of methodologies

This work was conducted in a cut grassland in the Basque Country (northern Spain). Treatments were as follows: Control (C), Slurry from cows fed a high forage-to-concentrate ratio (denoted diet A) and Slurry from cows on a lower forage-to-concentrate ratio (denoted diet B). The description of diets and average feed consumption is presented in Table 1.

Table 1. Diets fed to dairy cattle from which fresh slurry was obtained.

Composition (kg MS)	Diet A	Diet B
Forage:Concentrate	2.9	1.2
Triticale silage	11.7	9.4
Lucerne	1.75	1.75

Slurry was applied to the soil at a standard rate of 120 kg N ha^{-1}. Characteristics of the slurries are shown in Table 2. Faeces and urine for analysis were collected separately from the cows while in the stalls and immediately kept in containers. Faeces and urine volumes were estimated using CNCPS 5.0.

Table 2. Characteristics of slurries applied to soil and derived from urine and faeces collected from lactating dairy cows fed different diets.

Treatment	N total (% w/dw)	NH_4^+-N (% w/fw)	C:N	pH
Diet A	1.9a	0.11a	17.0a	7.74a
Diet B	2.07a	0.16a	15.3a	7.02a

Ammonia emissions were measured using an open chamber technique. Concentrations of NH_3 were measured at the air inlet and outlet of the chamber using a photoacoustic infrared gas analyzer. Emissions were continuously measured during three days from the day fertilizers were applied. Measurements were made at 2, 8 and 19 hours from fertilizer application on the first day.

Main results

Although the majority of urea is excreted in the urine, some diffuses into the milk. Milk urea has been reported to be affected by the ratio between energy and protein (Gustafsson and Carlson, 1993) and forage-to-concentrate ratio (Godden et al., 2001). Therefore, milk urea was measured in order to be used as an index of the adequacy of the energy and N intake with regard to the requirements in dairy cows (Table 3). Milk urea values were within the normal range in relation to diet protein deficiency or excess, being no significantly different between treatments.

Slurry applications to soil were made based on NH_4-N concentration. Thus, as slurry from diet A had a lower ammonium content, the amount of slurry applied per square meter was higher in this treatment. During the trial, urine N concentration in diet B was significantly higher than in diet A (Table 3).

Table 3. N excretion in milk, urine and faeces. Values are the mean for the four cows on each diet.

	Diet A	Diet B
Urine (total N), g/d	80.1b	115.9a
Faeces (total N), g/d	107.5a	105.2a
Urine urea-N (g/d)	68.7a	87.1a
Milk urea N	7.7a	8.2a

Once slurry was applied, NH_3 emissions from both treatments peaked approximately after 2 hours, and at least 81% and 97% of the total ammonia measured in diet A and B respectively was lost within 40 hours (Figure 1).

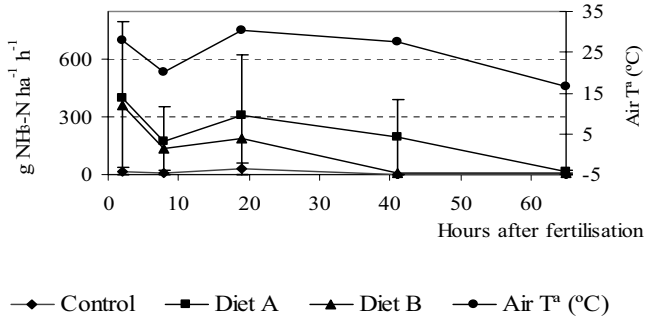

Figure 1. Pattern of ammonia emissions (g NH_3-N ha^{-1} h^{-1}) from slurry applied to soil. Each value is the mean of three replicates.

Cummulative ammonia emissions during the 65 h period was 10.1 and 4.7% of the ammoniacal N applied for diets A and B respectively, although statistically not significant due to the great variability found.

Conclusion

It is concluded that at 500 g N d^{-1} intake, by changing the forage-to-concentrate ratio, N output is significantly decreased only in urine. This affects the N composition of the slurry derived, allowing to apply higher amounts of slurry to reach the same N application at the higher forage-to-

concentrate ratio. Ammonia emissions were not significantly different between treatments due to the high variability found.

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Modelling methane emission from dairy cows

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Methane is produced by fermentation of feed in the digestive tract of ruminants and represents an inevitable energy loss from the animals. The methane emission from an average cow is in the order of 350 g daily, corresponding to approximately 70 000 tons per year from the 550 000 cows in Denmark. The proportion of gross energy (GE) of the feed which is lost as methane energy is about 6%, but this proportion is affected by feeding level and chemical composition of the feed.

The Nordic cow model Karoline is a dynamic, mechanistic whole animal simulation model described and evaluated by Danfær et al. (2006a, 2006b). In the present paper, Karoline was used to predict methane production from dairy cows in response to changes in feed intake, concentrate level in the diet, silage digestibility, level of dietary fat, and

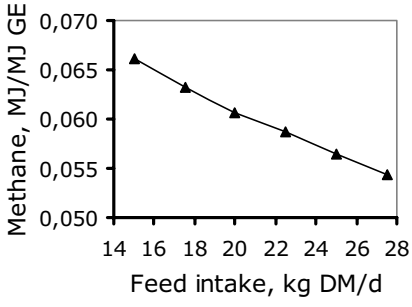

Fig. 1. Simulated effect of feed intake on methane production.

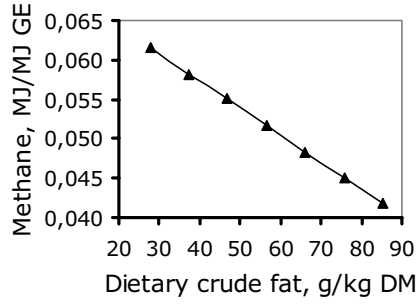

Fig. 2. Simulated effect of dietary fat on methane production.

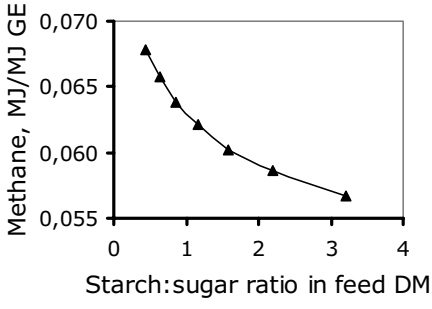

Fig. 3. Simulated effect of starch:sugar ratio on methane production. Starch+sugar: 43% of feed DM.

starch:sugar ratio in the diet. The simulations showed that the methane energy loss (expressed as percentage of GE) decreased with increasing feed intake (Fig. 1), decreased with high proportions of concentrates in the diet, and increased with increasing digestibility of grass silage (not shown here). Moreover, the methane production was reduced by increasing dietary fat (Fig. 2), and by increasing starch at the expense of sugar in the diet (Fig. 3). All these model predictions were in agreement with corresponding experimental results (e.g. Kirchgessner et al., 1994; Yan et al., 2000; Giger-Reverdin et al., 2003).

During a 10 year period (1991-2002), the composition of winter-feed rations for dairy cows in Denmark has changed, so that the proportion of fodder beets + beet pulp has decreased from 37.5 to 16.0%, while the proportion of maize silage has increased from almost zero to 21.8% on a dry matter (DM) basis (Weisbjerg et al., 2005). These changes have affected the chemical composition of the feed, i.e. a decrease in the sugar content from 20.0 to 8.5% and an increase in the starch content from 7.6 to 15.1% of feed DM (see Table 1). The estimated feeding level has increased from 18.2 to 19.5 kg DM per cow daily during the same period.

Table 1. Chemical composition (g/kg DM) of the winter-feed for dairy cows in 1991 and 2002.

Chemical fraction [1)]	Winter 1991	Winter 2002
Crude protein	166	167
Crude fat	43	48
Sugar	200	85
Starch	76	151
Cell wall carbohydrates	426	467

[1)] Calculated by Weisbjerg et al. (2005) based on reports from The Danish Cattle Organization.

These changes would suggest that the methane production from dairy cows in Denmark decreased between 1991 and 2002. In order to examine this question, the methane production during two winter periods, 1991/92 and 2002/03, was estimated by simulations with the Karoline model as shown in Table 2. The model inputs were based on information on the composition of the winter-feed rations for dairy cows in these two years (Weisbjerg et al., 2005).

Table 2. Methane production in dairy cows during winter periods 1991 and 2002 predicted with the Karoline model.

Year	1991	2002
Feed intake, kg DM per cow daily	18.20	19.49
(A) Methane, g per cow daily	387	377
(B) Methane energy, % of GE	6.7	6.0
Decrease in (B) 1991-2002, %		10.4

The simulations showed a decrease in methane production from 1991/92 to 2002/03, not only on an energy basis as a percentage of GE, but also in absolute quantities in spite of an increased feed intake. In a previous paper (Danfær, 2005), these results were compared with corresponding predictions with three different empirical regression models (Kirchgessner et al., 1994; Johnson and Ward, 1996; Hindrichsen et al., 2004). These models also predicted a decrease in methane production from 1991 to 2002. The calculated decline in methane energy loss as a percentage of GE varied among the three models from 6.0 to 22.1% with a mean value of 12.7%.

It is concluded that the enteric methane loss (as per cent of GE) from dairy cows in Denmark during the winter feeding period is likely to have decreased by approximately 10% from 1991 to 2002 as a result of changes in feed composition and feeding level. This corresponds to a 5-6% decrease on a yearly basis assuming a winter feeding period of 200 days. It is further concluded that the model Karoline is a useful tool for simulation of nutrient digestion, utilisation and excretion in lactating dairy cows. Parameters like milk yield, live weight gain, emission of heat and methane as well as excretions of faecal and urinary nitrogen are predicted from these simulations.

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Dietary electrolytes affect slurry composition and volume from dairy cows

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Introduction

Total slurry volume is of economical importance due to cost of storage and spreading. Furthermore, slurry dry matter content and composition might be of importance to biological processes (e.g. loss of ammonia), physical processing and separation of slurry. We hypothesized that dietary potassium (K) and sodium (Na) would affect slurry volume and composition in dairy cows through water and electrolyte homeostatic mechanisms.

Method

A 4*4 Latin square experiment with 20 lactating Holstein dairy cows, 4 periods of 7 days and 4 dietary treatments was conducted. Cows were on average 23 weeks in lactation (17-30 weeks), weighed 620 kg and had a mean milk yield of 31.9±2.4 kg per day at onset of the experiment. A total mixed ration based on maize silage was used on all treatments (Table 1).

Table 1. Dry matter composition of rations.

	Rations without Lucerne	Ration with Lucerne (high fiber)
Maize silage, kg	11.6	11.6
Rapeseed meal, kg	4.1	4.1
Rolled oats, kg	2.3	0
Perennial ryegrass straw, kg	1.3	1.3
Lucerne pellets, kg	0	3.6
NaCl, g	8-497	8-497
KCl, g	0-885	0-885
MgO, standard mineral mix and vitamins	+	+

Dietary treatments were: 1) Low-Na/Low-K; 2) Low-Na/High-K; 3) High-Na/High-K; 4) High-Na/High-K/high-fiber. Treatment levels were 12 or 35 g K and 1 or 10 g Na per kg dry matter, which was achieved by addition of chloride salts. Low or high indigestible fiber content was achieved by iso-energetic substitution of rolled oats by Lucerne pellets (approx. 4 kg dry matter cow^{-1} day^{-1}). Daily feed and water intake were measured individually during the last three days of each period, milk yield was measured (True-tester), and milk composition was analyzed (MilkoScan FT 120 infrared analyzer, Foss Electric A/S, 3400 Hillerød, Denmark) from the last evening and morning milking of each period. Fecal samples were collected from spontaneously dropped feces two times on the last two days of each period, and dry matter content (80°C) and water retention capacity (WRC) were measured (Canibe & Bach Knudsen, 2001).

Results

High versus low dietary KCl (both with low NaCl) significantly increased daily dry matter intake, water intake and milk yield (Table 2), but significantly decreased feces dry matter content. High versus low dietary NaCl (both with high KCl) further increased water intake significantly, but did not affect dry matter intake, milk yield or feces dry matter content.

Table 2. Daily milk yield and intake of feed and water. Analyzed water binding capacity of feces, and estimated daily excretion of feces and urine.

	Low-Na Low-K	Low-Na High-K	High-Na High-K	High-Na High-K High-fiber	SEM
Feed intake, kg	48.5^a	56.8^b	54.4^b	54.3^b	1
Feed intake, kg DM	21.4^a	25.6^b	24.7^b	25.5^b	0.5
Water intake, kg	72^a	136^b	157^c	159^c	4
Milk yield, kg	27.5^a	30.5^b	29.3^b	29.9^b	0.7
Feces, % DM (60°C)	17.3^a	14.8^b	15.1^b	14.9^b	0.2
Fecal WRC, g/g DM[1]	5.83^a	6.64^b	6.56^b	6.54^b	0.09
Feces, kg DM[2]	6.2	7.4	7.1	7.3	-
Feces, kg	36	50	47	50	-
Urine, kg[3]	45	98	121	119	-
Feces:urine	0.8	0.5	0.4	0.4	-
Feces+urine, kg	81	148	168	169	-

Different letters in superscript within row mean significant difference with $P \leq 0.05$
1: WRC=water retention capacity (Canibe & Bach Knudsen, 2001)
2: assumed DM-digestibility 71% (Hymøller et al., 2005)
3: etimated as (water intake + water in feed) – (water in milk + water in feces)

Fiber did not influence the mentioned parameters. Fecal water binding capacity was significantly lower on low dietary KCl and NaCl than on other treatments, which did not differ.

Increased milk yield accounted for 0-4% of the measured increase in water intake. Assuming no significant changes in other losses, the surplus water intake had to be excreted in urine and feces. The excretion of water with feces was estimated to account for 0-7% of the measured increase in water intake. On this basis it can be estimated that the high salt diets as compared to the low salt diet would increase slurry volume in the order of 25 to 30 ton per cow per year, and greatly influence the slurry dry matter content and the ratio between urine and feces.

Discussion and conlusions

According to Danish Standard Values for Farm Manure (Poulsen et al., 2001; Anonymous, 2006) the excreted volume of slurry from dairy cows (8750 kg milk) is 20 ton per cow per year, which corresponds to about 55 kg per day on the average. Nennich et al. (2005) estimated total manure excretion to be 75 kg and 39 kg per day from lactating (40 kg milk, 625 kg BW) and dry cows, respectively. The present data clearly shows a large variation in total daily manure volume depending on dietary electrolytes, which is not accounted for by the standard values. It must further be expected that manure composition will vary correspondingly.

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Chicken manure treatment and application – an overview of the ASIA-PRO-ECO project CHIMATRA

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Today, chicken manure can cause serious problems in many regions world-wide. Chicken manure is not only difficult to handle due to its consistency, it also generates problems from foul odours and may contain pathogenic bacteria, viruses and parasites. Uncontrolled use may lead to pollution of ground water and spreading of diseases. On the other hand, manure is a valuable resource for crop production due to its high nutrient content, especially with respect to phosphorus and nitrogen. As a result, manure application may also be interesting from an economical point of view.

In Asia, since chickens are a popular food in increasing demand because of the increasing standard of living and population size, it is expected that the amounts of chicken manure will increase drastically in the future. In Malaysia alone, the amount of chicken manure generated is approximately 14,000 tonnes per day. Today, land application of the untreated chicken manure is the most common outlet in Asia. Furthermore, the use of manure as feed, e.g. in fish ponds, is also a common practice. To overcome these uncontrolled and make use of the value-added properties of agricultural wastes, various treatments can be applied.

In Europe, a wide range of treatment methods are used already. The type of treatment chosen depends on the kind of waste and the regional situation. Options range from drying to composting, anaerobic digestion or pelletizing. Valuable products, such as pellets or composts, may be produced. They can be pathogen-free, easy to store and handle, enriched by nutrients and may be used in agriculture to substitute expensive mineral fertilizers. For anaerobic digestion, biogas is an additional valuable product. Another option with an energetic output is incineration.

The EU-funded project, Chicken Manure Treatment and Application (CHIMATRA), aims to spread knowledge and enhance the transfer of economical and environmentally-friendly chicken manure treatment and application technologies to Asia. Key activities so far have included the organisation of a workshop, conference and training course, the preparation of promotional material, and the setting-up of demonstration pavilions. More practical aspects included the development of a low-tech pelletizing system, the investigation of the distribution behaviour of pellets, and surveying the chicken manure challenges in Malaysia. The information will be disseminated to a network of professionals from sectors such as research organizations, authorities, manure producers and applicants, and equipment producers. More information can be found under www.chimatra.com. The project is a joint cooperation between the Institute of WasteResourceManagement, Hamburg University of Technology (TUHH), Germany, the Faculty of Environmental Studies, Universiti Putra Malaysia (UPM), Malaysia, and Wageningen University and Research Centre (WUR), the Netherlands. This work is financed by the European Union, under scope of the programme ASIA-PRO-ECO, and by the City of Hamburg.

The CHIMATRA-project has so far resulted the following outputs:

1) A workshop on "Chicken Manure Treatment and Application", which summarised state-of-the-art options, was carried out in Hamburg in January 2005, with more than 50 participants from around the world;
2) An international conference on "Agricultural Wastes", with a specific focus on South East Asian countries, was carried out in Kuala Lumpur in March 2006, with more than 160 participants mainly from Asia;
3) A training course on "Chicken Manure Treatment and Application", introducing the most important techniques to chicken farmers and managers, took place in Kuala Lumpur in March 2006, for 44 participants;
4) The proceedings to the workshop and an extended-abstract book of the conference were published (see References);
5) A brochure entitled "Chicken Manure Treatment and Application", summarising the problems and possibilities in a colourful and easy-to-understand way, in English, Chinese and Malay;
6) A video in the same 3 languages as an educational documentation of the topic.

Ongoing activities include the following:

1) The TUHH is developing a treatment method for chicken manure which includes agglomeration, hygienization and drying. The method aims to be economical and easy to implement on European and Asian farms. The end products are spherical, easy to apply pellets;
2) The UPM is to carry out an evaluation regarding chicken manure production in Malaysia, and to evaluate the manure quality on selected farms through a questionnaire and by laboratory analysis. Furthermore, a study tour around selected Asian countries will extend the focus and gain more involvement from other countries;
3) The WUR is to carry out investigations into the application of chicken manure products. These include plant tests, as well as pellet distribution tests;
4) An internet-accessible database containing contact details of all groups involved in the field of chicken manure is to be set up. It will cover research institutes, public, governmental and municipal authorities, chicken manure producers, fertilizer users and producers of technological equipment;
5) Demonstration pavilions are being constructed to contain modules such as equipment, posters and information leaflets, which demonstrate the variety of different treatment and product options. The main pavilion will be installed in Malaysia at the UPM campus, while smaller ones will be set up in Germany and the Netherlands.

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Chicken Manure Treatment and Application – Educational Video, TUHH, UPM, WUR, available online: www.chimatra.com

Animal manure management in China

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Introduction

Animal husbandry has become one of the pillar industries in rural economy and an important source of increasing farmer's income in China. The animal production value reached 1,217.4 billion Yuan, accounting for 33.6% of the Chinese agriculture production in 2004. According to the data by FAO, in 2004 the meat, egg and milk production in China accounted for 28.2%, 45.3% and 3.7%, respectively, of the world production. The number of swine, cattle, sheep and poultry in stock accounted for 49.7%, 8.0%, 18.7% and 27.8%, respectively, of the world figures. Associated with the rapidly increasing animal production are environmental challenges. As the animal production grows, livestock facilities tend to be more concentrated with larger amount of animal waste discharged. Animal manure changes from valuable resources to waste because of improper management and application. Consequently, animal manure management is considered one of the environmental priority issues in China. The objective of this paper is to describe the current state of pollution and emission controls in China, and to identify the challenges and opportunities.

Material and methods

A multitude of sources were reviewed for this overview. Data sources on animal production were from China official statistical data, while information on production parameters, housing types and manure management systems were from a field survey by the research group led by Dong (Liu and DONG, 2006).

Findings on current state

(1) In 2005 fresh animal manure production in China was estimated to be 3.0 billion tons, which was about 2.24 times the industrial and municipal solid waste output (1.34 billion tons). The average manure load is approximately about 24 tons of manure over the country's farmland of 1.837 billion Mu or 122 million hectares. The highest provincial average

manure N load was 127 kg N per ha, although the N load was much higher in some areas. Also because of the low profit of the livestock industry and poor transport means, 80% of the large and medium-sized farms are located in densely populated areas in Eastern China and around major cities. Furthermore, most of the farms lack effective pollution-prevention facilities. Consequently, animal production has become one of the largest pollution sources in China.

(2) The Chinese government has paid more attention to prevention of pollution from livestock production, thereby ensuring sustainable agricultural development. The National Strategy on Pollution Prevention and Control from Livestock Production was issued by the State Environmental Protection Agency (SEPA) in 2001. In 2003 the national discharge standards for pollutants from livestock production was established. The major parameters of discharge standards are lists in Table 1.

Table 1. Threshold values of average daily pollutant concentrations of discharged waste water in China

Parameters	BOD_5 mg/L	COD mg/L	SS mg/L	NH_3-N mg/L	TP mg/L	Coliform N/100mL	Ascarid N/L
Threshold	150	400	200	80	8.0	1000	2.0

(3) The Ministry of Agriculture (MOA) of China is implementing an Eco-household Project for promoting sustainable agricultural development. This program focuses on using biogas and solar energy as supplementary measures to increase farmer's income while improving their living environment. By the end of 2005, the number of household biogas digesters had reached 17 million with a total biogas production of 6.5 billion m^3. Ten percent of the rural families have the opportunity to use biogas. The number of large-scale biogas digesters for manure treatment had reached about 2000. This program will mitigate pollution from animal manure to some extent.

(4) A variety of manure treatment and utilization systems, such as compost and biogas digester, have been applied to meet the requirement of different scales of animal farms in China. However, the extent of utilization is limited. In recent years, combined treatment and utilization systems for animal manure have been

increasingly developed to meet the need of large-scale animal operations. The process includes pre-treatment of various raw materials, selection of better technical parameters for improved digestion, after-treatment of effluent for discharge, reuse and land application. Great improvements have been made in this type of system. Solid and liquid separator is widely used to reduce the solids content of slurries and thus required volume of digesters. Combined anaerobic-aerobic technologies are being adopted on large-scale farms located in the land-limited areas. Sequential batch reactors (SBR) are applied to treat the effluent of anaerobic digesters. However, the high operation cost has limited the extension of this technology to medium- and small-scale animal farms. Membrane technology is under investigation. The Chinese Academy of Agricultural Sciences (CAAS) is exploring the application of membrane technology in animal manure treatment. Land application is one of the most widely accepted ways to utilize nutrition in animal manure. However, excess nitrogen and phosphorus from animal residues have caused increasingly serious non-point pollution of soil and waters.

Conclusions

With the development of intensive animal production, large- and medium-scale biogas digesters and combined treatment systems have become a priority. A variety of technologies on further treatment and/or utilization of effluent from anaerobic digesters have been explored. However, a large gap in terms of the cost and the rate of industrialization or commercialization exists between the current technologies and the requirement. How to reduce the cost of manure treatment and utilization remains a challenge facing the Chinese researchers and farmers.

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Farm location and manure management practices - Insights from a case study on pig production in Central Thailand

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Introduction

Fuelled by a growing population, rising income and urbanisation, demand for animal products is burgeoning in the developing world. Among the developing countries, Asia has the fastest developing demand, especially for pig and poultry meat. Over the 1982 to 2002 period, demand for pig and poultry meat grew by 8.1% and 6.0% p.a., respectively (FAO 2006). The livestock sector is responding to this surge in demand for livestock products with some drastic transformations that impact the type of manure produced as well as its management practices. This in turn has substantial environmental consequences, especially as far as nutrient balances, water pollution and gaseous emissions are concerned, which policy makers are urged to address by a growing middleclass (FAO/LEAD 2004). In this paper, we compare environmental policy options for pig production in Thailand, focusing on two alternatives: pollution control at farm level and improvement of the spatial distribution of production units.

Trends in intensive livestock production systems and manure management practices

In Thailand, two major trends in the livestock sector have strong implications on manure management practices. First, the proportion of large scale livestock production met by specialised and intensive industrial systems is increasing rapidly. The industrialisation of production leads to a disconnection between livestock activities and cropping activities. The growing intensive production essentially relies on feed produced out of the production area, imported genetics, and equipment, and supplies local and international markets. Second, livestock production tends to concentrate in areas favoured by cheap input supplies (particularly feed), and by good market outlets for livestock products. Such conditions are found in the vicinity of large cities. This results in the rapid development of industrial

livestock production around urban areas, far from agricultural land. Both trends tend to diminish opportunities for manure recycling.

Main spatial parameters affecting production costs
Transport plays a predominant role among the spatial parameters that affect production costs. Transport costs affect the prices of feed, piglets, finished fatteners and, to a lesser extent, of health care. Price of land also affects production cost, following a general gradient inverse to transport costs. Water availability is another parameter that varies with location and which affects production costs. Given the current technology and water distribution infrastructure, water availability determines the area suitable for pig production, but does not substantially alter production costs.

Main parameters affecting manure management strategy
A number of factors influence how producers manage manure. At the centre of their decision is the estimated value of manure and its derived products, which depends on the opportunities for using manure as an input to other activities. Such activities can be within the same production unit or outside. In the latter case, and in the case of no government intervention, transfers are driven by market demand. Liquid waste is also exported from production sites, although quite rarely and most often at no cost to the crop farmer or fish grower receiving the waste.

Another factor shaping manure management practices is the environmental policy in place. In Thailand, discharge standards exist regulating N, Biological Oxygen Demand (BOD), Chemical Oxygen Demand (COD) and suspended solids content, as well as the pH-level of effluents from farms. Different maximum standards are defined for large farms (more than 600 livestock units) and for small and medium farms (less than 600 livestock units). These standards, however, are poorly enforced and often not implemented (FAO/LEAD 2004).

Materials and methods
A farm model was produced to test farmers' manure management practices at different spatial locations and under varying policy frameworks. Based on linear programming, the model selects manure management options to maximise farm's profit under resource and environmental policy constraints. A simple material flow analysis is built into the model to evaluate N, P and BOD emissions of the production unit (Gerber 2006).

For the analysis, we selected a large scale farm with access to state-of-the-art pig production and manure management technology. The standing pig population is set at 2,000 fattening pigs and the farm is endowed with 1 ha of land on which it can grow sugar cane. Four scenarios are investigated: peri-urban environment with no environmental policy (A1); peri-urban environment with enforcement of current discharge standards (A2); rural environment with no environmental policy (B1); and rural environment with enforcement of current discharge standards (B2).

Results

First, as expected, results showed that the enforcement of current standards would drastically reduce emissions: N emissions are reduced from 17.9 to 1.2 ton per year when moving from scenario A1 to A2. This improvement of environmental performances, however, goes with a substantial reduction of profit (*ca.* 25% in the case of farm A) and thus raises questions of actual enforceability.

To assess the combined financial and environmental performance of policy options, we calculated the ratio between changes in yearly profit and changes in pollutant emissions when shifting from one scenario to another (Table 1). We have seen that moving from scenario A1 to A2 goes with substantial losses. The different values calculated for N, P and BOD are explained by contrasting emission reductions. Comparing A1 and B1 showed that, in the absence of environmental policies, the simple relocation of farms increases profit while reducing emissions. The pollution reduction that can be obtained from a simple relocation of farms is, however, limited (3.7 tons per year for N), except when a market demand is estimated for liquid manure. Enforcing current standards in rural environment (moving from B1 to B2) achieved similar pollution reduction as in the case of urban setting at much reduced costs for the farmers (profit reduced by ca. 1.5%).

In the current context, several constraints prevent from taking advantage of an improved geographical distribution of production units. First, farmers are often not acquainted with the advanced manure management practices selected by the model and that contribute to increase profit in rural setting. Second, farm's financial performance is usually calculated on the sole basis of pig production which is more profitable in peri-urban

setting. Third, the relocation of farms requires mobility of people and capital, and farmers are often not willing to leave their current location.

Table 1. Reduction of profit per unit of emission reduction when moving from one scenario to another.

Change in scenario	Parameter	Reduction of profit per unit of emission reduction ($/kg)
A1 to A2	N_{total}	0.90
	P_2O_5	3.13
	BOD	0.20
A1 to B1	N_{total}	- 0.54
	P_2O_5	- 0.25
	BOD	- 0.04
B1 to B2	N_{total}	0.09
	P_2O_5	1.25
	BOD	0.05

Conclusion

This analysis shows that, from a private perspective, improved spatial location of farms is a cost efficient way to reduce environmental impacts of intensive production. Farms located in crop intensive environments can achieve pollution reduction while improving profit. The analysis further shows the strong synergy between regional planning and pollution control strategies. Even in the case of improved geographical distribution, simple discharge of manure will, however, most often appear as a simple and cheap option to farmers. Awareness raising and capacity building are therefore critical to improve management practices and yield the win-win opportunities identified in this paper. In addition, farm level control policies are necessary in the mid to long term. They shall not only address point source pollution, but also non-point source pollution related to waste application on agricultural land.

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Slurry acidification – consequences for losses and plant availability of nitrogen and sulphur

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Introduction

A new technique for slurry acidification has been developed and introduced in Denmark. The slurry is acidified in a tank by controlled application of sulphuric acid to pH about 5.5. The acidified slurry is then pumped back to the livestock buildings, resulting in a reduction of pH of the new excreta shortly after excretion. A 70-80% reduction of ammonia emission from buildings has been documented by this technique, and the utilization of nitrogen in the field may also be influenced. After acidification it is also possible to aerate the slurry without loosing ammonia, and thereby the manure properties may be improved.

Animal manured crops often need additional mineral sulphur fertilizer. Slurry acidification to pH 5.5 requires approx. 5 kg sulphuric acid ton^{-1}, equivalent to 1.6 kg SO_4-S ton^{-1}. This quantity will ensure sulphur sufficiency if this pool of S is plant-available at the time of application.

Materials and Methods

In 2005 the turnover and utilization of N in untreated and acidified cattle and pig slurry were compared in incubated soil and in small field plots. The slurries were acidified by slow addition of concentrated sulphuric acid to pH 5.5. Both slurry acidified in full scale on a farm and slurry acidified under controlled conditions were investigated. In addition, effects of aeration time (0 h, 6 h , 4 d) of the acidified slurry on the composition (volatile fatty acids, total N, ammonium N, DM) and N fertilizer value were measured. The aeration was made in small-scale containers (30 kg batch) at 6-8°C and an airflow of 1.2 ml air sec^{-1} kg^{-1}. The N fertilizer value was measured after slurry incorporation before sowing of spring barley, and after surface banding in a winter wheat crop.

The turnover of S was investigated in untreated and acidified pig slurry stored for up to 11 months at 2, 10 or 20°C. Furthermore, the fertilizer efficiency of radio-labelled sulphuric acid in acidified slurry was investigated in pot experiments with spring barley.

Results and Discussion

The aeration of acidified slurry had no influence on the content of volatile fatty acids or the content of organic N in slurry, neither on the N fertilizer value and the N turnover after application to soil. Zhang & Zhu (2005) found that organic acids in pig slurry can be reduced by aeration at 20°C. In this study, the slurry was aerated at a temperature of about 7°C, which is a normal slurry temperature in Denmark. It is not clear whether the lacking effect of aeration on microbial decomposition was due to the acidification or to the lower temperature. Slurry sampled on farms with acidification and short-term aeration also showed a high content of butyric acid and a relatively high content of organic N.

The N fertilizer value of untreated slurry applied by surface-banding to winter wheat was 20-25% lower than after soil incorporation to barley (Fig. 1). However, with acidification the fertilizer value was high and similar in both crops, indicating that ammonia volatilization was low after surface application of the acidified slurry. The manure analyses indicated that acidification inhibits the turnover of organic matter in slurry during storage, but the field experiment showed that a higher N utilization can be attained after the slurry acidification.

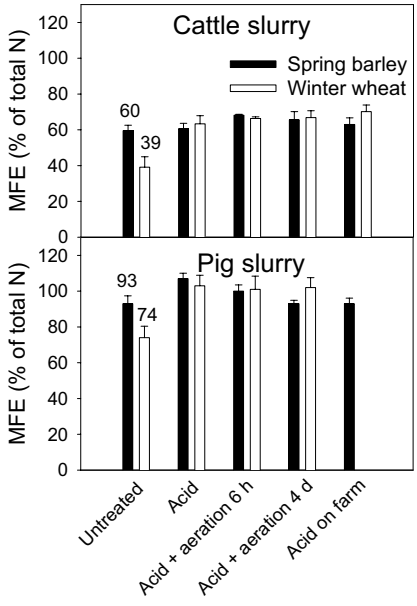

Figure 1. Mineral fertilizer equivalent (MFE) of total N in untreated, acidified, acidified+aerated pig and cattle slurry applied by surface-banding to winter wheat or by incorporation to barley. Error bars: SE (n=4).

Sulphate from acidification with sulphuric acid was relatively stable and even after 11 months of storage the main part was in the plant-available sulphate form (fig. 2). Sulphide accumulated during storage, especially at high temperatures, but the levels in acidified slurry did not exceed those of the untreated slurry for several months after addition. Only little sulphide originated from added sulphuric acid. At the highest temperature sulphide concentrations decreased during the last months of the storage period. The fertilizer value of sulphuric acid-S in slurry was considerable as result of the stability of sulphate during storage. However, there is concern that the high content of inorganic S in acidified slurry may potentially lead to development of bad odour from volatile sulphur-containing compounds. A high content of organic acids, especially butyric acid, in acidified slurry also contributes to bad smell.

The major benefits of the slurry acidification method are the reduced ammonia loss both from buildings, storage and after field application, and the increased S and N fertilizer value of the treated slurry.

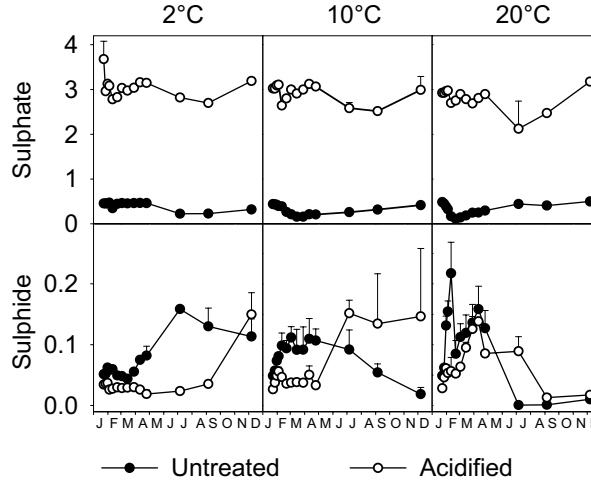

Figure 2. Content of sulphate and sulphide (kg/tonnes) in untreated and acidified pig slurry during 11 months of storage at different temperatures. Error bars: SE (n=4).

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Innovative technology for recycling of manure phosphorus with rapid amorphous phosphate precipitation

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A treatment system technology was developed for recovery of soluble P from liquid pig manure in calcium phosphate form as an alternative manure management method when soils are saturated with P and on-farm land application of manure is not an option (Vanotti et al., 2005). Soluble P is recovered as a P precipitate by increasing the pH of wastewater with controlled amounts of hydrated lime [$Ca(OH)_2$] after manure solids removal and biological N treatment steps.

This new manure treatment system was first pilot tested to replace the traditional anaerobic lagoon treatment at North Carolina State University's Lake Wheeler Road Swine Unit. A full-scale version of this innovative system was subsequently constructed at a pig farm in North Carolina. The full-scale system was constructed for demonstration and performance verification of environmental superior technology (EST) that was sponsored by a state government-industry framework to develop technologies eliminating anaerobic lagoons as a pig manure treatment method (Vanotti et al., 2006).

In this treatment system, manure solids are first separated from liquid with polyacrylamide (PAM) polymer and filtration. The polymer causes flocculation of suspended particles and enhances separation of solids from liquid. In a second step, ammonia and carbonate alkalinity are biologically removed from the liquid through nitrification. Once ammonia and carbonate alkalinity concentrations are reduced with nitrification treatment, small quantities of $Ca(OH)_2$ are added to rapidly increase the pH of the liquid above 9, thereby promoting formation of calcium phosphate precipitate (Vanotti et al., 2003). In average, 94% of soluble phosphate is recovered in the precipitate for wastewater containing 77 to 191 mg/L soluble P (Vanotti et al., 2006). The phosphate precipitate is

then dewatered using a combination of a flocculant (anionic PAM) and polypropylene filter bags (Szogi et al., 2006).

A study was conducted to determine the elemental and mineralogical composition of precipitates from both the pilot and the full-scale systems, and to determine the potential use of P precipitates as a phosphate fertilizer. Samples of the precipitates were subjected to chemical analysis, x-ray diffraction (XRD) and scanning electron microscopy (SEM) for mineral identification. Chemical analysis indicated that the precipitates contained both Ca and P with Ca/P molar ratios in a range of 1.3 to 1.67, characteristic of amorphous calcium phosphate (ACP) precipitates (House, 1999). The XRD spectra revealed the precipitates contained calcium carbonate as calcite, but no crystalline phosphate minerals; while SEM confirmed the predominantly amorphous structure of the precipitate.

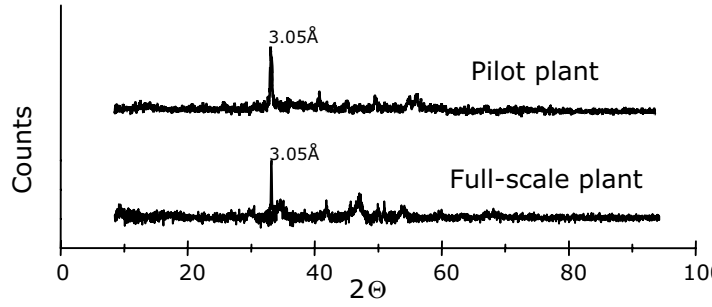

Figure 1. X-ray diffraction patterns of precipitate samples obtained from pilot and full-scale systems. Diffraction patterns suggest the samples contain calcium carbonate as calcite (d-spacing 3.05Å), but no crystalline phosphate minerals.

Chemical analysis also indicated that the precipitates contained > 20% P_2O_5, mostly as plant available P. A soil fertility test using ryegrass showed that the recovered phosphate applied in two particle size forms (0.5 – 1.0 mm and 2.0 – 4.0 mm) was an excellent source of P. Ryegrass dry matter yields obtained using recovered P were similar to commercial triple superphosphate (TSP) at five application rates (0, 20, 40, 80 and 160 mg P/kg soil). However, plant uptake was lower for the recovered phosphate (RP) source with respect to TSP (Figure 2). These results indicated that RP may be a slow release fertilizer because it is less soluble than commercial TSP.

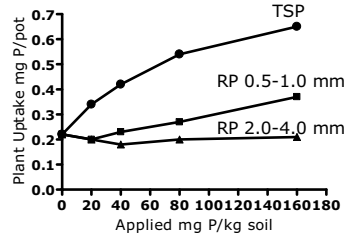

Figure 2. Relationship between applied P and plant uptake. Ryegrass was fertilized with commercial triple super phosphate (TSP) and recovered phosphate (RP) using two particle sizes (0.5 – 1.0 mm and 2.0 – 4.0 mm).

This innovative technology for the recovery of phosphates from liquid pig manure is useful for solving distribution problems of excess manure P in soils, and it allows significant amounts of this nutrient to be transported off the farm in concentrated form and recycled as plant fertilizer.

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Removal of *Salmonella* contamination before using plant nutrients from household waste and wastewater in agriculture

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Abstract
For a sustainable society, plant nutrients consumed in foods and feedstuffs need to be recycled back to arable land. However, closing this loop from the urban to the rural area increases the risk for transmission of infective agents. In the present study we followed the presence and reduction of *Salmonella* spp. due to waste treatment and also the occurrence of recontamination during the post-treatment handling. Pasteurisation of biowaste in a biogas plant or ammonia treatment of faecal matter was found to remove the *Salmonella* spp. However, post-treatment handling carried a major risk for *Salmonella* recontamination of the treated material. The only treatment that protected the treated biowaste from re-growth, if performed correctly, was urea treatment.

Introduction
In today's society, nutrients are exported from arable land in harvested foods and feedstuffs, distributed throughout the society and consumed by people. After digestion, most of the nutrients are directly or indirectly transferred to the water recipients via the excreta. Nutrient resources are scarce and those most used (macronutrients) are in danger of becoming scarce within the coming century (Jönsson et al., 2004). Today, the energy use both for nutrient removal from wastewater and for production of new fertilisers is high (Vinnerås, 2002). If some of the nutrients, i.e. nitrogen and phosphorus, are not removed from the wastewater they can cause eutrophication in water recipients. To decrease the use of energy and to increase the sustainability of society, these nutrients have to be recycled.

Different treatment systems available today make it possible to recover parts of the nutrients in biowaste, i.e. waste and wastewater derived biodegradable household products, and to recycle them back to food and

feed production areas, e.g. source separating sewage systems or reuse of sewage sludge and biowaste (Jönsson et al., 2004). However, when the nutrient cycle becomes smaller, the risk for transmission of diseases increases. To avoid increased transmission of diseases the recycled material has to be properly treated for removal of pathogens.

After infection the excretion of pathogens is mainly via the faeces, as few organisms are excreted via the urine (Höglund, 2001). Therefore, analysis of sewage sludge that reflects the content of pathogens in wastewater mainly reflects the content of pathogens in faecal matter. The sewage sludge, faecal matter and blackwater (toilet water) can be considered to hold similar pathogen concentrations, somewhat dependent on the dilution.

The objective of this study was to compare different treatment alternatives for biowaste regarding the risk for survival of pathogens, using *Salmonella* spp. as a model organism, to enable safe nutrient recycling.

Results and Discussion

Selection of treatment must depend on the material to be treated and the intended use of the product. The risk of finding pathogens in biowaste increases with the size of the collection system. The post-treatment handling is important to avoid re-growth or contamination of the treated material. In Table 1, the different treatment alternatives are compared to storage, which is the reference scenario. Storage is the simplest treatment method, but prior to storage the material has to be stabilised. If the biowaste is not stabilised, there is a risk of odour emissions and of attracting potential vectors, such as flies and rodents.

Urea treatment was the most efficient and safe of the three treatment alternatives compared, one of the main advantages pointed out in Table 1 being the protection towards re-growth of pathogenic bacteria and the short time of treatment needed, i.e. recommended treatment using 1% urea-nitrogen is 2 weeks at a temperature $>10°C$, and 1 month at temperatures $<10°C$. Using 0.5% ammonia (30% aq solution), which results in a higher pH, only 7 days of treatment will be enough for assuring no risk for transmission of *Salmonella* with the manure.

Table 1. Comparison of different treatment alternatives for biowaste. A zero indicates that the scenario is comparable to the reference scenario (R). The symbols + or – symbolise whether a treatment is better or worse than the reference scenario. '++' indicates a significantly better system, '+' a somewhat better system.

Treatment	Time	Cost	Re-growth	Infrastructure	Tech. S.P.*	Efficiency
Storage	R	R	R	R	R	R
Urea	+	0	+	0	0	+
Heat	++	-	-	-	-	+
Heat + biol. stabilisation	+	-	0	-	-	+

*Tech. S.P. = Need for technically skilled personnel

Heat treatment needs to be combined with stabilisation of the biowaste. The normal case is that the material is digested after the heat treatment, or that it is composted. The stabilisation makes the material less sensitive to re-growth or contamination of the biowaste.

The most important consideration when selecting a treatment alternative using the matrix presented in Table 1 is to identify the factors that are important for treatment of the biowaste in each case, and to adapt the method to the prevailing circumstances. The aim of the biowaste treatment should be to enable safe recycling of plant nutrients, and thus to reduce the use of finite resources.

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Effects of slurry additives and ozone treatment on odour emissions from pig slurry

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Introduction

Livestock manure is an important source of odour, and odour emissions from slurry stores and slurry application damage the reputation of livestock producers both regionally and nationally. Reactions to odour nuisances from neighbours are increasingly affecting existing livestock operations, and may constrain the further development of livestock production. Abatement technologies such as various types of slurry additives and ozonation systems have emerged as strategies to reduce the odour nuisances from livestock manure. Aeration of pig slurry by ozone has shown to be able to reduce the concentration of malodorous metabolites in pig slurry (Watkins et al., 1997; Wu et al., 1999), and various types of additives are asserted to be able to reduce odour nuisances of slurry. However, documentation of effects on odour emissions are generally lacking (McCrory et al., 2001).

The aim of the present study was to evaluate the odour reduction effects on pig slurry by ozonation, and by addition of two different slurry additives, Terra Biosa (TB) and Siolit plus (Siolit). Both additives are asserted to be able to reduce the odour nuisances of slurry. According to the producers, the active ingredients in TB are a mixture of aromatic herbs and other plants which are fermented by a special combination of lactic acid and yeast cultures, while mineralized quarts and minerals seem to be the active ingredients in Siolit.

Materials and methods

Untreated and ozone treated pig slurry were sampled from the same pig finisher house. The ozone treated slurry had recently been aerated by 2-5 g ozone m^{-3} slurry using a commercial ozonation system (WaterAir-clean, APS). The untreated slurry was homogenized and amended with 2 litres of TB or 20 g of Siolit per m^{-3} slurry, or it was left untreated. Forty-three

litres of each treatment was stored under identical conditions in 63 l plastic barrels in triplicate. To determine the olfactometric response of each treatment, the air above the stored slurry was sampled following thorough mixing of the slurry samples; this took place 1, 21 and 56 days after the treatments. The olfactometric responses were quantified using dynamic dilution olfactometry, and by determining concentrations of more than 30 volatile organic odorous compounds by use of thermal desorption tube sampling coupled to gas chromatography/mass spectrometry (GC/MS) analyses. During sampling, barrels were closed and ventilated by 1.8 l air min^{-1}. In addition, periodic analyses of slurry samples were performed to determine effects on pH, and concentrations of dry matter, nitrogen, and volatile fatty acids.

Results

Aeration of slurry by ozone was found to reduce odour emission and slurry concentrations of malodorous volatile fatty acids, and to influence the nutrient composition and physical properties of the slurry. However, the odour reduction effects diminished during subsequent storage (Fig. 1).

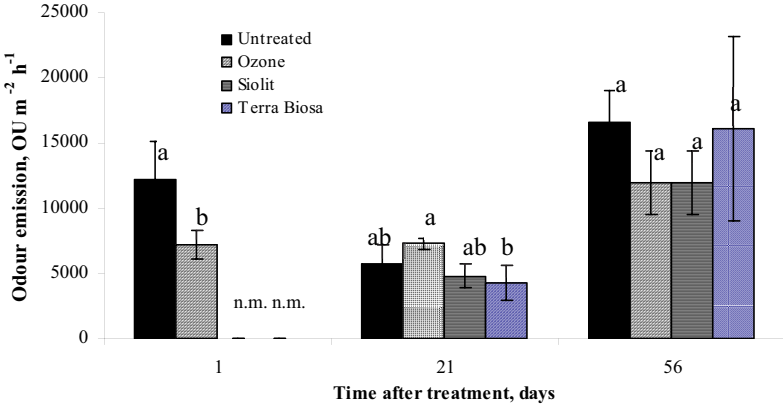

Figure 1. Effect of ozonation and addition of two different slurry additives on odour emission from stirred pig slurry during subsequent storage. Significant differences for the same day of observation are indicated by different letters ($P < 0.1$). Bars indicate standard deviation. n.m. = no measurements taken.

Ozonation of pig slurry was found to have long-term effect on reduction of emission of indolic and phenolic odorous compounds, while higher concentrations of sulphurous and especially carboxylic acids odorants were

observed in air emitting from the slurry treated by ozone compared to the untreated slurry (Table 1). Addition of the slurry additives, Siolit plus and Terra Biosa, in quantities recommended by the producers, was not found to influence neither odour emission (Fig. 1) nor concentrations of odorous gasses in air leaving the storage facilities during subsequent storage (Table 1). Besides, addition of the slurry additives did not affect neither the chemical, nor the physical composition of the slurry.

Table 1. Effect of ozonation and addition of the slurry additives, Terra Biosa and Siolit Plus, on concentrations (ug m^{-3} air) of selected odorants in air sampled above slurry stored 1, 21 and 56 days after treatment. Values in parenthesis are per cent of the concentrations measured above untreated slurry.

Treatment	**Untreated**			**Ozonation**			**Terra Biosa**		**Siolit Plus**	
Days after treatment	1	21	56	1	21	56	21	56	21	56
Butanoic acid	337 (100)	383 (100)	112 (100)	16206 (4801)	13414 (3503)	4991 (4456)	575 (150)	138 (123)	412 (108)	191 (170)
3-methyl butanoic acid	76 (100)	83 (100)	46 (100)	1970 (2606)	1870 (2249)	1393 (3061)	95 (114)	63 (137)	75 (90)	77 (169)
Indole	1263 (100)	695 (100)	3107 (100)	626 (49)	256 (37)	2210 (71)	525 (76)	4252 (137)	578 (83)	3153 (101)
Skatole	1072 (100)	563 (100)	16381 (100)	726 (68)	488 (87)	-	632 (112)	-	463 (82)	-
Phenole	1333 (100)	1361 (100)	1548 (100)	706 (53)	584 (43)	853 (55)	1257 (92)	282 (167)	1405 (103)	1929 (125)
4-methyl-phenole	750 (100)	-	-	601 (80)	-	-	-	-	-	-
Dimethyl-sulphide	22 (100)	-	21 (100)	8.3 (39)	-	40 (191)	632 (92)	14 (66)	-	57 (274)
Dimethyl-trisulphide	18 (100)	34 (100)	63 (100)	100 (322)	142 (415)	28 (67)	88 (258)	28 (67)	47 (138)	50 (119)

- = not detected

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Wet oxidation pre-treatment – the way to improve economics of energy production from manure?

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Introduction

Anaerobic digestion (AD) of manure offers several benefits for agricultural waste management: It improves the fertilizer characteristics of manure, reduces its odour and converts organic matter into valuable energy resources like, for example, biogas. Using manure for bioenergy production in centralized biogas plants results often in a marginal economical benefit, since the value of the biogas yield per ton is only slightly higher than the costs for treatment and transportation. A newly developed pre-treatment method, wet oxidation (Lissens et al. 2004), has recently been successfully applied for the pretreatment of straw for bioethanol production at BioCentrum-DTU, the Technical University of Denmark. This pre-treatment method is a very promising option to increase both bioethanol and biogas yields from lignocellulosic biomass, i.e. all kinds of agricultural waste.

Throughout recent years, several physical, chemical and biological pretreatment methods have been tested with limited success in order to increase hydrolysis of the lignocellulose structure of manure fibers (Fan et al., 1982; Gharpuray et al., 1983; Grethlein, 1984; Hartmann et. al. 2000; Hobson and Wheatley, 1993). The combination of high temperature and pressure, together with the addition of oxygen in the wet oxidation process, has shown a superior potential for increasing the hydrolysis of lignocellulosic biomass with subsequent ethanol fermentation. In the present study the wet oxidation process was tested as pretreatment for increasing the biogas yield of the fiber fraction of manure.

The wet oxidation pretreatment in combination with solid liquid separation of manure by using either chemical precipitation or decanter centrifuge (Møller et al., 2004) could be a promising option for a significant increase of the economical benefit in the AD treatment of manure.

Methods

Reactor experiments (3 L active volume) were performed on wet oxidized manure and manure fibers using non-pretreated manure and manure fibers in control experiments. One reactor experiment was performed by shifting the feed from raw manure to wet oxidized manure. In another reactor experiment, wet oxidized fiber material was introduced in co-digestion with raw manure, and the ratio of wet oxidized fibers was increased until only wet oxidized fibers were used as feed.

Results and Discussion

Shifting the reactor feed from raw cow manure to wet oxidized manure resulted in an increase of the biogas yield from 0.26 m^3/kg-VS (control reactor R2) to 0.44 m^3CH_4/kg-VS (reactor R1, see Figure 1). With a typical organic matter (VS) content of 6.5% in cow manure, the increase was 71% from 17 m^3 biogas to 29 m^3 biogas per ton manure.

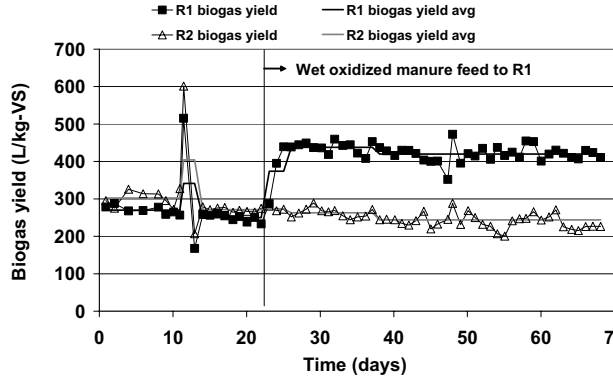

Figure 1. Increase of biogas yield after feed shift from raw to wet oxidized cow manure in reactor R1.

In the second reactor experiment two reactors were operated in co-digestion of a mixture of cow and swine manure with addition of higher ratios of raw fibers (reactor R2) and wet oxidized fibers (R1), respectively. Figure 2 shows the biogas yields of the raw and wet oxidized fiber fraction per ton of fiber material with a VS concentration of 30%, which will be typical for separation from a decanter centrifuge. The biogas yield from the wet oxidized fiber fraction was 54-77 m^3/t higher than of the non-treated fibers. The increase to 100% wet oxidized fibers in reactor R1 showed no inhibiting effect at biogas yields of 132 m^3biogas/t.

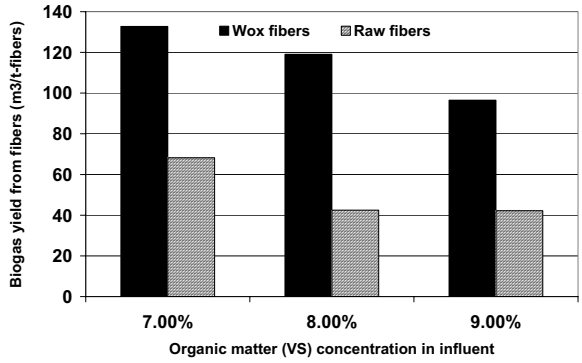

Figure 2. Increase of biogas yield from manure fibers after wet oxidation

Economical considerations

The current design of a pilot-scale facility of the wet oxidation process at BioCentrum-DTU allows for initial calculations of the process economy of future industrial-scale facilities. The wet oxidation process can be implemented at a centralized biogas plant in combination with solid liquid separation of raw or digested manure.

Regarding the energy balance of the wet oxidation pretreatment, results show that an increase of the biogas yield by 54-77 m^3/t is equivalent to an increase in gross energy output of 0.36-0.51 MWh/t (65% CH_4 in biogas) while the energy consumption of the process will be 0.1 MWh/t. If the heating energy for the wet oxidation is provided by surplus heat from a combined heat and power unit (CHP) connected to the biogas plant, operational costs are reduced to a minimum.

Investment costs for wet oxidation equipment for the treatment of 20,000 t/year of manure fibers in continuous mode are currently estimated to around 1.7 mio. Euro. Depending on the increase of biogas production the revenue will be 292,000 – 416,000 Euro per year (at a price of 0.27 Euro/m^3 biogas), i.e. the capital payback period will be 4-6 years.

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Optimal plant nutrient distribution in and biogasification of manure separation products, separated with $FeCl_3$ as coagulant

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Introduction

Separation of animal slurry has the potential to improve a) the nitrogen (N) and phosphorous (P) distribution among crop fields; b) manure management through concentrating the P in a smaller volume; c) N use efficiency of the liquid fraction; and d) the biogas production from the solid fraction. Separation is, thus, economically sound, environmentally friendly and can improve manure management.

Addition of coagulant and flocculent can be used to enhance slurry separation. Coagulants are electrolytes reacting preferably with polyvalent anions such as phosphate (Gregory, 1989). Coagulants enhance aggregation of colloids in the manure by reducing the effective surface potential and the diffusive layer, and can thus enhance slurry separation. The coagulants $Al_2(SO_4)_3$, $AlCl_3$, $Fe_2(SO_4)_3$ and $FeCl_3$ are commonly used. Ammonia volatilisation upon storage may potentially be reduced by using an acidifying coagulant for the separation. $FeCl_3$ has been shown to provide the largest pH reduction of the liquid fraction (data not shown).

Volumetric biogas production is expected to increase due to the higher dry matter content of the solid fraction. However, biogasification depends on the relative effect of the coagulant on a) NH_3 inhibition of the fermentation; b) volatile fatty acid (VFA) inhibition of the fermentation; and c) microbial growth conditions. The most promising coagulant was considered to be $FeCl_3$ due to known toxic effects of Al and SO_4^{2-} (i.e. H_2S) on microorganisms.

The objective of this study was to investigate the effect of separation using varying concentrations of $FeCl_3$ on P-to-N ratio, on NH_3 volatilisation during storage of the liquid fraction, and on biogasification of the solid fraction.

Methods

Separation was performed on a sample of pig slurry (6% dry matter) using coagulant and flocculent together with a belt separator. For the separation 0, 2.5 and 10 ml (i.e. no, little and much) 40% $FeCl_3$ was used as coagulant per litre manure. During storage of the liquid fraction, pH and NH_3 emissions were observed using dynamic chambers. Mesophilic anaerobic digestion was performed on the solid fraction in batch experiments.

Results and discussion

The P concentration of the liquid fractions obtained decreased relative to the N concentration with increasing $FeCl_3$ addition (Figure 1), reflecting the fact that both the N and P concentrations decreased, but P most significantly (data not shown). The dry matter content of the liquid fractions was 1.2-1.5%. A Danish wheat field requires an N-P ratio at approximately 9; hence the obtained liquid fractions are short on P. However, due to a fertilization history with excess application of P on most agricultural land in Denmark, P shortage in the applied manure would be

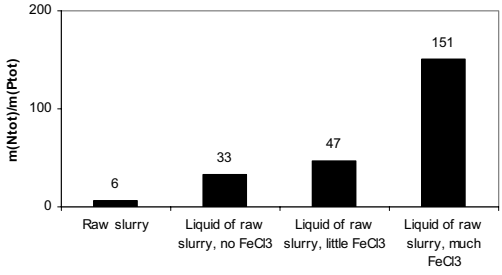

Figure 1. N-to-P ratio of the liquid fractions.

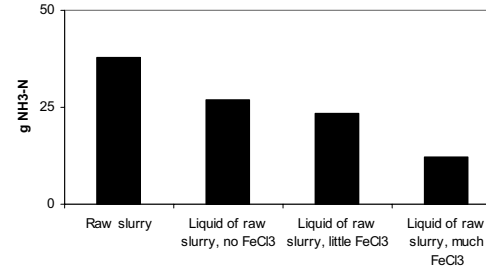

Figure 2. Total NH_3 emission upon 50 days of storage of the liquid fractions.

an advantage.

Ammonia emitted during storage of the liquid fractions was lower compared with the loss from raw slurry (Figure 2), and the NH_3 loss decreased with increasing $FeCl_3$ addition. This was probably a pH effect, i.e. the pH value was reduced by approximately one unit upon separation with a large amount of $FeCl_3$ compared to separation without $FeCl_3$. The result is in agreement with a previous study showing a reduction of NH_3 emissions from unseparated pig manure by addition of the coagulant $AlCl_3$ (Smith et al., 2004). Despite the fact that pH during the storage period was similar in the raw slurry and the liquid slurry fraction separated with

little $FeCl_3$, the NH_3 loss was approximately 60 % higher from the raw slurry. This may be explained by the content of ammonium, which was 3.8 g/l in raw slurry, but only 2.9-3.0 g/l in all of the liquid fractions.

Larger methane (CH_4) production was observed during anaerobic digestion of the solid fraction of separated slurry compared to raw manure (Figure 3). The CH_4 production did not improve using $FeCl_3$ as coagulant for the separation of the manure, neither on volatile solids or kg slurry basis. Previous separation experiments using $Fe_2(SO_4)_3$ at a rate corresponding to half the little $FeCl_3$ amount used in this study, also showed no significant effect on the biogas production (Møller et al., 2006). The reduced production upon large addition of $FeCl_3$ may be a result of the acidifying effect of $FeCl_3$ leading to increased inhibition of anaerobic bacteria by a shift towards protonised forms of VFAs.

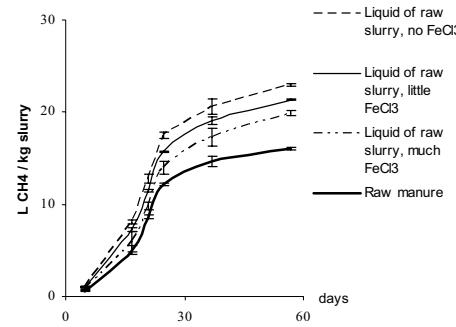

Figure 3. Total CH_4 yield from differently separated solid fractions.

Conclusion

By separation of pig slurry (6% dry matter) using a low concentration of $FeCl_3$, a flocculent and a mechanical belt separator, two fractions of the slurry were obtained: 1) a liquid fraction useful for fertilization of P rich agricultural land due to low P content and low NH_3 emission, and 2) a solid fraction useful for biogasification.

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Monitoring and optimisation of biogas production

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The use of biogas from anaerobic digestion (AD) of readily available bio-resources from agriculture and food sources can provide environmental and social benefits to the rural community. In mainland Europe biogas production has been successful. For example, in Germany there is a thriving AD sector with over 3000 biogas plants (compared to about 30 in the UK). There is a need to encourage the use of more efficient designs of plant, particularly the on-farm units. In the UK, biogas production needs to be made more efficient for realistic financial returns to encourage the implementation of new biogas plants.

The production of biogas by anaerobic digestion has lacked stability because of poor process monitoring and control, with commercial plants relying on reduced organic loading rates to ensure that a stable operation is maintained. There are two specific targets for the optimisation of anaerobic digestion: First, AD plants that charge a gate fee for the treatment of organic wastes will require a high organic loading rate, with gas production being of secondary importance. Second, the alternative optimisation strategy involves those processes that utilise an energy crop for gas production and will therefore require the highest gas output from a minimum organic loading rate. This study aims to make improvements in the monitoring of key indicators in biogas plants that would allow maximum input of materials and / or methane production without the possibility of process or microbial community failure.

A four–stage digester fed with a constant input composition was constructed from four 60 litre plastic containers connected in series and each situated in a 90 litre container to act as a water bath. The heating water was circulated between the four baths for five minutes every thirty minutes to ensure the temperature remained constant between the vessels. In order to maintain a consistent substrate composition, the feedstock chosen was a commercially available pig feed with a

composition of 72.5% carbohydrate, 16% protein, 4.5% lipid and 6.15% inorganic material. The digester was operated at mesophilic temperatures (38°C), and mixing of each vessel was by biogas recycling at a rate of 6 litres per minute for one minute in every hour. Feedstock was set at 5% total solids, and was pumped into the system by a peristaltic pump using the same one minute per hour timing schedule as the gas mixers. Flow of feedstock between vessels was by gravitational equalisation, the digestate level being controlled by the level of the output tubing. The connection tubes linking the vessels were positioned at the top to prevent leakage of heating water, therefore it was considered necessary to add a single baffle plate in each vessel to reduce the possibility of 'short circuiting' between the input and output. Control of the input feed pump and the gas mixing pumps was by relays, operated by National Instruments LabVIEW$_{TM}$ software *via* the computers parallel port.

By separating the process into four discrete vessels, and placing sampling points and probes between each vessel, a progression of the stages towards methanogenesis were detected. The system was monitored during start-up, failure and recovery of the anaerobic digestion process. Stabilisation of the optimum production of biogas has been identified as the next research step.

The digestate parameters of pH, redox and conductivity were measured online using conventional probes with 4-20mA outputs connected to a National Instruments data acquisition interface. The data was recorded using the LabVIEW$_{TM}$ software. The probes were fitted in flow cells, situated in the transfer tubing between the individual vessels, and included a temperature sensor for pH compensation (data not recorded). The flow cells each contained a ball-valve sampling point for removal of digestate samples for offline analysis. Offline digestate measurements of total bicarbonate buffering capacity, dry matter and organic matter were also made and recorded. Gas volume measurements were initially taken online using a single wet flow meter for the total output. This was later replaced by four individual water displacement vessels. Gas composition was measured offline by infra red for methane (0-100% volume) and by electrochemical cell for hydrogen (0-2000ppm). Both sensors were housed in a single instrument. Both online and offline data are to be compared with spectra obtained by Fourier transform near infra-red reflectance (FT-NIR) spectroscopy. The FT-NIR measurements of the digestate were used to investigate the start up and failure of the system. Measurements of

digestate for recovery and stabilisation were in reflectance mode and transmission mode, following the acquisition of suitable sampling equipment.

The optimum AD performance was determined using regression models developed from the large monitoring dataset. The model should predict the key process parameters and ultimately select the best 'soft' sensor algorithm to find the optima that produce the maximum methane output or allows the highest organic loading rate while maintaining a stable AD process. The optimum AD performance is a fine balance between the production of short chain fatty acids from the hydrolysis and acidogenesis stages and the buffering capacity of the methanogenic stage. A low pH is inhibitory to the methanogenic organisms, but if the buffer capacity is sufficient to prevent a drop in pH when fatty acid concentrations are high, gas production should be at a maximum. Results showed the optimum AD process for methane production was at an alkalinity of 6000 $mg.l^{-1}$ and process stability was above 2000 $mg.l^{-1}$ (Figure 1).

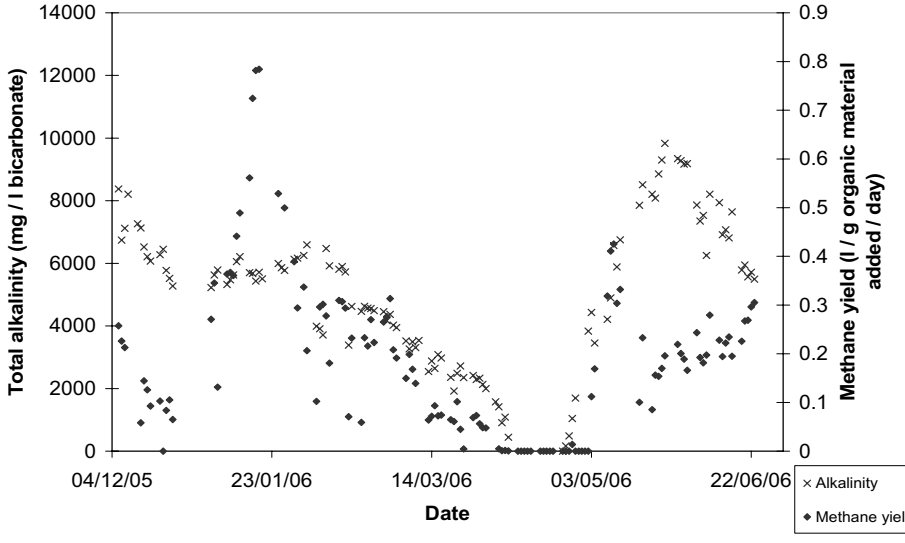

Figure 1. Vessel 3 (3rd stage) total bicarbonate buffering capacity and total methane yield.

The ability to determine the optimum performance by a rapid method of digestate analysis such as FT-NIR is a significant step forward as no sample preparation is required and measurement is complete in less than

two minutes. We were able to determine the alkalinity using FT-NIR spectroscopy with good accuracy and reliability (R^2 = 86.99%).

Future work involves investigating the microbial species present using phospholipid etherlipid (PLEL) analysis, and also by amplification of 16S DNA sequences and DNA sequencing. Also we will investigate the AD chamber hydrodynamics using a lithium tracer.

Anaerobic co-digestion of pig manure with fruit wastes. Process development for the recycling in decentralised farm scale plants

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Objective

The main objective was to study the utilization of fruit wastes, rejected from centralised fruit storage and distribution facilities of producers, as a co-substrate for co-digestion with pig manure (PM) in farm scale digesters. This paper presents the results of the lab work developed in batch trials and in continuous trials for the preparation of a site demonstration in a pilot plant of 2 m^3 CSTR digester.

Introduction

Fruit wastes (FW) are produced in large quantities in centralised fruit storage and distribution facilities of producers, during the selection and rejection processes before fruit enter into the market. The waste stream targeted by this project originated from a group of apple and pear producers. During the past few years these wastes have been landfilled and part of it used to produce ethanol, but still they constitute an environmental problem, and producers want to have an alternative.

One of the most promising alternatives for managing these organic wet wastes is anaerobic digestion (AD). A major limitation of anaerobic digestion of FW is a rapid acidification of these wastes decreasing the pH in the reactor, and a larger volatile fatty acids production (VFA), which can stress or inhibit the methanogenic biomass activity (Bouallagui et al, 2005). Decentralised management of these flows in farm scale digesters, could be a solution to avoid this limitation and improve the economy of digester investments already done in the past, and to contribute to the recycling of nutrients in the local agriculture areas.

The location of the fruit production unit is also a region where a strong pig production activity has developed. Pig production units in Portugal generate very dilute slurries, 1.5-2 % total solids (TS) with a total volatile

solids (VS) of 67%, and this represents a barrier to establish economically feasible AD processes.

Methods

Origin of materials - The substrates used were screened pig slurry (PS) from a farrow-to-finish pig farm and fruit wastes (FW) characterised by a mixture of refused flows of apples and pears. Samples of pig slurry were collected according to a procedure in order to get weekly composed samples. Fruit waste was pulped with a fruit mill. Inoculum was obtained from a mesophilic (35ºC) sewage digester. Characteristics of these materials are presented in Table 1.

Continuous trials – Continuous lab trials using a stainless steel digester (CSTR) with V= 11 litres were performed at 37 ± 4ºC. The digestion performance of FW:PS (v:v) composition (5%:95%, 10%:90% and 15%:85%) at HRT=16 d, was tested and compared with pig slurry digestion HRT =15 d (OLR = 0.66 kg VS/m^3.d^{-1}) and HRT= 11 d (OLR = 0.85 kg VS/m^3.d^{-1}). For each composition, the influence of the respective organic loading rate (1.0, 1.5, 2.0 and 2.95 kg SV/m^3.d^{-1}) on the main process operational parameters (methane, carbon dioxide, H_2S and COD fractions of the digestate) were investigated. Both substrates and mixtures were stored at –20º C before use.

Analytical methods – COD, TS, VS, lipids, TK-N, N-NH_4^+ and T-P were determined according to standard methods (APHA, 1992).

Table 1. Initial characteristics of the waste materials

		Pig slurry A	Pig slurry B	Fruit waste pulp
pH		7.77	7.42	3.49
TS	g/l	14.72	38.91	157.58
VS	g/l	10	28.48	154.17
Crude Fibre	g/l	2.53	-	13.97
Crude Fat	g/l	0.42	-	0.45
COD	mg O_2/l	16398	-	186960
COD soluble	mg O_2/l	8707	-	166050
TK-N	g/l	1.78	-	0.461
NH_4^+-N	g/l	1.04	-	0.110
T-P	mg/l	342.92	-	65.49

Results and discussion

It is possible to see from Table 1 the low TS content of pig slurry after the screening operation. In comparison, fruit waste pulp had almost ten times more solids and 98% of them were volatile. Buffiere et al. (2005) reported very similar characteristics for apple wastes.

Regarding biogas yield obtained from batch trials (50 days), the results were 0.878 l biogas/g VS and 1.051 l biogas/g VS, respectively, for FW and PS. Figure 2 illustrates the increase in the biogas production rate with different OLR. On the other hand, the observed (Table 2) biogas resulting from an increment of FW in the mixture became poor in methane, even for the mix FW15:PS85 (a).

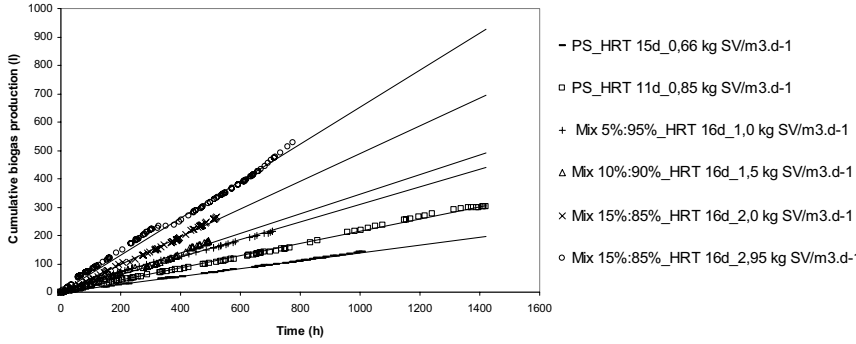

Figure 2. Biogas production rate at different OLR.

Table 2. Operating and performance data for different loading rates

Mix. FW:PS (% v/v)	HRT	OLR kg SV/ $m^3.d^{-1}$	Biogas l/h	Biogas quality % CH_4	COD removal %	m^3 biogas/m^3 biomass
0:100	15	0.66	0.139	73	64	4.55
0:100	11	0.85	0.214	73	70	5.14
5:95	18	1.0	0.309	69	68	10.79
10:90	17	1.5	0.347	69	67	12.11
15:85	16	2.0	0.490	58	77	17.11
15:85 (a)	16	2.95	0.653	58	69	22.79

(a) Mixture prepared with pig slurry B.

Conclusions

The utilisation of fruit wastes as a co-substrate during the digestion of pig slurry has a significant effect on the biogas production rate. Further

research focused on the biological pre-treatment of the fruit waste is being executed in order to evaluate operating and digestion performance after pre-treatment.

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In-vessel composting of food wastes in the UK: feedstock and output characteristics

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Composting is a key component in any strategy designed to meet targets for diversion of biodegradable municipal wastes under the Landfill Directive. However, while food wastes form a major component of this waste stream, only recently has composting been adopted as a disposal route for such material in the UK (TCA, 2005). To date, limited data has been available on the characteristics of these new feedstocks or the effects of different processing technologies on the quality of composted materials. This project was designed to examine the quality of composted food wastes from feedstocks to finished product.

Two main sites were selected for study: Castle Cary and Heathfield, in south west England. At the former, domestic kitchen waste is received in biodegradable bags and shredded with a mixed cardboard and green waste feedstock in a ratio of 1:2 food:co-compost (by volume). At the latter, the feedstock comprises mixed cardboard, green and food waste, which requires no further mixing before shredding and composting. The former site also composts green waste in conventional outdoor turned windrows (without aeration).

Food waste feedstocks at both sites are composted to the UK catering waste standard, which requires that material be reduced to a particle size of 400mm, which must then achieve a temperature of 60°C for forty-eight hours. The material must then be moved to a second 'barrier', where it must achieve 60°C for another forty-eight hours (OPSI, 2005). At Site 1, the partially composted material is then matured in turned (outdoor) windrows for three months prior to screening and use. At Site 2, the material is matured in the same way for only two to three weeks prior to screening and use. Both sites process material as batches in fixed clamps – floor channels allow continuous forced aeration with fresh or re-circulated air.

Table 1. Characteristics of different feedstocks and composts. Pathogens and pH quoted for fresh material, all others on dry weight basis.

		Inputs: Mean (standard deviation)		
Parameter	Units	Green + card	Food waste	Green waste
E. coli	cfu/g	2.57E+06 (3.94E+06)	6.16E+05 (3.41E+05)	5.11E+06 (9.09E+06)
Salmonella	in 25g	Absent	Absent	Absent
pH		6.54 (0.26)	4.40 (0.29)	6.98 (0.48)
N-total	g/kg	7.32 (1.13)	20.3 (4.03)	8.00 (0.32)
C:N		48.0 (10.5)	22.5 (3.08)	40.6 (7.91)
P	g/kg	1.30 (0.41)	3.32(1.06)	1.50 (0.21)
K	g/kg	5.60 (1.22)	6.84 (0.48)	5.86 (0.57)
Mg	g/kg	1.73 (0.36)	0.76 (0.03)	1.59 (0.36)
Ca	g/kg	16.8 (2.28)	9.89 (0.91)	20.1 (5.70)
Zn	mg/kg	94.6 (17.9)	19.1 (0.75)	76.7 (21.9)
Ni	mg/kg	30.1 (9.91)	5.82 (1.03)	43.8 (14.5)
Cu	mg/kg	33.6 (0.42)	11.1 (1.15)	25.5 (5.32)
Cd	mg/kg	0.73 (0.15)	0.79 (0.09)	0.50 (0.43)
Pb	mg/kg	47.4 (11.5)	2.64 (1.65)	40.7 (18.9)

		Outputs			
		Composted food waste	Composted food waste	Composted green waste	Horticultural peat
Maturation	months	3	0.5	3	-
E. coli	cfu/g	<10	<10	<10	-
Salmonella	in 25g	0	0	0	-
pH		8.50	7.46	8.73	4.24
N-total	g/kg	15.6	14.3	11.9	8.62
C:N		18.9	28.2	16.4	61.0
P	g/kg	2.53	2.17	2.74	0.13
K	g/kg	7.55	5.95	9.32	0.18
Mg	g/kg	3.68	2.12	3.34	1.45
Ca	g/kg	35.7	28.6	34.4	3.72
Zn	mg/kg	176	112	173	5.19
Ni	mg/kg	61.0	47.9	79.5	6.29
Cu	mg/kg	49.2	46.1	40.1	1.86
Cd	mg/kg	0.34	0.30	0.42	0.00
Pb	mg/kg	88.9	66.9	115	117

Representative samples of individual shredded feedstock materials were collected from each site every fortnight for two months. It was not possible to trace these batches through the entire composting process, so single batches of final compost were sampled. Fresh material was tested for moisture content, pH and microbiological parameters. Freeze-dried material was tested for all other parameters: laboratory tests followed standard methodologies as listed in PAS-100 (BSI, 2005). Some of the resulting data are presented in Table 1. Material was also tested for contamination by plastic, glass and other materials (BSI, 2005). These data are summarized in Table 2.

Table 2. Contamination levels in composted materials, compared with horticultural peat and a quality assurance limit (% air dry weight). *Limits for stones vary according to required end use of product

	Horticultural peat	Composted food waste	Composted food waste	Composted green waste	PAS-100 limit
Compost	99.3	88.0	73.1	91.4	-
Stones	0.73	6.91	2.98	8.57	8 or 16*
'Other'	0.00	5.12	24.0	0.03	0.5
Cardboard	0.00	4.10	23.5	0.01	-
Plastic	0.00	0.17	0.32	0.01	0.25
Metal	0.00	0.85	0.17	0.01	-

Salmonella spp. were found in 2% of feedstock samples, but were completely eliminated by the composting process. *E. coli* populations were also reduced sufficiently to allow all composts to pass the requirements of the Animal By-Products Regulations (UK Statutory Instruments 1482 (2003) & 2347 (2005)). Food wastes were found to contain higher concentrations of nitrogen than green wastes, and this persisted into the final composts. Concentrations of phosphorus were also higher in food wastes, but this was not shown to persist into the final composts. Other nutrient elements (calcium, magnesium, copper, zinc, nickel) were present in lower quantities in food than in green wastes. Use of green waste somewhat buffered these differences to produce final composts with similar concentrations. Potentially toxic elements (cadmium and lead) were present at much lower concentrations in food wastes, and this was reflected in the final composts – which readily exceeded the quality requirements of the PAS (Publicly Available Specification) 100 voluntary quality assurance standard (BSI, 2005). Some input characteristics

demonstrated wide variation during this study, and further research is recommended to examine the impacts of such variation on the nutrient content of composted food wastes over a longer time period. Research into the fertilizer replacement value, nutrient-release and physical characteristics of composted food wastes is also recommended.

Both food waste composts failed voluntary standards due to contamination by cardboard, and one would have failed due to the presence of plastic. Cardboard does not degrade sufficiently under current commercial composting regimes, and its removal from compost feedstocks is therefore recommended. Overall, composted food wastes met all PAS-100 pathogen and potentially toxic element limits, and fine-tuning of feedstocks should eliminate contamination to produce safe, high-quality composts.

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Behaviour of enteric micro-organisms in Canadian and French swine manure treatments

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Introduction

In spite of the effectiveness of manure treatments which reduce the transfer of nutrients to the environment, microbiological contamination of manure has become a great concern in Canada and European countries. There is indeed a sanitary risk related to the use of the effluents from piggeries. Even when healthy, pigs can excrete pathogenic micro-organisms which may be subsequently found in liquid manure. The pathogenic micro-organisms most often isolated from animal manure are *Salmonella*, *Campylobacter*, *Yersinia*, and *Cryptosporidium* (Guan and Holley, 2003; Heitman *et al.*, 2002).

Biological treatment of manure can reduce both indicators and pathogenic microorganisms (Côté et al., 2006). However, several parameters, including the composition of liquid manure (Nicholson et al., 2002), the time of storage, the temperature and the level of aeration (Hutchison *et al.*, 2005), influence the effectiveness of liquid manure treatment technology.

The aim of the present study was to compare antimicrobial effectiveness of 5 different proprietary treatments (3 Canadian and 2 French technologies). Microbiological analyses were carried out on samples of raw and treated manures after biological, chemical or physico-chemical treatments over periods of 6 to 12 months. The effectiveness of treatments was evaluated by comparing the extent of *E. coli*, enterococci, *Salmonella* and *Cryptosporidium* recovery.

Materials and methods

Five commercial technologies, presented in Table 1, were evaluated in Quebec (Canada) and in France. Samples were taken from the raw manure and from liquid by-products of the different steps of each process. Microbial analyses were done within 48 h after sampling. The results were

expressed in terms of the MPN (Most Probable Number) of bacteria detected per volume of sample. *E. coli* and enterococci were counted using Colilert™ and Enterolert™, respectively (IDEXX Laboratory). *Salmonella* was enumerated using an enrichment broth (Dulcitol-Selenite-Cysteine), followed by inoculation onto XLT4 agar and identification of typical colonies with API 20E strips (BioMérieux). *Cryptosporidium* were detected by filtration, immunomagnetic separation of the oocysts from the material captured, and an immunofluorescence assay for determination of oocysts.

Table 1. Description of the treatments.

Technologies	Step 1	Step 2	Step 3
Bioterre (Canada)	low temperature anaerobic digestion		
Biosor (Canada)	separation by floculation/ coagulation	bio-filtration	bio-filtration
Biofertile (Canada)	separation by a sieve and a screw press	aerobic digestion	electrochemical polishing[a]
Valetec (France)	separation by centrifugation	aerobic digestion	decantation
Balcopure (France)	separation by centrifugation	liming[b]	separation

[a]: treatment which allowed the purification of a liquid manure by-product by simultaneous electrocoagulation, electroflotation and electrooxydation.
[b]: volatilization of ammonia by stripping after addition of lime.

Results and discussion

Independently of the geographic location of the piggeries, variations in concentrations of indicator bacteria were observed in raw manures with mean values ranging between 5×10^3 to 10^5 per g for *E. coli* and 8×10^2 to 4×10^4 for enterococci (Table 2). In contrast, numbers of indicator organisms in liquid by-products intended for spreading ranged from undetectable to 38 bacteria per mL. As indicated by the high values of standard deviations, variations in numbers of bacteria also occurred within each piggery over the period of the study. However, except for centrifugation of manure of the two French processes, all steps of each process reduced the survival of both indicators, the concentrations of which decreased by at least 0.7 logarithmic unit after each treatment. The physico-chemical treatments such as polishing, stripping or settling increased the effectiveness of the processes. It was also clear that enterococci survived better than *E. coli* regardless the treatment. *Salmonella* were detected in all piggeries, but at variable frequency during the present study.

Table 2. Concentrations of bacteria and occurrence of *Cryptosporidium* in raw manures and in by-products from 5 processes.

Process (sampling period)		Type of manure or treatment	*E. coli*[b]		enterococci[b]		*Salmonella*[b]		*Cryptos-poridium* %[c]
			mean	SD[d]	mean	SD	mean (%[c])	SD	
Biofertile	6[a]	raw manure	$1.1\ 10^5$	$8.9\ 10^4$	$4.3\ 10^4$	$6.0\ 10^4$	57 (100)	$1.2\ 10^2$	33
(7 Sept 04		filtration	$6.9\ 10^3$	$7.8\ 10^3$	$3.3\ 10^3$	$4.9\ 10^3$	22 (83)	31	50
-29 Oct 05)		aerobic digestion	$1.0\ 10^3$	$1.3\ 10^3$	$6.3\ 10^2$	$6.6\ 10^2$	1.2 (67)	0.7	50
		polishing	24	32	<1.8		0.4 (50)	0.3	0
Biosor	11[a]	raw manure	$8.5\ 10^4$	$1.6\ 10^5$	$9.6\ 10^3$	$1.5\ 10^4$	$1.9\ 10^2$ (100)	$5.3\ 10^2$	45
(14 Sept 04		floc/coag[e]	$1.5\ 10^3$	$1.6\ 10^3$	$3.1\ 10^3$	$9.9\ 10^3$	27.7 (91)	64.9	18
-13 April 05)		bio-filtration	89	$1.8\ 10^2$	$5.9\ 10^2$	$3.0\ 10^2$	3.0 (64)	5.6	0
		bio-filtration	4.1	2.7	ND[f]	ND	1.3 (27)	0.9	0
Bioterre	10[a]	raw manure[g]	$1.7\ 10^4$	$3.5\ 10^4$	$4.1\ 10^3$	$5.1\ 10^3$	50 (56)	73	40
(7 Sept 04		raw manure[h]	$7.0\ 10^4$	$7.4\ 10^4$	$8.1\ 10^2$	$9.7\ 10^2$	$2.9\ 10^3$ (100)	$5.3\ 10^3$	0
-27 April 05)		anaerobic digestion	$1.2\ 10^2$	$1.0\ 10^2$	$2.0\ 10^2$	$2.9\ 10^2$	1.9 (30)	1.8	40
Valetec	10[a]	raw manure	$5.1\ 10^3$	$3.7\ 10^3$	$1.6\ 10^3$	$1.2\ 10^3$	0.2 (30)	0.0	70
(14 June 05		centrifugation	$3.6\ 10^3$	$1.6\ 10^3$	$1.4\ 10^3$	$1.1\ 10^3$	45 (44)	90	55
-8 Nov 05)		aerobic digestion	64	38	$2.3\ 10^2$	$1.1\ 10^2$	0.2 (20)	0	50
		settling	13	12	38	70	0.4 (10)	0	10
Balcopure	10[a]	raw manure	$6.8\ 10^3$	$1.7\ 10^4$	$8.5\ 10^2$	$4.4\ 10^2$	10 (90)	19.5	70
(21 June 05		centrifugation	$2.8\ 10^3$	$6.1\ 10^3$	$6\ 10^2$	$6.4\ 10^2$	7.9 (60)	15.8	80
-8 Nov 05)		liming	9.2	11	<24		0.3 (20)	0.1	50

[a] number of samples for each step of the process; [b] bacterial numbers-MPN/g wet weight ; [c] percent of samples positive ; [d] Standard Deviation ; [e] floculation / coagulation ; [f] Not Determined ; [g] manure from finish barn ; [h] nursery pig manure

The concentrations of *Salmonella* in raw manure varied from 0.2 to 2.9 × 10^3 MPN g^{-1}. The higher concentration which was observed in manure from nursery barns was also reported by Hutchison *et al*. (2005). Regardless of the initial level in manure, the concentration of *Salmonella* did not exceed 2 bacteria per g in the final liquid by-product. The presence of *Cryptosporidium* which was detected in the final liquid by-product from 3 of the 5 piggeries confirms the existence of sanitary risk.

Conclusion

The manure treatments studied, initially designed for the removal of nitrogen and phosphorus, make it possible to decrease the level of enteric bacteria from 1 to 4 logarithmic units, but they do not achieve complete sanitisation of the by-products which are intended for spreading on agricultural land.

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Managing manure by a greater understanding of its metabolic profile? Adopting new technologies

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To improve the management of organic resources to be spread to land or biogas output from an anaerobic digester requires better understanding of the biochemical processes that are performed by the microbial community. More efficient treatment processes are demanded by society and better quality products are required for farmer acceptability if we are to meet the need to spread more and new organic resources to land. Current research focuses on the emission of small molecules such as NH_3, CH_4, N_2O, volatile fatty acids (VFAs) and indirectly other odorous molecules by olfactometry. However, to improve our understanding of degradation processes requires that we look at large numbers of larger molecules and the diets used to produce the manure and metabolic profile.

There are a range of analytical technologies for measuring complex manure decomposition products or metabolic profiles including gas or liquid chromatography linked to a mass spectrometric detector. Often multidimensional chromatography can resolve more complex mixtures, and Fourier Transform Near Infrared Reflectance (FTNIR) spectroscopy and Nuclear Magnetic Resonance (NMR) have more rapid scanning capabilities. The objectives of producing a large dataset obtained from studying a biological system can be to 1) determine the variation of the decomposition process, 2) to optimise the system by mining the data for relationships, and 3) to investigate individual stages of bioresource systems, e.g. fresh manure to determine dietary and genetic effects on manure composition.

In this study, manure samples were produced from 8-month old lambs fed four different diets [1]. Manure (as both urine and faeces) was collected on three separate 14-day periods from Suffolk-cross lambs fed ryegrass, peas, kale, red clover, or lucerne as silage. There were 3 replicates for all samples, except for peas (one sample) and kale (2 samples). We investigated several analytical technologies to see if they differentiated the diets from the resulting analysis of manure samples. Samples were

analysed for amino acid profile after acid hydrolysis, emissions of volatile organic compounds, metabolic products using derivatisation gas chromatography-mass spectrometry (GC-MS) and spectral information from ^{1}H-NMR. The manure microbial community structure was obtained by comparing the extracted phospholipids fatty acids (PLFAs) present in the cell walls of viable micro-organisms.

The resulting datasets were analysed by multivariate techniques, especially principal component analysis (PCA) that was used to reduce the size of datasets for similarity tests within and with other datasets. The first component or dimension from such PCA treatment contains information describing the largest variation in the original dataset and each subsequent component describes the next highest portion of variation. The datasets reduced to the same sizes were compared using the Mantel test. The Mantel test determines the extent to which two similarity/distance matrices describe the same relationships among for example the metabolic products and microbial profile which can be measured by comparing their off-diagonal elements as a matrices or visual blocks within a triangular shape.

PCA enables data from the analysis of the amino acid content of each sample to be shown in 3D graphical format (Figure 1). The biplot in Figure 1 shows the first 3 principal components of the amino acid analysis of the diet, but also the effect of diet on the amino acid composition. Note the first principal component describes most (over 95%) of the variation in the manure samples analysed.

We can compare the data with others in similarity matrices as shown in Figure 2. Figure 2 shows that manure from diet replicates 1,2 and 3 differed from manure from diet replicates 4,5 and 6 by the different 3x3 square areas. We can compare the analysis datasets with other datasets of a different size, but we need to reduce the dimensionality, in this case to 12, by PCA analysis Comparisons were performed using Mantel test for similarity. The different chemical analysis proved mostly to be similar. However, because of the variability of the analytical data from such a small dataset we estimate that relationships between analyses could be identified with larger sample numbers.

The results showed differences in manure composition for different diets with a range of analytical measurements. Datasets were large with over

100 metabolic by-products present in manure. Decreasing effects of diet were shown for the different analyses in the order: amino acid > metabolic > VOC > NMR. Specific molecules that are indicative of a particular decomposition process they were the amino acids, the isoflavonoids (clover), and the hydrocarbons nonacosane and

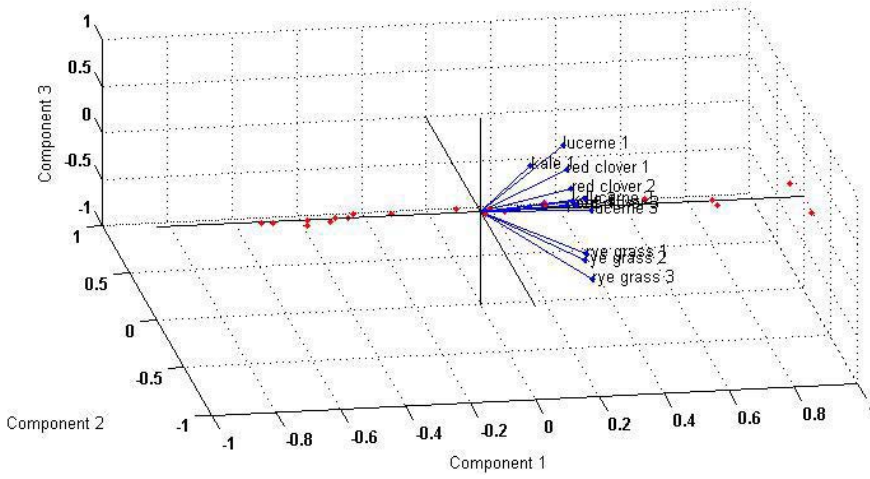

Figure 1. A Biplot of amino acid composition in the manure samples showing the different dietary sources.

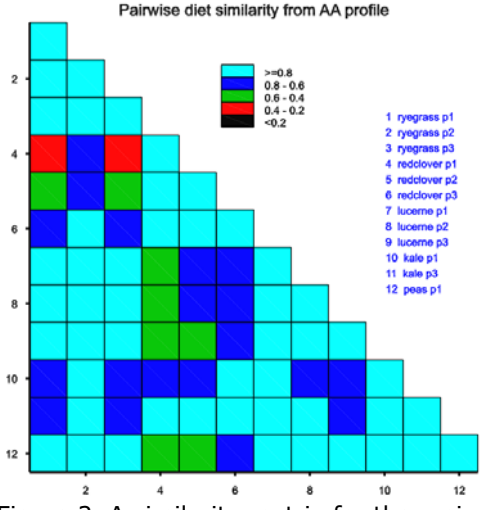

Figure 2. A similarity matrix for the amino acid profile of each manure type.

heptacosane. The amino acids cystine and cysteine identified the kale diet; aspartic acid, glutamic acid, glycine and alanine the red clover diet. The VOCs such as the methyl sulphides and some alcohols were significant in identifying the dietary intake.

Our study shows that we can characterise manure by showing the effect of diet. The next stage is to identify the effect after landspreading, for example monitor the chemical species to recognise the build up of organic material in the soil, especially those compounds that enable the retention of nitrogen and other nutrients.

Development of Anaerobic Ammonium Oxidation (Anammox) technology using immobilized biomass from swine manure

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Research was conducted to develop process applications for anaerobic ammonium oxidation (anammox) bacteria acclimated to animal wastewater conditions using microbial immobilization techniques. In the anammox reaction, under anaerobic and autotrophic conditions, ammonium (NH_4^+) serves as the electron donor using nitrite (NO_2^-) as the electron acceptor resulting in production of harmless di-nitrogen (N_2) gas (Strous et al., 1998).

Traditionally, removal of nitrogen (N) from wastewaters uses a combination of nitrification and denitrification. Nitrification consumes large amounts of oxygen to convert NH_4^+ into nitrate (NO_3^-), while denitrification requires addition of organic carbon to convert NO_3^- into N_2 gas. In the anammox alternative approach, NH_4^+ is converted to N_2 gas in a totally autotrophic process (without organic carbon) by two sequential reactions: partial nitritation (eq. 1) and anammox (eq. 2, Strous et al., 1998).

$$2\ NH_4^+ + 1.5\ O_2 \rightarrow NH_4^+ + NO_2^- + H_2O + 2\ H^+ \qquad (1)$$
$$1\ NH_4^+ + 1.32\ NO_2^- \rightarrow 1.02\ N_2 + 0.26\ NO_3^- + 2\ H_2O \qquad (2)$$

Compared to conventional nitrification-denitrification, this combined pathway saves more than 50% of the oxygen supply for nitrification and 100% of the external organic source for denitrification. This leads to a significant reduction in energy needs of treatment, and a decrease in operational costs. In addition, by-products do not include greenhouse gases.
The isolation of anammox adapted to animal wastewater environments can be of significant importance to farming systems, because excess ammonia in modern, industrial-type livestock production is a global

problem, and the use of conventional biological N removal methods is usually hindered by cost; thus, we think that the more economical anammox based treatment can greatly facilitate adoption of advanced wastewater treatment technologies by farmers.

The anammox cultures were successfully established using continuous-flow unit processes and biomass carriers seeded with sludges of manure origin. The sludges containing anammox bacteria were obtained from three swine farms: Two were located in North Carolina, USA, and the other in Santa Catarina, Brazil. The sludges were obtained from diverse environments in the swine farms: an aerobic nitrification tank treating anaerobically-digested swine manure, an anoxic denitrification tank treating liquid swine manure after solid-liquid separation, and sediment in an old (inactive) anaerobic lagoon used to treat swine manure. Laboratory bioreactors were seeded with the manure sludges after acclimation with nitrate solution to remove endogenous carbon.

In a first phase of the research, two sequential experiments were conducted over a 5-year period (2000-2005) to investigate conditions to 1) isolate anammox bacteria from the swine sludges, and 2) optimize anammox treatment. The bioreactors were operated in continuous flow and contained polyvinyl alcohol (PVA) hydrogel biomass carrier beads for immobilization and enrichment of the slow growth microorganisms. A distinct red biomass growth, which is typical of the anammox planctomycete bacteria, developed in the reactors only when high N (300 mg/L) and salt strength concentration was used in the synthetic enrichment medium. Under these conditions and protocol, it took about 100 days for the anammox reaction to develop from farm sludges obtained in both USA and Brazilian farms.

Removal of NO_2^- and NH_4^+ was simultaneous at the stoichiometric ratios shown in fig. 1 and summarized in eq. 3.

$$NH_4^+ + 1.24\ NO_2^- \rightarrow 1.00\ N_2 + 0.24\ NO_3^- \qquad (3)$$

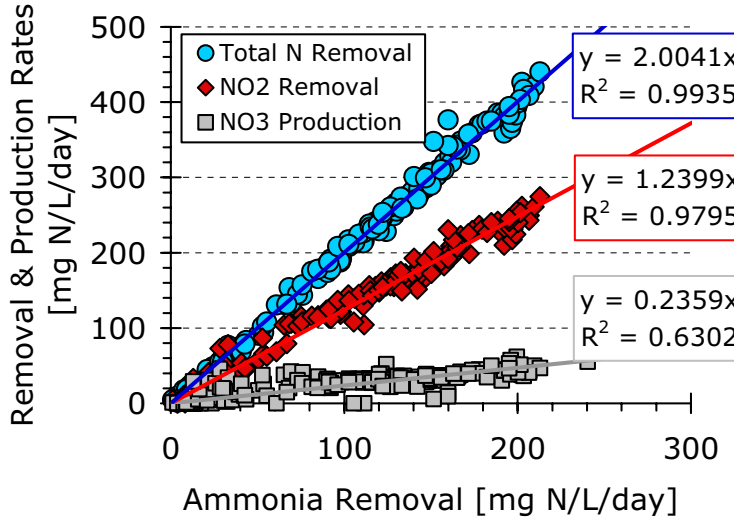

Figure 1. Stoichiometric ratio of the anammox isolated from swine manure.

Fluorescent in situ hybridization (FISH) analysis of the biofilm using 16S rDNA oligonucleotides Ana-1 and Amx 820 confirmed Anammox bacteria; imaging revealed a high density of cells growing in clusters. Nitrogen removal rate obtained during first-phase development was 0.7-0.8 kg N/m^3/day, which is in the range of industrial bio-treatment applications. In a second phase of the research, new reactors were seeded directly with the anammox red sludge biomass produced before. As biomass carrier, the reactors used either a polyester non-woven material coated with pyridinium type polymer (pilot reactor) (Furukawa et al., 2003), or a net type acryl-resin fiber material (bench reactors), both materials designed to enhance retention of microorganisms. Removal of NO_2^- and NH_4^+ at typical ratios of the anammox reaction occurred from day 1 of operation of the reactors. Nitrogen removal rates increased exponentially with time in the reactors seeded with the anammox enriched sludge. These findings overall may lead to development of more economical treatment systems for livestock wastewater and other effluents containing high ammonia concentration.

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The effect of bioaugmentation on selected microbial parameters of swine slurry pits

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Bioaugmentation is a promising methodology for treatment of all types of wastes. It allows *in-situ* treatment of pig slurry and can also be applied at different steps of their handling. Commercially available products are often used in pig farms in Spain as a pre-treatment in slurry pits. Their composition is complex and range from pure mineral salts to, presumably, selected bacteria. Efficiency of these products with regard physico-chemical variables has proved to be very high in reducing organic matter, odours and helping slurry management (see e.g. McCroy and Hobbs 2001 for a review). Masking, disinfecting, and oxidizing agents can provide a short-term control of odour from livestock wastes, but the use of microbial-based additives is not recommended (McCroy and Hobbs 2001). Generally, bacterial additives have proved to have little effect in pig litter systems, as there are sufficient endogenous populations of bacteria (Tiquia 1996). Tan (1995) found that the use of microbial compost additives showed no significant advantage against a control without additive. Although disinfection is also among their claimed properties, almost no data are available on their efficiency for removal of faecal bacteria under *in-situ* conditions.

Materials and methods

Two farms in the Castilla-León region (north-west central Spain) were sampled on different days after routine addition of Actilith®.This additive is formed by litothamne (marine carbonate), bentonite, algae powder and cellulolytic bacteria. Faecal coliforms (FC, ufc/100 ml), faecal streptococci (FS, ufc/100 ml), *E. coli* (EC, ufc/100 ml), *Clostridium perfringens* (CP, ufc/ml) and staphylococci (ST, ufc/100 ml, only in spring at Farm A) were analysed on each of the sampling occasions following conventional standard procedures. Pit slurries were differentiated in Farm A depending on the origin of the slurry (gestating sows or farrowing sows), the season (spring or summer) and days after addition of Actilith® according to the

recommended procedure. A control pit without additive was not available with gestating sows at farm A. At farm B, additive was added monthly to pits with slurry from the finishing pigs. Samples were taken after 2 and 3 months of treatment and compared with another pit without treatment (control). Chemical oxygen demand (COD), total suspended solids (TSS) and total Kjeldahl nitrogen (TKN) were also analyzed in the slurry following standard methods.

Results

With regard to Farm A, treatments realised in spring showed a decrease over time (gestating pit) or relative to the control (farrowing pits) of up to 3 log units (Fig. 1). Treatments applied in summer also showed the aforementioned reductions with the exception of gestating pits, where some bacteria, like FS, increased up to 0.5 log units. Bacterial densities were generally higher in summer, independently of type of treatment or origin of slurry. Additive was only applied in summer at Farm B, and a general increase in bacterial counts was observed with time, especially in FC.

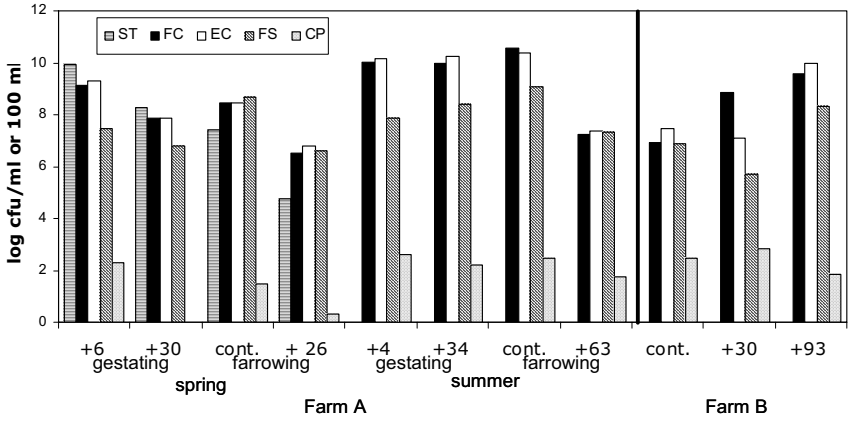

Fig 1. Bacterial counts in two farms after Actilith® addition under spring and summer conditions for gestating and farrowing sows (Farm A). + means days after Actilith® addition.

Patterns of physico-chemical characteristics (Table 1) were similar to those found for bacteria. Additive amendment seemed to give a better reduction in COD and TSS in spring than in summer at Farm A. Results from Farm B could have been affected by the recent addition of slurry to the pit some days before the last sampling. The additive may help the hydrolysis of organic matter, giving an increase in organic soluble

compounds. This increase in readily biodegradable organic matter would support growth of bacteria, including faecal groups. Therefore, a relationship would be expected between slurry hydrolysis and bacterial densities.

Table 1. Physico-chemical characteristics (g/l) of the slurry on selected days (+) after Actilith® addition. Cont.: control pit without additive.

	Farm A								Farm B		
	spring				summer						
	gestating		farrowing		gestating		farrowing				
	+6	+30	Cont.	+26	+4	+34	Cont.	+63	Cont.	+30	+93
TSS	38	11	15	4	18	14	7	5	44	10	43
COD	44	19	45	9	30	27	16	23	108	54	104
TKN					15	6	2	4	12	5	16

In this case a decrease in both bacterial counts and organic matter was observed only in spring at Farm A. The rest of the results do not provide clear evidence for an effect of the additive on faecal bacteria and their related organic matter concentration.

Conclusions

Despite the presence of cellulolytic bacteria in the additive, the experiments have provided evidence to support the conclusion that additives containing bacteria do not grow in livestock manure, even in the absence of competing indigenous micro-organisms (Hobbs and Merry 2000). A better option should be to stimulate the growth of indigenous bacteria, mostly belonging to the faecal group, by using hydrolytic additives or stimulating the bioavailability of important oligonutrients (see e.g. Coates et al. 2005).

The efficiency of bioaugmentation under practical conditions at pig farms seems to be highly dependent on the season and patterns of slurry application to the pits.

Acknowledgements

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Strategies to reduce diffuse nitrogen pollution from cattle slurry applications

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Introduction

Around 90 million tonnes of farm manures supplying 450,000 tonnes of nitrogen (N) are applied to agricultural land in the UK each year. These applications need to be carefully managed to minimise N losses to the air and water environments. Nitrate Vulnerable Zone legislation, which covers 55% of agricultural land in England, prevents autumn applications of slurry and poultry manure on sandy and shallow soils in order to reduce nitrate leaching losses. Bandspreading equipment (e.g. trailing shoe and trailing hose machines) can be used to apply slurry evenly to growing crops in spring, with reduced ammonia emissions and crop contamination compared with surface broadcast spreading. However, reductions in nitrate leaching losses by moving slurry applications from autumn to spring may be offset by increased ammonia losses, as a result of warmer spring/summer temperatures and reduced slurry infiltration rates into the soil compared with autumn application timings. There is a need to develop integrated slurry management strategies that maximise crop N recovery and minimise losses to the wider environment.

Methodology

Experiments were set up on free draining sandy soils on a commercial mixed dairy/arable farm in Cheshire (England). Slurry was applied at different timings to grassland before first and second cut silage in harvest season 2002/03, and to winter wheat in harvest season 2004/05 (Table 1). There were three replicates of each application timing and an untreated control arranged in a randomised block design. The slurry treatments were applied using a 11m^3 Joskin slurry tanker fitted with a 12m trailing hose boom to plots 24m x 24m.

Ammonia emissions were measured for 7 days after each slurry application, using the micro-meteorological mass balance technique. Nitrate leaching losses were measured from the autumn application timings and the untreated control, using porous ceramic cups (10 per plot) installed at 60-90cm depth. Nitrate-N concentrations in the porous cup water samples (collected after every 25mm of drainage) were combined with estimates of over winter drainage volumes to calculate nitrate leaching losses (kg/ha). Crop dry matter yields and N uptake were measured at harvest (May and July 2003 for the grass, and August 2005 for the winter wheat).

Table 1. Slurry application timings

Harvest year	Crop	Application timing
2003	Grassland	October 2002, February, March and April 2003 (before first cut) Early June and late June 2003 (before second cut)
2005	Winter wheat	November 2004, January, February, March and May 2005

Results

On grassland, ammonia emissions (Figure 1a) were highest ($P<0.05$) following the slurry application in early June (before second cut) at 17% of the total N applied and lowest following the October timing at 4% of the total N applied. The higher ammonia losses following the early June timing were most probably due to a combination of higher soil temperatures (15°C), 'dry' soil conditions and lack of grass cover (<5cm height), compared with the other application timings (soil temperature range 1-10°C before first cut, and grass heights > 7.5 cm).

Nitrate leaching losses following the October application (208 mm of drainage) at 69 kg/ha N were not different from the untreated control ($P>0.05$) at 64 kg/ha N, with losses equivalent to 3% of the total slurry N applied. The low leaching losses were probably a reflection of the uptake of slurry N by the grass sward following application. There was no effect of slurry application ($P>0.05$) on either yield or grass N offtake at harvest for either the first or second cut silage.

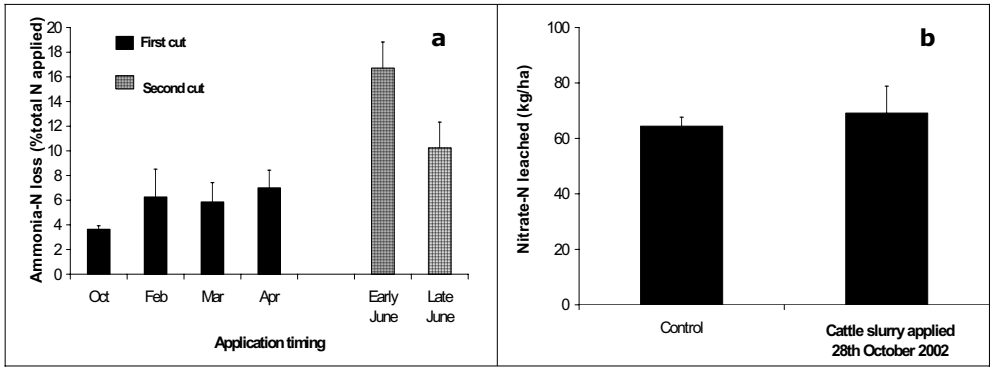

Figure 1. Ammonia emissions (a) and nitrate leaching losses (b) following cattle slurry applications to grassland 2002/03.

On arable land, ammonia emissions were greatest ($P<0.05$) following the March application timing at 18% of total N applied, and lowest following the May application at 5% of total N applied (Figure 2a). Emissions from the other three application timings ranged between 7% and 11% of the total N applied. The higher emissions from the March application were most probably a reflection of reduced slurry infiltration rates into the 'capped' sandy soil.

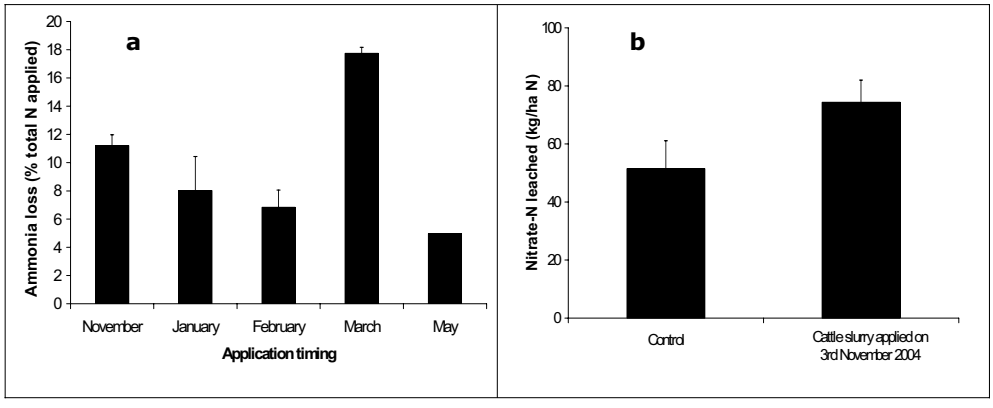

Figure 2. Ammonia emissions (a) and nitrate leaching losses (b) following cattle slurry applications to arable land 2004/05.

Nitrate leaching losses following the November application timing (127 mm of drainage) were 23 kg/ha higher ($P<0.05$) than from the untreated control (Figure 2b), with losses equivalent to 20% of total slurry N applied. The elevated leaching losses were largely a reflection of low N uptake by the cereal crop in the period after application up to the end of

winter drainage. Grain N offtakes were higher from the January-May application timings (10-21% total N applied) compared with the November application timing (<1% total N applied), reflecting nitrate leaching losses following the November application timing, although it was not possible to confirm the differences statistically ($P>0.05$).

Conclusions

There is a need to ensure that slurry management practices that aim to reduce nitrate leaching losses (i.e. moving from autumn to late spring/early summer application timings) do not exacerbate ammonia emissions under warmer conditions and where slurry infiltration rates into soil are reduced (so called 'pollution-swapping'). An integrated approach to slurry N management is needed that considers all N loss pathways and aims to maximise crop N recovery.

Acknowledgement

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Nitrous oxide emissions from riparian buffers and treatment wetlands

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Riparian buffers and treatment wetlands are becoming widely used for protection of water bodies from nonpoint pollution. They are one of the main farm management tools for protecting against the deleterious impacts of livestock manure. Their wet nature makes them especially suited for removing excess nitrogen via denitrification. Yet, there are questions about the potential impact of these wetland landscape components relative to denitrification and nitrous oxide production. Incomplete denitrification carries the potential for significant nitrous oxide production and the associated harmful air quality implications (Davidson et al., 2000). Thus, it is important to know if riparian and treatment wetland denitrification typically proceeds to completion with the production of di-nitrogen gas, or if the denitrification is incomplete (stopping at nitrous oxide). Furthermore, it is important to know if riparian/wetland nitrous oxide production is exacerbated by application of livestock manures to contiguous fields.

Relatively few studies of riparian denitrification consider the production of nitrous oxide. Wick et al., (2001) found higher nitrous oxide production in both forested and pasture lands in Brazil when soils were wet. Dhondt et al. (2004) conducted an investigation to determine the extent of potential nitrous oxide production in three different types of riparian zones (mixed vegetation, forest, and grass) of the Molenbeek River of Belgium. Their focus was to assess the tradeoff of denitrification to improve water quality vs. potential air quality degradation via nitrous oxide emissions. They concluded that observed nitrous oxide emissions in riparian zones were not a significant "pollution-swapping phenomenon."

These investigations provide insight into denitrification in agricultural and forested riparian buffers, but they do not provide definitive conclusions. The objectives of this research were to 1) ascertain the level of potential nitrous oxide production in riparian buffers and 2) identify controlling factors for nitrous oxide emissions.

The study was conducted within the Herrings Marsh Run Watershed in North Carolina (Stone et al., 1995). Soil samples were obtained from seven distinctly different agricultural and riparian landscape positions. Soil samples were collected in March 2004 and August 2005. Samples (5-cm diameter x 15.2-cm length) were collected from three depths at each site: (*i*) at the upper 15 cm of the soil surface; (*ii*) midway between the soil surface and the water table; and (*iii*) 15 cm above the water table. Denitrification enzyme activity (DEA) was measured on all samples by the acetylene inhibition method (Tiedje, 1994).

The mean DEA (with acetylene) was 80 µg N_2O-N kg^{-1} soil h^{-1} (Std. Dev. ± 136) for all 283 soil samples from the entire watershed. If no acetylene was added to block conversion of nitrous oxide to di-nitrogen gas, only 15 µg N_2O-N kg^{-1} soil h^{-1} (Std. Dev. ± 44) was accumulated. The median value of DEA was 37 µg N_2O-N kg^{-1} soil h^{-1}. However, the median value for the nitrous oxide accumulated without addition of acetylene was 0 µg N_2O-N kg^{-1} soil h^{-1}. The site with the highest DEA was the riparian buffer contiguous to a swine wastewater sprayfield (132 µg N_2O-N kg^{-1} soil h^{-1}). The site with the lowest DEA was a forested riparian zone at exit of watershed (25 µg N_2O-N kg^{-1} soil h^{-1}).

Without acetylene, nitrous oxide accumulation was found to be essentially zero (Fig. 1) in the control treatment when the soil C/N ratio exceeded 25. This ratio of 25 was a threshold, and significant nitrous oxide production occurred only in soils with lower C/N ratios, particularly below 20. Similar suppression of nitrous oxide production from soils has also been recently reported by Klemedtsson et al. (2005) when the soil C/N ratios were > 25. They found that soil C/N ratio could be used as a parameter to predict nitrous oxide production in forested histosols of northern Europe. They used a different technique (chambers emissions) in a different ecosystem. Yet, they found very similar results. The findings of our investigation further indicate that soil C/N may be a robust threshold controller of nitrous oxide production in different ecosystem and management conditions. Our current research is focused on better understanding the extent of and controlling factors for nitrous oxide production in riparian/wetland systems.

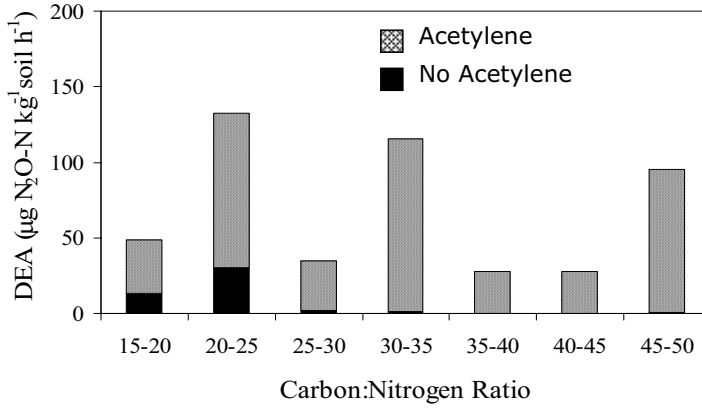

Figure 1. Denitrification enzyme activity in riparian buffers of the eastern Coastal Plain of the USA.

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Land application of solid and liquid fractions from sedimented dairy slurry

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Abstract
This study evaluates Decanted and Sedimented sludge fractions from settled dairy slurry for use on crops. The Decanted portion emitted less NH_3 and N_2O and enhanced crop N uptake relative to whole slurry. The sediment fraction provided starter P for corn. This low cost approach may help medium sized producers improve slurry utilization.

Introduction
As livestock feeding facilities increase in size, land for application of manure becomes more remote and transport costs rise (Fulhage, 1997). Fields near feeding facilities tend to become loaded with nutrients, particularly P, so transportation distances must increase even more. Transportation of dairy slurry is costly because of its low nutrient concentrations (4 g N and 1 g P kg^{-1} manure) (Moreira and Satter, 2006). It is usually not possible to meet the N requirement of crops using only manure without adding excess P to the soil, because the ratio of N:P required by crops exceeds the ratio available in manures (Schroeder and Stevens, 2004). The N:P ratio of excreted manure declines in animal housing and storage due to gaseous losses of N, and further gaseous N losses occur after field application. Even with N-conserving surface banding of dairy slurry, maximum yield of grass could not be sustained without accumulating P in soil (Bittman et al., 2005).

Separating solids from slurry can reduce transportation cost because nutrients are more concentrated in solids and hence cheaper to transport. The liquid fraction often has a greater N:P ratio than whole slurry, so that a greater volume may be applied without accumulating P. Better flow properties of the liquid fraction facilitates soil infiltration and reduces NH_3 volatilization (Stevens and Laughlin, 1997). Techniques available for removing solids range from passive (e.g. sedimentation) to mechanical (e.g. filters and centrifuges) to advanced chemical agents (Møller et al., 2000). Separation efficiencies and costs are correlated, so farmers need

cost-effective strategies suited to their operations. The goal of this project was to assess the efficacy of N in the decanted fraction and P in settled fraction obtained from sedimented dairy slurry.

Materials and methods

The study was conducted in south-western British Columbia, Canada using dairy slurry manure from a freestall barn with sawdust bedding, containing 7% dry matter (DM), 2.4 g kg^{-1} total-N, 1.2 g kg^{-1} total ammoniacal N (TAN) and 0.5 g kg^{-1} P. The slurry sedimented over winter in a 2.5-m deep tank covered with a roof. After the storage period, the surface crust was removed and the upper 1.2-m of liquid was decanted. This fraction, called Decanted, contained approximately 2.7% DM, 1.8 g kg^{-1} total-N, 0.9 g kg^{-1} (TAN) and 0.18 g kg^{-1} P. A 0.6-m thick layer at the bottom of the tank, called Sediment, contained 6.6% DM, 2.6 g kg^{-1} total-N, 1.2 g kg^{-1} TAN and 0.44 g kg^{-1} P. Decanted and Whole manures were applied by broadcasting or banding over aeration slots (called sub-surface deposition), prior to planting Italian ryegrass (*Lolium multiflorum* L.). The treatments were tested for N-uptake and losses of NH_3 (with wind tunnels) and N_2O (with vented chambers). The mean of the two application techniques is reported. The Sediment fraction was placed in 15-cm deep furrows centered 10- or 15-cm from the corn rows at planting.

Table 1. Comparison of emissions of NH_3 and N_2O and crop N-uptake from Whole and Decanted dairy slurry applied in spring, summer and fall, 2005.

	Spring	Summer	Fall	Mean
Ammonia Emission	---------% of applied TAN---------			
Whole slurry	33a	31b	31a	32
Decanted slurry	27b	38a	13b	26
N_2O Emission	----------g N_2O-N ha^{-1}----------			
Control	319b	112b	378c	270
Whole slurry	1224a	584a	1368a	1059
Decanted slurry	953a	571a	883b	803
Herbage N-Uptake	----------kg N ha^{-1}----------			
Control	NA[2]	42b	NA	
Whole slurry	80b	74a	NA	77
Decanted slurry	101a	70a	NA	85

[1] Values within columns and variables, followed by same letter are not different at $P<0.05$; [2] not available.

Results and discussion

Decanted manure emitted 15 and 58% less NH_3 than Whole manure in spring and fall, respectively, but 23% more NH_3 in mid-summer (Table 1). Reduced emissions in spring and fall can be explained by faster infiltration of the thinner Decanted fraction into the soil; summer results were unexpected but may be due to drying and crusting of the surface of the Whole but not Decanted slurry (Stevens and Laughlin, 1997). Whole manure generally produced more N_2O emissions than Decanted fraction, but the effect was significant only in fall, perhaps due to higher rainfall. Greater herbage uptake of N from decanted slurry was observed in spring but not in summer, when loss of ammonia was comparatively high.

Injecting the Sediment fraction increased P uptake at the critical 6-leaf stage by 46 and 118% over control from bands placed 15- and 10-cm from the seed (Table 2). These 15- and 10-cm treatments provided 49 and 73%, respectively, of the P supplied by an equivalent rate of chemical fertilizer. With small amounts of fertilizer (7 kg P ha^{-1}) placed in the seed furrow, P in manured corn (10-cm treatment) fully matched that of fertilized corn, saving of about 23 kg ha^{-1} of fertilizer P. In addition, the injected slurry would emit little ammonia, although it may emit N_2O.

Table 2. Response of silage corn to P fertilizer and dairy slurry sediment

Treatment	Distance (cm)	P applied (kg ha^{-1})	P uptake 6-leaf stage (kg ha^{-1})	Whole plant yield (t ha^{-1})
Control	--	0	0.41 d[1]	11.3 b
Fertilizer	5	33	1.24 a	13.2 ab
Slurry sediment	10	33	0.91 b	13.6 a
Slurry sediment	15	33	0.61 cd	13.3 ab
Slurry sediment+fertilizer[2]	10	40	1.21 a	14.0 a

injected 15-cm deep at 10- and 15-cm from the seed furrow

[1] Values in columns followed by same letter are not different at $P<0.05$

[2] P fertilizer at 7 kg ha^{-1} applied in seed furrow

Conclusion

Slurry sedimentation produced a Decanted fraction with low nutrient concentration that can be applied at higher rates near the feeding facility with less loss of N (higher effective N:P ratio). The Sediment can be injected into the soil to supply starter P to corn, replacing mineral fertilizer

that is often used. Our study shows that it may be possible to improve the sustainability of manure utilization for crops with little additional costs.

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Utilization and losses of nitrogen and phosphorus from field-applied slurry separation products

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High livestock densities in agricultural areas are frequently accompanied by surpluses of N and P leading to over-fertilisation and environmental risks. Separation of animal slurry into a solid and a liquid fraction improves the opportunities for distributing especially the solid fraction to nutrient-poor areas. Slurry can be separated into a solid and a liquid fraction by several techniques, e.g. by centrifugation. The dry matter content of the solid fraction (SOL) of pig slurry is typically 25-35 %. It contains 60-80 % of the dry matter and phosphorus content of the original slurry, but only 20-25 % of the nitrogen and 10-15 % of the potassium. Consequently the fertilizer characteristics of both the solid and the liquid fraction (LIQ) differ from the untreated slurry, and storage, spreading techniques and timing of the spreading have to be adapted to ensure minimum losses and maximum utilisation of the nutrients in each fraction. We have investigated the leaching losses of N and P after applying SOL to winter wheat and the N utilization following application of SOL and LIQ to winter wheat and spring barley.

Materials and methods

The SOLs used were obtained from three different separation processes: (1) DEKANTER is a physical separation technique based on centrifugation. (2) SEPTEC is a screen separation of pig slurry following chemical precipitation of P. (3) GFE is a high-tech treatment involving liming, heating, anaerobic digestion and finally centrifugation like in the DEKANTER treatment. In this study, the manure treated by (1) and (2) was pig slurry, while mixed manure was treated by (3).

In September 2003, before the seeding of winter wheat, the SOLs from slurry separation were incorporated into the soil in lysimeters containing either loamy sand or a sandy loam soil. Other lysimeters were prepared with winter wheat for spring application of SOL, untreated pig slurry or mineral fertilizer. The SOL application rate was adjusted to 90 kg P ha^{-1}. Inorganic N application was adjusted to 180 kg N ha^{-1} all in treatments by

adding extra mineral fertilizer in spring. Residual effects of the SOL applications were monitored the following year. During the two years excess water was continuously collected at a depth of 1.5 meters, and the water was analyzed for nitrate and total P. The potential for P leaching from SOLs applied to sandy loam soil was also evaluated by applying controlled high intensity rain to intact soil cores of 20 cm depth in the laboratory one week after application of SOLs (facilitating preferential flow).

The utilization of the nitrogen in SOLs and LIQ from different separation plants using centrifugation was determined in field trials with winter wheat and spring barley. SOLs were applied to winter wheat in the autumn before ploughing and seeding. Other plots were prepared for application of SOLs, LIQs or mineral fertilizer in the spring. The LIQ was either applied with trailing hoses or by shallow injection. Similar trials were carried out in spring barley with the exception that autumn spreading of SOL was excluded and a treatment with dried, pelleted SOL was included. N utilization in SOLs and LIQs were calculated as the mineral fertilizer equivalent (MFE) by relating measured N uptake in grain to N uptake after application of increasing amounts of mineral N fertilizer.

Results and discussion

Application of SOLs in autumn, just before sowing winter wheat, doubled the nitrate leaching during the following leaching-season (Fig. 1). The extra N leaching was equivalent to 19-34% of N applied with SOLs. There was no significant difference in N leaching between the three SOLs when relating to the total amount of manure N applied, and no significant difference between the two soil types. In the second year, no residual treatment effects on leaching of nitrate could be detected (Fig. 1).

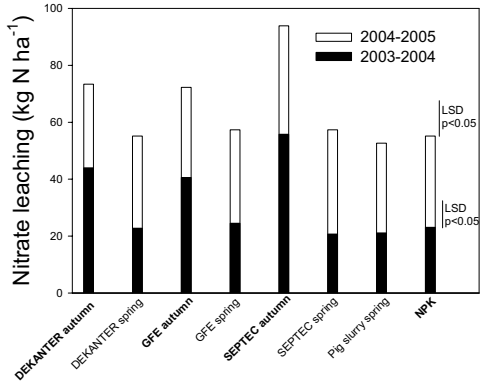

Figure 1. Leaching of nitrate during two seasons after application of SOLs to winter wheat on a sandy loam soil in autumn 2003 or spring 2004 (n=3).

In general, the leaching of phosphorus in the lysimeters was low (40-165 g P ha^{-1} y^{-1}) and not related to the manure treatments. However, the lab studies inducing high intensity rainfall showed that the potential for leaching of dissolved P doubled after SOL incorporation (Fig. 2). This can be a problem on clayey soils with a high potential for preferential flow to drains.

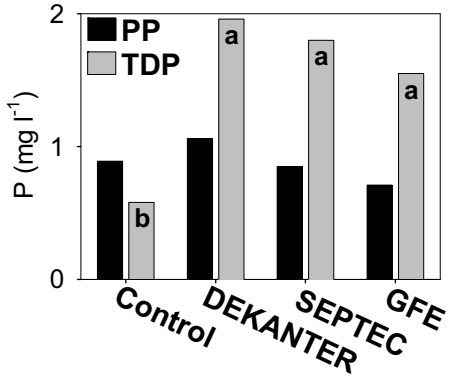

Figure 2. Mean particulate P (PP) and mean total dissolved P (TDP) concentrations in effluents after 37 mm intensive rain. Within each P form, different letters indicate that P concentrations differed significantly (P<0.05).

The MFE of total N in SOL was 18% after autumn application and 29% after spring application to winter wheat (Table 1). The MFE of SOL incorporated before sowing spring barley was 50%. For dried and pelleted SOLs the MFE was only 8%.

Table 1. Mineral fertilizer equivalent (MFE) of total N in slurry separation products measured in field trials in 2003, 2004 and 2005 (% of total N). Numbers of trials are indicated in brackets.

Product	Crop	Spreading method	Application time	MFE			
				2003	2004	2005	Ave-rage
SOL	Winter wheat	Broad spread, ploughed down	Autumn	20	14	12	18 (11)
		Broad spread	April 1	50	24	19	29 (11)
	Spring barley	Broad spread, ploughed down	Before sowing	52	50	49	50 (15)
SOL pellets	Spring barley	Broad spread, ploughed down	Before sowing	11	2	10	8 (15)
LIQ	Winter wheat	Trailing hoses	April 1	86	86	65	78 (11)
		Injected	April 1	100	97	71	89 (11)
	Spring barley	Trailing hoses	Before sowing	91	82	79	84 (15)
		Injected	Before sowing	96	84	76	85 (15)

About 30% of total N in SOL is ammonium, and in winter wheat most of this N is lost by leaching after autumn application, and by volatilisation after spring application. The average MFE of LIQ was 78% when applied with trail hoses and 89% when injected to winter wheat. In spring barley the utilization of LIQ was 85% irrespective of application technique.

Conclusion

- Application of SOL in the autumn before sowing winter wheat resulted in extra N leaching equivalent to 19-34% of total manure N whereas spring application had no detectable effect on N leaching in the following year.
- There is a risk of increased P leaching after application of SOL under conditions with significant water flow through macro-pores to drains, but no extra P leaching from SOL was observed in the lysimeter study.
- The highest utilization and the lowest losses of nutrients are attained with the SOL from slurry separation applied to spring-sown crops. After application of SOL to winter wheat, a high loss of nitrogen seems to be unavoidable due to nitrate leaching or ammonia volatilization. The utilisation of N in LIQ is generally high and this fraction can be used on almost any crop.

Uptake of cation micronutrients by wheat from two different animal manures applied to a sandy loam soil

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High concentrations of animals living in intensive farms, industry development, and life modernization can result in the production of wastes in excess of safe levels for disposal on adjacent land. On the other hand, such materials can be recycled to enhance the future crop production by improving soil quality. They may also serve as a source of plant macronutrients, such as nitrogen (N), phosphorus and potassium, and micronutrients, namely copper (Cu), zinc (Zn), iron (Fe) and manganese (Mn). In fact, organic sources can often provide sufficient or even excessive quantities of cation micronutrients (Cu, Zn, Fe, and Mn), and toxicity might be a bigger problem in many cases than deficiency.

To evaluate the effect of applying pig (Pi) and poultry manure (Po) to a sandy loam soil (Cambic arenosol) on the uptake of cation micronutrients by wheat, a 4 months experiment using Kick-Brauchmann pots was carried out under semi-controlled environment conditions. Pots were periodically weighed and watered to keep soil at 60 % WHC. Previously dried and sieved pig and poultry manure were mixed separately with the soil (Table 1) in two doses corresponding to 80 and 160 kg total N ha^{-1}. Half of the pots received only manure as fertilizer, and the other half received a basal mineral N fertilization as well (120 kg N ha^{-1}). A control treatment (only soil - Z) was tested, as well as a treatment with only the same mineral N fertilization, but no manure (T). Both soil and manures were fully chemically characterized (Table 1).

Table 1. Chemical characteristics of the soil and wastes.

	pH	OM	TOC	Nkj	C/N	Fe	Cu	Zn	Mn
	(H_2O)	(g kg^{-1})	(g kg^{-1})	(g kg^{-1})		(mg kg^{-1})			
Soil	6.1	5.9	3.4	0.34	10	10	45.0	9.9	14.7
Po	8.4	765.9	421.0	35.5	12	2300	43.8	430.4	306.7
Pi	6.6	714.0	414.0	9.1	46	5100	490.7	823.0	633.3

(OM - organic matter; TOC - total organic C; Nkj- kjeldahl N; C/N – C to N ratio; Po – poultry manure; Pi – pig manure)

Figure 1 shows that the application of the manures and mineral N fertilizer promoted a higher wheat production, compared to treatment T. Also, wheat production in pots that received only manure as nutrient source, was greater than the yield obtained in the Z treatment. Poultry manure led to higher biomass production than pig manure, except when dose 80 was applied simultaneously with mineral N fertilization. The higher initial N content in this poultry manure, with higher readily available N contents (Cordovil *et al*., 2005), allowed a greater N availability to wheat plants.

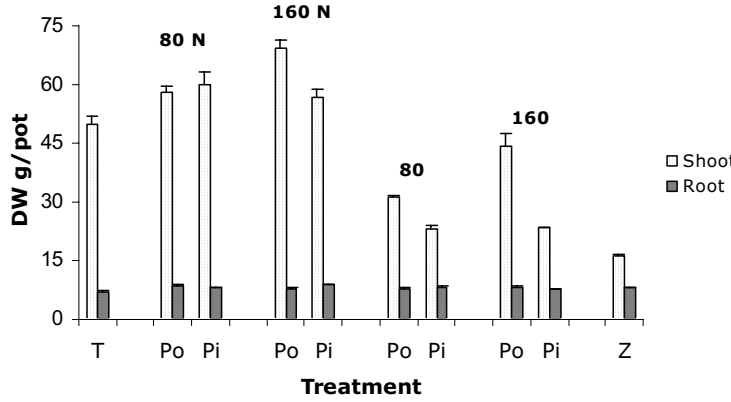

Figure 1. Wheat dry weight (g/pot). (Po – poultry manure, Pi – pig manure; 80 N, 160 N – manure plus mineral N; 80, 160 – only manure as source of N).

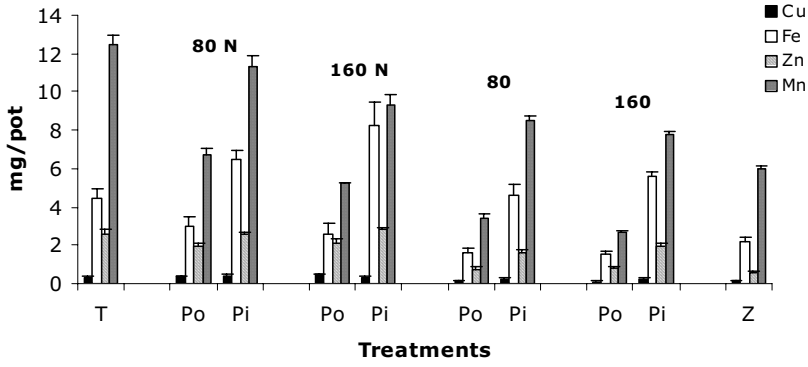

Figure 2. Wheat uptake of copper, iron, zinc and manganese (mg/pot).

The much higher micronutrients concentration of pig manure, compared to the poultry manure, resulted in a greater uptake of all the micronutrients studied (Figure 2). Treatment T, with application of mineral N, promoted the highest manganese uptake by wheat plants. This may have resulted from the fact that soils amended with NPK tend to have lower pH, resulting in a greater amount of Mn available to plants uptake (Warman & Cooper, 2000).

The plant uptake of all the cation micronutrients in treatments receiving manure was stimulated by the presence of mineral N fertilizer at both manure levels tested. A decrease in pH due to mineral fertilization can have resulted in a greater availability of micronutrients for wheat (Scherer, 1997). Moreover, the higher dose of manures applied depressed the micronutrient absorption by wheat, probably due to the complexation of these elements with organic matter. Warman and Cooper (2000) found the same effect in mixed forage micronutrient uptake when cultivated in a sandy soil amended with chicken manure, and Selvi *et al*. (2002) found that organic matter in FYM reduced micronutrients availability due to the binding to humic and fulvic acids.

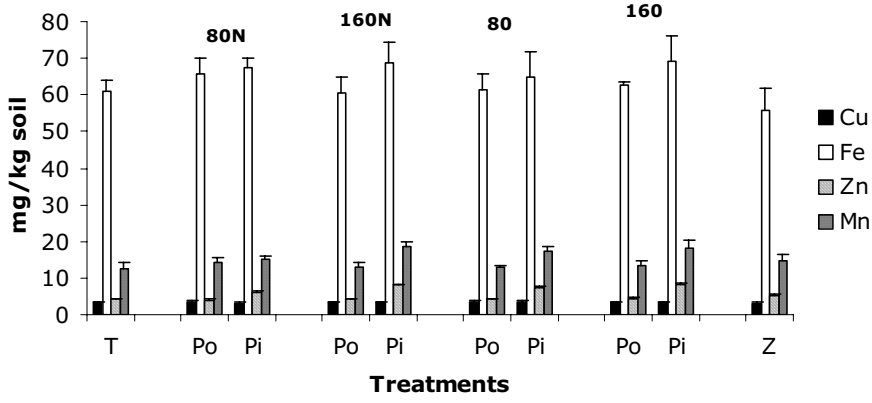

Figure 3. Micronutrients content in soil after the experiment.

Contents of micronutrients in soil (Figure 3) were not significantly different when comparing both manures tested, and were consistent with the results of plant uptake shown in Figure 2. Iron was the micronutrient that presented a higher accumulation in soil. Nevertheless, Fe accumulation did not depress neither plant growth nor increased Fe absorption. The presence of extra organic matter due to manure

application, probably enhanced soil fertility, attenuating micronutrients growth depressing effects (Burgos *et al*., 2002).

None of the manures tested promoted the absorption by the plants of cation micronutrients to toxic levels. Application of manures at doses up to 160 kg N ha^{-1} did not induce toxic levels of cation micronutrients in wheat plants.

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NH_3 and GHG emissions from a straw flow system for fattening pigs: housing, and manure storage

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Introduction

Housing systems must be found that consider animal welfare while emitting little NH_3 or greenhouse gases (GHG). Often, animal welfare and environmental protection are seen as being mutually exclusive. Consumers demand pork to be produced in straw-based systems that they consider animal friendly. On the environmental side there is a tendency to favour slurry based systems because these are assigned lower NH_3 and GHG emissions. A solution to this conflict must be found.

Straw-based systems are often equated with deep litter, where the lying and excretion areas are not separated. In deep litter systems, most of the pig's requirements are fulfilled, however some disadvantages are encountered. The straw consumption is high, the pigs are likely to be dirtier, and some authors have found deep litter systems to emit high levels of NH_3 and GHG. The straw flow system, however, distinguishes a lying and an excretion area. Only a small part of the pen is soiled with excreta. Additionally, excreta may be frequently removed from the pig house by a scraper. The small emitting surface and the frequent manure removal may contribute to a reduction in emissions. It was to be investigated, if the straw flow system emitted less NH_3 and greenhouse gases (GHG) than a conventional fully slatted floor system.

Materials and methods

Emissions of NH_3, N_2O, CH_4, and VOC were continuously measured at a commercial farm in Upper Austria from June 2003 to April 2004. The animal house consisted of three fully separated compartments. Each compartment was forced ventilated by a central exhaust fan. Excreta were collected in a dung channel in the rear of the pen (= dung channel system). In two compartments the dung channel was additionally

equipped with a scraper that was operated twice a day and moved the slurry to an outside store (= scraper system). Each compartment was separated into 16 pens that held 10 – 12 pigs each. Gas concentrations were measured with high resolution FTIR spectrometry. VOC was analysed by a flame ionisation detector. The ventilation rate was continuously recorded in the central exhaust fan.

From May 2004 to June 2005 emissions were followed from storage of pig slurry received from a straw flow system. The measurements compared emissions from covered and uncovered slurry stores. The influence of the commercial additive "effective micro organisms (EM)" on the emission level in the animal house, and during slurry storage was investigated.

Results

Animal house

Emissions of CH_4, N_2O, NH_3, GHG, and VOC from the straw flow system were always lower than default values for forced ventilated fully slatted floor systems. From the straw flow system CH_4 emissions of 1.2 (dung channel system), respectively 0.5 (scraper system) kg CH_4 per pig place and year were lost. The default value for fully slatted floor systems is 4.0 kg CH_4 per pig place and year (UBA 2001). N_2O emissions from the straw flow system amounted to 40 (dung channel system), respectively 25 (scraper system) g N_2O per pig place and year. Fully slatted floor systems are estimated to emit 100 g N_2O per pig place and year (UBA 2001). NH_3 emissions from the straw flow system were 2.1 (dung channel system), respectively 1.9 (scraper system) kg NH_3 per pig place and year. The default value for fully slatted floor systems is higher: 3.0 kg NH_3 per pig place and year (Döhler et al. 2002, UBA 2001). VOC emissions were 3.5 (dung channel system) and 1.6 (scraper system) kg VOC per pig place and year. The spraying of the commercial additive “EM” resulted in a reduction of CH_4, NH_3, N_2O, GHG, and VOC from the dung channel system.

Manure storage

A solid cover placed on the pig manure store was an efficient means to reduce NH_3 and GHG emissions during pig slurry storage. Pig slurry stored under cool climatic conditions emitted less NH_3, and GHG than when stored under warm climatic conditions.

Table 1. Emissions during slurry and FYM storage

Treatment	Cumulated emissions of …			
	CH_4 [kg m^{-3}]	N_2O [g m^{-3}]	NH_3 [g m^{-3}]	VOC [kg m^{-3}]
Surry, summer				
covered	0.17	30	66	0.54
uncovered	0.25	23	118	0.75
EM_addition	0.44	24	110	1.33
Slurry, winter				
covered	0.10	18	19	0.79
uncovered	0.16	36	22	0.88

Conclusions

The straw flow system emitted less NH_3 and GHG than a conventional fully slatted floor system. When the dung channel system was additionally equipped with a scraper and the pig manure was daily removed to the outside storage, emissions from the straw flow system were further reduced. Stores for pig slurry should be equipped with a solid cover. It is recommended to set up the national emission inventory by applying separate emission factors for slurry storage in the cooler and in the warmer half of the year. The straw flow system combines recommendations of animal welfare and environmental protection.

Acknowledgements

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Spanish Ministry of Agriculture, Fisheries and Food project for the IPPC Directive implementation in Spain. Results of 2004-2005 and future work

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The aim of this paper is to describe the current situation in the Spanish pig and poultry sectors regarding the control of emissions regulated in the IPPC Directive, as well as the approach and actions of the Spanish Ministry of Agriculture, Fisheries and Food for its implementation.

During 2004-2005 period, this Ministry has developed an Spanish Guide Document on Best Available Techniques (BAT) for intensive rearing of poultry and pigs. The Spanish Guide Document has been updated with the latest information available, both technical and economical. The sources for this update were the new literature, as well as the results obtained from different experiments performed for BAT assessment under Spanish conditions. The results have been transferred to technicians and farmers. The primary goal for the Spanish Ministry of Agriculture was, therefore, to evaluate the cost/efficiency of these techniques at commercial farm level.

The candidate BATs were selected from the Reference Document on Best Available Techniques for Intensive Rearing of Poultry and Pigs (BREF, 2003) based on their potential efficiency, applicability, cost-effectiveness and eligibility under Spanish conditions. The BATs selected were assessed for the pig and poultry sectors under commercial conditions in the different production phases. The main results obtained were the following:

Poultry:

- Laying hens. Frequent manure removal and manure drying were able to abate ammonia and methane emissions by 50 and 31%, respectively ($P<0.05$).
- Broilers. Non-leaking drinking systems were able to decrease methane emissions by up to 57 % ($P<0.05$).

Pigs:

- Gestating sows. Reduced manure pit and hence, emission surface was able to reduce ($P<0.05$) ammonia, methane and nitrous oxide emissions by 49, 28 and 68%, respectively. Frequent manure removal was able to reduce ($P<0.05$) methane and nitrous oxide emissions by 19 and 83%, respectively.
- Lactating sows. Manure pan underneath reduced ammonia, methane and nitrous oxide emissions by 32, 65 and 43%, respectively.
- Nursery. Frequent manure removal, manure channel with sloped side walls and low protein diets were highly effective in the reduction of emissions ($P<0.05$): 24, 51 and 63% for ammonia; 10, 65 and 63% for methane and 41, 27 and 39% for nitrous oxide, respectively. Productive performance was not affected ($P>0.10$), except that average daily gain was compromised in low protein diets (9%).
- Growers-finishers. Frequent manure removal, manure channel with sloped side walls, partially slatted floor and low protein diets were also highly effective in decreasing ($P<0.05$) ammonia (10, 36, 42 and 60%), and methane (65, 52, 34 and 33%) emissions.

Storage:

- Natural crust, chopped straw and plastic cover decreased ammonia emissions by up to 28, 93 and 99 % ($P<0.05$), respectively.

Spreading:

- Trailing shoe reduced ammonia emission in grassland by 67%, and band spreader by 51%. On arable soil, reductions of ammonia represented 64% when incorporated within 6 hours.

Cost calculations were also carried out according to the methodology suggested in the BREF (2003).

Table 1. Cost calculation. Results.

	Techniques	Extra-cost (€/place and year)
Poultry		
Laying hens	Frequent manure removal	0.013
	Manure drying	0.182
Broiler	Non-leaking drinking systems	0.250
Pigs		
Gestating sows	Reduced manure pit	5.69 – 6.83
	Frequent manure removal	0
Lactating sows	Manure pan underneath	17.52 – 37.18
Nursery	Manure channel with sloped	0.23 – 2.67
	Frequent manure removal	0
Growers-finishers	Manure channel with sloped	0.73 – 7.74
	Partially slatted floor	0 – 4.33
	Frequent manure removal	0
		Extra-cost (€/m^3)
Spreading	Trailing shoe	0.92 – 1.41
	Band spreader	0.79 – 1.21
	Incorporation	0.23 – 0.61

On going activities are focused on measuring emissions in other Spanish regions with different climate conditions, on the spreading of the results to broader climatic and soil type conditions, and on the developement of a tool-box to calculate emissions at farm level with emphasis on detecting the critical points to control them.

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Nitrous oxide and di-nitrogen losses following application of livestock manures to agricultural land

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Introduction

Around 90 million tonnes of livestock manures supplying 450,000 tonnes of nitrogen (N) are applied annually to agricultural land in the UK (Williams et al. 2001). However, land application practices can result in unwanted N losses, both in terms of environmental pollution and agronomically. In order to minimise such losses, it is important to balance manure N supply with crop demand. To enable farmers to estimate the manure contribution to crop available N supply and to estimate environmental losses (e.g. via nitrate leaching and ammonia volatilisation) the MANNER decision support system (DSS) was developed (Chambers et al. 1999). However, the existing version of the MANNER-DSS does not include estimates of N losses by nitrous oxide (N_2O) and di-nitrogen (N_2) emissions.

Nitrous oxide is a greenhouse gas with a global warming potential of 310 times that of carbon dioxide (IPCC, 1996). The current UK emissions inventory (2004) estimates that 66% of N_2O is produced from agriculture, with the majority emitted from agricultural soils, which includes emissions after livestock manure application (Baggott et al. 2006). Utilising new research data, information on N losses as N_2O and N_2 are being incorporated into an enhanced and updated version of MANNER (MANNER-NPK) through the derivation of N_2O and N_2 emission factors (EF).

Methodology

The Intergovernmental Panel on Climate Change (IPCC) default N_2O EF (i.e. EF_1) is 1.25% of total N applied, although when calculating emissions following manure spreading, this has to be corrected for the loss of N by ammonia (NH_3) volatilisation and NO_x emissions. The first step in the MANNER-DSS estimates N loss by NH_3 volatilisation. Hence, N_2O emissions following livestock manure application were estimated based on the amount of readily available N (i.e. ammonium-N, nitrate-N and uric acid-N) remaining after NH_3 loss, assuming that N_2O emissions from

organic N mineralisation would be relatively small. Also EFs were expressed as the % of total N applied, the % of total N applied remaining after NH_3 loss and the % of readily available N applied (Table 1).

Nitrous oxide EFs were derived from a database containing the results of field studies carried out by ADAS and IGER between 1994 and 2003. Experimental data had to satisfy certain criteria (e.g. measurement period >21 days) before they were included in the data set, which generated 92 EFs from a range of sites in England under grassland and arable cropping. However, it should be noted that *c*. 75% of the EFs were generated from a measurement period of <90 days. The effect of manure type, land use, slurry application method and rapid incorporation on N_2O EFs was also evaluated. The N_2:N_2O loss ratio was derived from a database containing results from 21 experimental studies where both N_2O and N_2 had been measured following the application of manure, using the acetylene inhibition technique (Ryden et al. 1987).

Results and discussion

Nitrous oxide EFs were dependant upon manure type where the EFs were expressed as the % of total N applied, the % of readily available N applied and the % of total N applied remaining after NH_3 loss ($P<0.05$) (Table 1), with the greatest losses from poultry manure. However, when the results were expressed as the % of readily available N applied remaining after NH_3 loss, there was no difference in the EF between manure types ($P>0.05$).

Table 1. Livestock manure N_2O emission factors. Values in parentheses = one standard error of the mean

Manure type (number of measurements)	% total N applied	% total N applied remaining after NH_3 loss	% readily available N applied	% readily available N applied remaining after NH_3 loss
Slurry (51)	0.57 (0.13)	0.67 (0.15)	1.06 (0.24)	1.76 (0.41)
FYM (27)	0.28 (0.08)	0.30 (0.08)	1.29 (0.27)	1.97 (0.46)
Poultry manure (14)	0.75 (0.14)	0.79 (0.14)	2.05 (0.41)	2.70 (0.54)
Mean (92)	**0.51 (0.08)**	**0.58 (0.09)**	**1.27 (0.16)**	**1.96 (0.27)**

The data showed that there was no consistent effect ($P>0.05$) of slurry application technique (i.e. band spread/shallow injection vs surface broadcast) on N_2O EFs, although there were only seven directly comparable studies with

considerable variability in the EFs following band spreading/shallow injection. The rapid incorporation of manure resulted in a greater ($P<0.05$) N_2O EF (% total N applied) than following surface broadcast manure applications. However, when the results were expressed as the % of readily available N applied, the % of total N applied remaining after NH_3 loss and the % of readily available N applied remaining after NH_3 loss, there was no effect of rapid incorporation ($P>0.05$) on the N_2O EF. Additionally, there was no consistent effect ($P>0.05$) of land use on the N_2O EF.

Conclusions

Within MANNER-NPK, the mean N_2O EF across the whole data set was 1.96% (range –0.10 to 14.00%) of the readily available manure N remaining after volatilisation, and the mean N_2:N_2O ratio was 2.9 (range 1.0 to 9.1).

The mean N_2O EF expressed as a % of total N applied (0.51%) was considerably lower than the mean IPCC default value, but within the IPCC estimated range of 0.25-2.25%. Although, these values may be an underestimate of the annual emission because of the short term nature (< 3 months) of many of the measurements.

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Acknowledgements

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Inventory of gaseous emissions (CH_4, N_2O, NH_3) from livestock manure management in France using a mass flow approach

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Agriculture and more particularly manure management is a major source of gaseous emissions. Although some national emission inventories were previously compiled using IPCC Guidelines and the EMEP/CORINAIR Guidebook, more accurate inventories are required, particularly to identify and evaluate potential mitigation strategies. For this, the mass flow approach was identified as a useful tool (Webb and Misselbrook, 2004; Dämmgen and Webb, 2006). In this context, a methodology based on the mass flow concept was developed to quantify NH_3, CH_4 and N_2O emissions from livestock manure (from cattle, swine and poultry breeding) in France.

Considering the mass flow concept (Figure 1), NH_3 and N_2O emissions originate from the pool of nitrogen excreted by the animals. These emissions may occur successively from the nitrogen entering each stage of manure management. A similar balance was made for CH_4, considering the potential CH_4 initially contained in the manure excreted (B_0). To ensure consistency in the mass flow approach, N_2 and CO_2 emissions were also included where necessary.

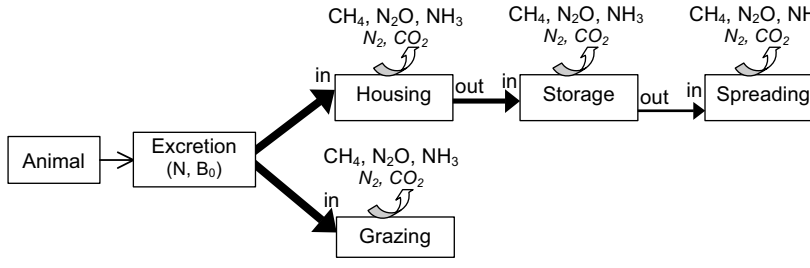

Figure 1. Description of the mass-flow approach used for calculation of gaseous emissions

Homogenisation and statistical processing of data from 167 publications allowed us to determine emission factors for each animal type and each stage of management, adapted to the national context (Table 1).

Table 1. Emission factors (NH_3 and N_2O in % of $N_{(in)}$, CH_4 in % of C-$CH_{4(in)}$)

			CH_4			N_2O			NH_3		
			slurry	FYM	p.drop	slurry	FYM	p.drop	slurry	FYM	p.drop
GRAZING	Cattle		0.6			2.6			6.5		
	Swine		0.04			0.9			13.8		
	Poultry		0.04			0.9			10.7		
HOUSING	Cattle	Dairy	11.7	9.4	-	0.17		-	17	10.8	-
		Others	5.7	9.45							
	Swine	Piglet	31.8	17.4	-	0.09	9.47	-	8.6	14	-
		Fattening							14.4	23.9	
		Sows							17.4	28.3	
	Poultry	Hens	31.8	4.4	4.4	0.09	1.2	1.2	29.2	30.4	12.3
		Broilers		0	0		0	0			
STORAGE			16.7	10.4	10.4	0	0.3	0.15	3.5	9.5	8.5
SPREADING			0.04			0.9			19.6	10.7	10.7

FYM: farmyard manure; p.drop: poultry droppings

This literature review provided many data concerning ammonia emissions while few results were available concerning greenhouse gas (GHG) emissions, particularly for cattle and poultry. Therefore, further measurements are needed to improve the quality of the data.

An Access® database containing these emission factors, the census data and the manure compositions was developed to allow the calculation of gaseous emissions by the mass flow approach previously described. Figure 2 shows the national gas emissions inventory from manure management for the year 2003 obtained from this database.

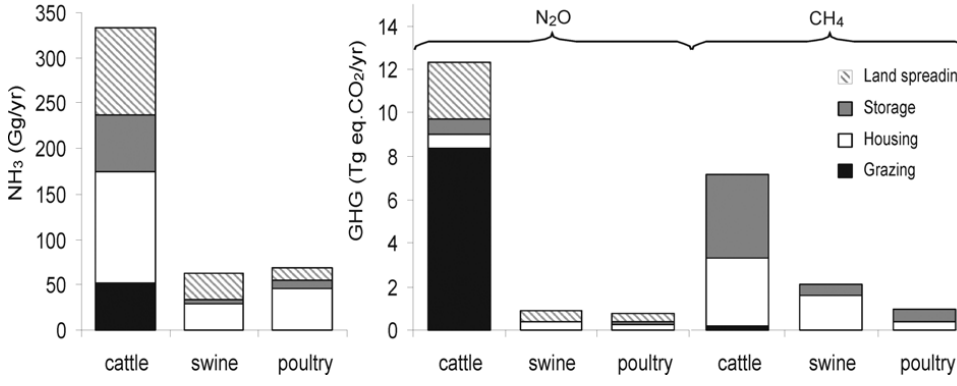

Figure 2. Inventory of the national NH_3 and GHG emissions in 2003.

Total ammonia emissions were estimated at 463.8 Gg, arising mainly from cattle (72%). Greenhouse gas emissions were estimated at 24.2 Tg CO_2-eq (N_2O: 58%, CH_4: 42%). Cattle grazing is the major source of N_2O (57%). N_2O emissions occur mainly when manure is applied to the soil, and in France, cattle spend half of their time outside. Although some differences were observed, overall the results obtained during this study were close to those obtained with the IPCC and EMEP/CORINAIR methodologies (variation : from –10% to +24%).

Some mitigation options were evaluated using the database. The mass flow approach allows the consequences of reducing emissions at one stage on emissions at later stages of manure management to be taken into account. As an example, slurry flushing in swine buildings reduces methane emissions from the house (95.6%), but results in increases during storage (122%) and landspreading (30%). However, across the whole management system, a reduction is observed (40%). Although covering slurry storage tanks is known to be an effective measure for reducing ammonia emissions, the application of this technique in the national context gives only a small reduction in total emissions (2%).

This database is a new tool to assess national gaseous emissions. It could also be used to evaluate the wider impact of mitigation strategies, which could be useful to develop effective abatement policies. Other mitigation options such as anaerobic digestion and aerobic treatment could be evaluated with the database. However, few data exist about the different techniques to assess accurate emission factors for each of them. Consequently, further full scale measurements are required.

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A farm-scale internet-based tool for assessing the effect of intensification on losses of nitrogen to the environment

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Introduction

When farmers wish to extend or intensify their operations, the authorities must assess the likely effect on nitrogen (N) losses to the environment. This is particularly difficult for farms with ruminant livestock, because losses from animal housing, manure storage facilities and fields are interlinked. The FARM-N internet tool was developed to provide a tool to assist in this process.

The model

The underlying philosophy of the tool is that the farm N surplus (N input – N output) can usually be more accurately estimated than the internal N flows, so the latter should be constrained within the former. The farm N inputs are imports of the purchased items; mineral fertiliser, animal feed, bedding, animal manure, livestock and seed, and the non-purchased items; N fixation and atmospheric N deposition. The farm N outputs are the crop and animal products sold, including any livestock manure. An N flow approach is then used in the calculation of internal N flows and emissions. The model can currently simulate pig, cattle and arable farms.

The following inputs are required for the current and proposed farm structures and management; the number and type of livestock to be kept, the animal housing and manure storage facilities to be used, the fields available to the farm and field management. The field data required are the previous land use (for cattle, pig, or arable farming), the area, soil type, whether irrigated or not and the crop mixture within one or more crop rotations. Using the crop mixture given, a linear programming (LP) model generates a plausible sequence of cropping within each crop rotation. The crop rotation model maximises the value of the carry-over effect of previous cropping on the N availability to the current crop, within certain agronomic constraints. Standard values for the carry-over effect are used here. The crop rotation is then presented to the user, who must indicate which crops can receive animal manure, the proportion of the

production of each crop to be sold, whether any straw produced is to be sold and whether a crop is to be grazed. The dry matter, energy and protein in crop production are estimated using standard values.

The data required for livestock are the number, production parameters and the housing type used. The production parameters include: for all cattle; % energy from grazed fodder, for dairy cattle; annual milk production, for other cattle; daily weight gain, for all pigs; annual energy and protein requirement, for sows; annual production of piglets and piglet weaning weight, for piglets and finishing pigs; initial and end weights. For cattle, simple regression models are used to estimate the energy and protein requirement.

The model calculates the import of animal feed that is necessary to satisfy the livestock requirements. If crop production exceeds livestock requirements, the surplus is sold. The model estimates the N excreted in livestock faeces and urine, based on the livestock diet and the N partitioned to animal products. The type of animal housing determines the type of manure produced and the addition of N in bedding. An LP model is used to distribute the manure to those crops that can receive it, maximising the effectiveness of the N in the manure. In doing so, the LP model uses standard values for the effectiveness of the N for each manure type x manure application method x crop type combination. Standard values for the crop requirement for plant-available N and the plant-availability of the N in different manure types are used to calculate the need for supplementary mineral N fertiliser. The emission of N as ammonia (NH_3) from animal housing, as NH_3, nitrous oxide (N_2O) and dinitrogen (N_2) from manure storage, and as ammonia following field application, are then estimated using standard emission factors for each combination of manure type x application method (Hutchings et al, 2000).

The model results for the production and utilisation of manure and crops are presented to the user. The user must check whether the production of grazed fodder crops, if any, matches the demand for grazed feed, adjusting their previous assumptions if necessary.

The N input to the fields (mineral N + manure N – field NH_3 emissions) and the amount exported from the fields (from crops harvested) are now calculated. The difference must then be partitioned between losses of N_2O, N_2, nitrate (NO_3) and changes in the soil N. A simple sub-model of

denitrification (Vinther and Hansen, 2004) is used to estimate soil N_2O and N_2 emissions. The change in soil N is calculated using CTOOL, which is a simple model of soil C dynamics (Petersen et al; 2002). The remaining N is assumed to be lost via NO_3 leaching.

Discussion and conclusions

The tool has a number of advantages. Farmers can be reasonably expected to provide the quantity and type of data demanded. The calculation time is short, so many scenarios for future management can be investigated. The use of the internet means that there is the possibility of linking to existing databases held by the authorities, which would reduce further the demand for input data, and that upgrading the tool is simplified. The integration at the farm scale means that the environmental authorities are made aware of situations where abatement measures proposed for one source (e.g. NH_3 from animal housing) might increase losses from another (e.g. NO_3 leaching from fields). Farmers might not consider this an advantage.

The tool has a number of disadvantages. Assuming that NO_3 leaching equates to the farm N surplus minus all other losses and changes in soil N means that errors are accumulated here. This is currently being addressed by incorporating an empirical NO_3 leaching model. However, this will then usually result in an unexplainable difference between the farm N surplus and the sum of the losses and the changes in soil N. Given the philosophy underlying the tool, this difference will be partitioned in an as yet undecided way between the N losses and changes in soil N. The standard values for crop yield and quality are only valid for situations where the underlying assumptions concerning the input of plant-available N are satisfied. Presently, it is assumed that the farmers apply N to the maximum permitted by current legislation. This means that the tool cannot examine scenarios where current or future fertilization rates are at lower or higher levels. Finally, the tool is dependent on the availability of a range of standard data. This is the case in Denmark, but may not be elsewhere.

The tool is currently being modified for use in environmental impact assessment in Denmark. A demonstration version is available at www.farm-n.dk.

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Ammonia concentrations around 5 farms in the Central Plateau of Spain

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A large proportion of the ammonia (NH_3) emitted locally is, in contrast to other pollutants, deposited in the immediate neighbourhood of the source rather than transported over long distances. Quantitative information about the spatial location of emission sources, as well as about the fate of the nitrogen after emission, is crucial for NH_3 target-oriented abatement. This spatial perspective is especially important, as previous studies have shown large variations in NH_3 emissions and deposition in Europe. Few studies are undertaken which show the distribution of NH_3 emissions, deposition and impacts at the field level.

The aim of this trial, financed and coordinated by the Spanish Ministry of Agriculture, Fisheries and Food through a broader project carried out by Tragsega. S.A., was to describe NH_3 concentrations in the surroundings of several sources (pig and poultry farms) at field level. Five farms were selected as representative for the area: laying hens, broiler meat, grower finisher-pigs, sows with piglets, and one-site pig farm. All trials were carried out simultaneously (in autumn) within a radius of less than 100 km, where similar climatic conditions were prevailing and the topography was uniform, rather flat terrain. The time periods and the location of the experimental farms are shown in Table 1.

Table 1. Experimental farms, located in Segovia (Spain)

Farm	**Type**	**Animal numbers**	**Date**
Valtiendas (A)	Pig farm	5400	21-26/09/2005
Cantalejo (B)	Grower-finisher pigs	586 (sows)	29/09-04/10/2005
Aguilafuente (C)	Sows with piglets	480 (sows)	23-28/09/2005
Navalmanzano (D)	Broiler meat	26000	22-27/09/2005
San Cristóbal (E)	Laying hens	375000	28/09-03/10/2005

With the aim of optimizing the experimental setup in the different experiments and sites, a cup anemometer was installed before of beginning of measurements at each of the study sites (wind speed and wind direction with a 10 minutes average period were measured).

Ammonia concentrations were measured with passive samplers, box sampler type - Ferm type (Sanz et al., 2005) at four different directions (N, S, E, W) at different distances from the farm (up to 500 m). Twenty one sampling points per farm at a height of 1.8 meters (n=3 replicates) were established at 100, 200, 300, 400, and 500 meters.

The software program Industrial Source Complex–Short Term (ISCST3), based on a double reflected Gaussian dispersion model, was used to simulate the emissions, (EPA–454/B-95-003a, 1995). The model needs two input files to calculate a preliminary estimation of the concentration distribution at the study surface: 1) emission rates, location of sources within the study area, and the receptor net in one file; and 2) meteorological variables such as temperature, wind speed and wind direction, stability class, and mixing height in a second file. Wind direction and speed were measured with the cup anemometer, and temperature was collected from Agro Climatic Towers (Gomezserracín-Segovia and Vadocondes -Burgos).

Emission rates were calculated based on the Gaussian dispersion equation (Price et al., 2004). These emission rates were introduced in ISCST3 as input parameters, and the output concentration files of the model were compared with the experimental concentration measurements at surface level.

In addition to the surface concentration distribution, a dispersive modelling exercise for certain cases was made. Figure 1 shows the concentration surface predicted by the model ISCST3, along with the concentrations measurements with the passive samplers, on two farms: Cantalejo and Aguilafuente. On both farms a good correlation ($r > 0.80$) was found between the concentration measurements and the ammonia concentrations predicted by the model. It is important to mention that high uncertainty is associated with the model estimates, and there is a strong dependency on the meteorological parameters and the emission rates. The model was used to test the consistency of the ammonia concentrations measured by the passive samplers.

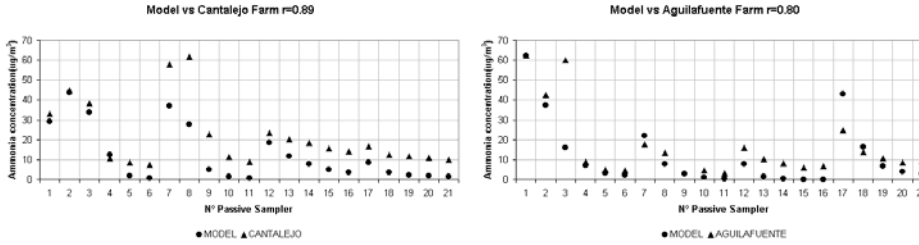

Figure 1. Ammonia concentrations predicted by the model, and ammonia concentrations measured on the farms at Cantalejo and Aguilafuente.

Maximum concentrations of 80 to 100 μg/m^3 were found at the source depending on the type of farm. Figure 2 shows the ammonia concentration surface maps for the farms of grower-finisher pigs and sows with piglets. These maps have been made with the experimental concentration measurements of each farm, interpolated with the software program SURFER (Surfer version 7.0,1999. Golden Software,Inc).

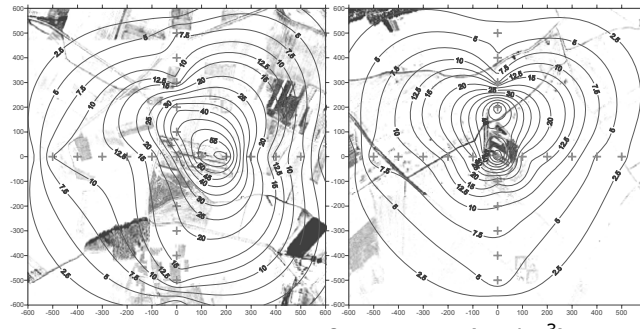

Figure 2. Ammonia concentration surface maps (μg/m^3). Grower-finisher pigs and sows with piglets farms.

Concentration fields decreased in all cases (5 farms) to a levels of 2 to 5 μg/m^3 within a distance of less than 1 km (approximately 600 m).

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Simulation of odour dispersion by natural windbreaks

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Introduction

Odour emissions from livestock operations create major environmental and social issues. Atmospheric dilution is perhaps the most simple and effective way of reducing such odour nuisance, but this solution may imply leaving considerable setback distances between the livestock facilities and the neighbouring structures. Natural windbreaks have been proposed to help disperse these odours and reduce setback distance while also hiding the livestock buildings. Although it is now a common practice in North America to plant trees around livestock facilities, best management practices have not been established.

In 2003, Lin et al. (2006) measured the size of odour plumes developing around 4 types of natural windbreak when a single isolated odour source is emitted. In general, a natural windbreak with an optical porosity of 0.35 was found to reduce the length of the odour plume by 20%. A porosity exceeding 0.35 reduced the odour dispersion effect of the natural windbreak, as opposed to reducing wind speed which requires an optical porosity above 0.60. The analysis of the measured field data was complicated by the absence of control on the prevailing climatic conditions and strength of the odour source. Modelling the dispersion capability of natural windbreak is therefore a more appropriate tool to compare the effect of different parameters and come up with best management practices, such as the tree type, height and density, the distance between the windbreak and the odour source, the orientation of the windbreak with respect to prevailing winds and the climatic conditions.

Therefore the objective of this project was to build a simulation model based on principles of mass, momentum, energy and gas species conservation, and evaluate the effect of type, porosity and height of trees forming the windbreak, and distance between the source and the

windbreak. The model was calibrated using data measured in the field and in an olfactory laboratory, using 12 trained panellists.

Method

Two of the Fluent software models were tested to simulate odour dispersion about windbreaks. The initially selected k-ε model was found to require too many cells to maintain the ratio of turbulent viscosity to molecular viscosity under the physical limit of 10^5. The SST model was found to better simulate the large turbulence created by the windbreak and was thus retained (Fluent inc., 2005).

Using field data measured by Naegeli in 1953 (Eimern, 1964), the SST model was calibrated to properly simulate wind velocity recovery rate defined as the ratio of wind speed at a height of 0.5H (H is the windbreak height) and a downwind distance of 30H, to that undisturbed at the same height, upwind from the windbreak.

The SST model was calibrated for odour dispersion using the data collected during five field tests which measured the odour plume created downwind from a natural windbreak consisting of a uniform single row of deciduous trees, measuring 7m in width, 9.2m in height and 1050m in length. The optical density of the windbreak, of 0.35, translated into an aerodynamic porosity of 0.66 (Lin et al., 2006). The inertial resistance coefficient C_{ir} of the windbreak was determined at a height of 0, 9.2 and 6.9m as 0.4, 0.29, 0.18m^{-1}, respectively. Once calibrated using a multiplication factor of 3.7 for the odour strength, the windbreak model was used to observe the effect of tree porosity, type and height, as well as distance to the odour source.

Results

The odour dispersion model indicated that the dispersed odour plume is shorter when:

1) the windbreak optical porosity is as low as 0.2 as compared to 0.6;
2) the odour source is located 15m upwind from the windbreak, as compared to 30 and 60m;
3) the windbreak is taller, such as 15m as compared to 5m;
4) the natural windbreak is made up of deciduous as opposed to coniferous trees, because coniferous trees are more opened at the crown whereas deciduous tress are more open close to the ground.

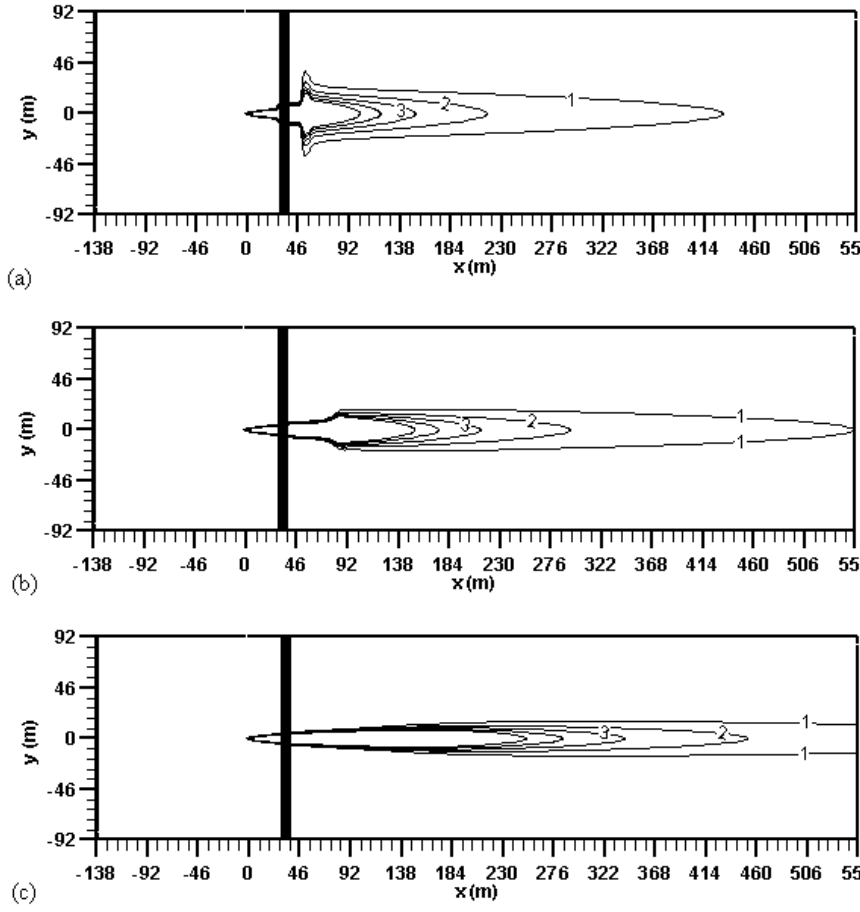

Figure 1. Effect of porosity (a) 0.2, b) 0.4 and c) 0.6) on windbreak odour plume dispersion.

As observed by Lin et al. (2006), conditions required to disperse an odour source, are the opposite of those required to reduce wind speed. This should be taken into consideration when locating and planting a natural windbreak for odour dispersion.

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Comparison of models and measurements for whole-farm ammonia emissions

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Introduction

Models of ammonia (NH_3) emissions at the whole-farm scale are needed for e.g.: assessment of environmental impacts at the local scale; assessment of the impact of potential mitigation strategies; as part of whole-farm nutrient flow models; the compilation of regional and national emission inventories. The objectives of the present study were to measure NH_3 emissions on two commercial farms to assess the effectiveness of the mitigation strategies on commercial farms (as opposed to controlled experimental studies), and also to provide data for, and make comparisons of, a number of whole farm NH_3 emission models used to support policy decision making.

Methodology

A dairy and a pig farm were selected for the study. The 101 ha dairy farm comprised 185 cows and 50 followers, housed in cubicles, with an uncovered slurry lagoon and small farm yard manure (FYM) store. The intended mitigation strategies for the dairy farm were that slurry was to be applied by shallow injection and FYM applications would be ploughed in within 24 h. The pig farm was a 500 sow unit with 1,000 places for fatteners <20kg and 800 places for fatteners >20 kg housed in a variety of naturally and mechanically ventilated buildings. The mitigation strategies implemented on the pig farm were that the slurry lagoon was covered, slurry was applied using a band spreader (trailing hose) and FYM was ploughed in within 24 h of application.

Measurements of NH_3 emissions were made from the major sources on the two farms (excluding grazing cattle on the dairy farm) over a period of two years, using established measurement techniques. Five models were

used to estimate total annual emissions from the farms and assess the impacts of the mitigation strategies employed. The UK Ammonia Emissions Inventory (UKAEI, Pain et al., 1998) and the NARSES (Webb and Misselbrook, 2004) models are national scale inventory models. UKAEI uses fixed emission factors expressed per animal, whereas NARSES uses a mass-flow approach with emission factors expressed as a percent of the available N at each management stage. MAST (Ross et al., 2002), was developed from these inventory models for use specifically at the farm scale. The Farm Emissions Model (FEM, Pinder et al., 2004) is a process-based model specifically for dairy farms (and therefore not applied to the pig farm). The final model used in the study was the Multiple Environmental outcomes from Agricultural Systems (MEASURES) model which gives long-term whole farm analyses including interactions with profitability and other pollutant losses.

Results

There were very large differences between models used in the study in the estimates of total emissions from the two study farms, giving ranges of 3,400 – 7,100 and 9,300 – 21,700 kg NH_3 per year for the dairy and pig farms, respectively (Tables 1 and 2). Modelled reductions in emission following implementation of the discussed mitigation strategies again varied widely, in the range 11 – 50% and 5 – 36% for the dairy and pig farms, respectively. The models using nationally derived empirical mean emission factors (MAST, UKAEI and NARSES) might not be expected to give accurate predictions at a farm-specific scale. It is recommended that MAST be updated to include current national emission factors and changes in inventory methodology, making it useful as a tool for scenario analyses using standard farm types. FEM is a model specifically for slurry-based dairy farms and the predictions compared reasonably well with measurement derived values. MEASURES gave reasonable predictions for the dairy farm, but considerably over-predicted emissions from the pig farm, mostly due to it being unable to account for different livestock categories.

It is recommended that further research is aimed at producing robust, process-based models for predicting farm-specific whole-farm NH_3 emissions. Further development of fully integrated multi-functional models is required to aid policy development aimed at reducing all forms of pollution from livestock agriculture.

Table 1. Model estimates (kg NH_3) for each source on the dairy farm.

Source	MAST	UKAEI	NARSES	FEM	MEASURES	Measured
Base scenario:						
Grazing	110	600	730	220	380	ND
Housing	1430	2130	1520	1190	1890	1140
Yards	-	480	840	-	1370	630
Storage	410	340	2690	630*	990	1290†
Spreading	1430	2490	1320	2160*	2030	1310
TOTAL	3380	6040	7100	4200	6650	4370#
Mitigation scenario:						
Spreading	860	860	450	60*	1270	
TOTAL	2810	4410	6230	2100	5890	
% reduction	17	27	12	50	11	

ND, not determined; *FEM does not include emissions from FYM storage or spreading; †For slurry storage only, no measurement data for FYM heap. #Excluding grazing and FYM storage

Table 2. Model estimates (kg NH_3) for each source on the pig farm.

Source	MAST	UKAEI	NARSES	MEASURES	Measured
Base scenario:					
Housing	4410	5160	5770	9140	5690
Yards	ND	10	ND	ND	1940
Storage	1380	1520	1250	10050	ND
Spreading	3970	2610	2320	2540	ND
TOTAL	9760	9300	9350	21730	
Mitigation scenario:					
Storage	530	400	540	9880	4950
Spreading	1350	1700	1840	1670	1290
TOTAL	6290	7270	8150	20690	13870
% reduction	36	22	13	5	

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Physical assessment of the environmental impacts of centralised anaerobic digestion (Holsworthy in North Devon, England)

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Introduction and approaches

Centralized anaerobic digestion (CAD) has already been adopted in other parts of Europe (e.g. Denmark and Germany), but so far, only one CAD Plant has been built and commissioned in the UK, near Holsworthy in Devon. The aim of this research was therefore to complete a Life Cycle Assessment (LCA) of this Plant to enable comparison with other manure management strategies (see Sandars *et al*. 2003), and to help in assessing the possible impacts of other CAD plants that may be proposed in the future. Accordingly, along with farmer interviews and other data sources, an emissions and biogas monitoring system was designed, installed and commissioned and operated 01/01/2003-30/04/2005. .

CAD Plant design, operation and monitoring

When monitored, the Plant treated approximately 277 m^3 /day of input materials, comprising: 57% farm slurry, 19% blood, 11% food waste, 8% chicken manure and 5 % other non-farm wastes. On average, these inputs produced 10,085 m^3/day of biogas when monitored between March 2004 and April 2005, which generated approximately 1.32 MW of electricity, corresponding to 3.1 kWh/m^3 of biogas. Plant operation and the extra road transport required (mainly slurry and digestate) consumed 9.7% and the equivalent of 5.1% to 8.6%, respectively, of the gross electricity produced by the Plant.

Handling of input materials and digestate in the Reception Hall (RH) was the main source of fugitive emissions from the Plant. Although this included a ventilation system that exhausted 1.2 air changes per hour (chg/hr) through a wet scrubber and biofilter (WSB), the total flow increased to 2.4 chg/hr during working hours, as the RH doors opened for vehicles. Thus, although the WSB reduced the concentrations of carbon dioxide (CO_2), ammonia (NH_3) and methane (CH_4) by 84%, 87% and 44% respectively, only half of the air from the RH passed through the

WSB, leading to corresponding annual emissions from the RH and WBS of 1091 t, 0.36 t and 21 t, respectively. The open doors emitted 70% of the carbon dioxide, 95% of the NH_3 and 36% of the methane. Options for reducing the RH and WBS emissions were reviewed.

Sample tracer gas (SF_6) studies confirmed that biogas was not leaking from the digesters. The average concentrations of CH_4, CO_2, and NH_3 in the biogas were 51%, 28% and 0.05% (v/v), with most of the remainder being water vapour. Thus, the ratio of CH_4 to CO_2 was 64% to 36%, as expected. Modelling of air quality impacts from the Plant indicated minimal levels of ground level contamination from carbon monoxide, oxides of nitrogen, hydrogen sulphide, mercaptans and odours. Nearby sites of wildlife and notable species were predicted to lie beyond the deposition envelopes.

Effects on farms enrolled with the CAD Plant

Enrolment with the CAD Plant improved most on-farm slurry-handling facilities, thus avoiding the need for "emergency spreading". Most farmers benefited from access to new on-farm digestate stores. Where these served outlying areas, farmers could use biologically sourced plant nutrients (i.e. digestate) for the first time, and so use less inorganic fertiliser. These stores allowed more use of energy-efficient umbilical handling systems (20.3 MJ/m^3) replacing vacuum tankers (69.4 MJ/m^3).

The mean total solids (TS) and total phosphate (P_2O_5) concentrations in the digestate were between the typical mean values for pig and cattle slurries. The total nitrogen (TN) concentration of digestate was approximately 64% higher than cattle slurry, 33% higher than pig slurry and 43% lower than poultry (layer) slurry, on a fresh weight basis. Hence, by using digestate, a sample set of three farms gained the potential to reduce annual fertiliser purchases to around 12 kg [P_2O_5]/ha and 167 kg [K_2O]/ha, on intensive grassland production system.

Many of the enrolled farms adopted new approaches to achieve light applications of digestate, including band-spreaders, which reduced "scorch" (i.e. short-term destruction of chlorophyll following surface application of digestate). Fresh digestate had an average pH of 8.2, which was higher than the original farm slurries, (typically pH 7.5), and was estimated to increase NH_3 emissions by up to four-fold compared with pig slurry. However, low-emission storage or spreading systems were not a

priority issue amongst farmers, except (rarely) where odours had caused complaints.

Results and discussion

Implications for the wider community - Overall, the CAD Plant was found to reduce greenhouse gas emissions by 12,100 tonnes CO_2 eqv (100 year global warming potential, GWP) and if the enrolled farmers use a very high proportion of the nitrogen in the digestate to replace inorganic fertiliser, nitrate eutrophication of water may be reduced by 48 tonnes NO_3 eqv. However, the Plant increases environmental acidification by emitting 310 tonnes SO_2 eqv, mostly as increased NH_3 losses during the storage and use of digestate. These emissions also increase nutrification by 59 tonnes PO_4 eqv. Thus, for every 1000 tonnes of each total Western European burden, the CAD Plant reduces GWP by 2.5 kg and eutrophication by 2.2 kg, but increases acidification by 11.4 kg and nutrification of terrestrial habitats by 4.7 kg. If the elimination of each burden carries the same priority, then the increases in acidification and nutrification substantially outweigh the GWP and eutrophication benefits. Building the proposed 400 kW district heating system would improve the GWP benefit by 602 t CO_2 eqv (i.e. 5%) and reduce acidification by 3%.

Annually, the Plant's atmospheric emissions plus the associated on-farm operations save 670 t CO_2, 410 t CH_4 and 2.8 t N_2O and 28 t NO_3, but emit an extra 175 t of NH_3. Nationally, these represent a saving of 0.0001% CO_2, 0.02% CH_4 and 0.002% N_2O, but an increase of 0.06% NH_3. In other words, 50 such CAD plants would be needed in the UK to reduce the national CH_4 inventory by 1%.

Enrolled farmers appear to supply more P_2O_5 than they receive in digestate, since the digestate accounted for only approximately 65% of the P_2O_5 in the slurry supplied to the Plant. The difference could be due to progressive accumulations of solid matter in the digesters or elsewhere.

Sensitivity Analysis - The LCA included some important parameters that had to be assumed rather than measured. The sensitivity of the LCA to these parameters was tested as follows.

The Plant might produce between 43% and 66% of the maximum theoretical CH_4 yield, thus affecting subsequent methane losses from

stored digestate. Acidification and eutrophication were largely unaffected within this range, but higher yields increased the GWP benefit by 7%.

The increased pH due to digestion increases the risk of NH_3 volatilisation, during digestate storage, by a factor between 1 and 4, compared with undigested pig slurry. This leads to maximum increases in acidification and nitrification of 76% and 72% respectively. Nevertheless, this loss of NH_3 improved the reduction in eutrophication by 25-fold. GWP was largely unaffected.

The enrolled farmers indicated that they used less inorganic nitrogen fertiliser, although the amounts replaced with digestate were variable. If only 25% of the crop-available nitrogen in the post-application digestate were utilised (as typical of many slurries), then the CAD Plant would increase eutrophication by 388 t NO_3 eqv. However, increasing this to 75% nitrogen utilisation would reduce eutrophication by 48 t NO_3 eqv. The "neutral point", with no net change in eutrophication, was 69.5%. GWP and acidification are largely unaffected by this factor.

Some environmental improvements could be achieved by operational changes, as follows.

<u>Increased abatement of NH_3 emissions from digestate stores</u> (e.g. by fitting covers); 95% abatement would achieve net reductions in acidification and nitrification of 105 t SO_2 eqv and 17 t PO_4 eqv, respectively. However, eutrophication would increase to 193 t NO_3 eqv due to extra nitrate leaching.

<u>Increased abatement of NH_3 emissions from land spreading of digestate</u> (e.g. by deep injection); 85% abatement would yield net reductions in acidification and nitrification of 370 t SO_2 eqv and 86 t PO_4 eqv, respectively, but eutrophication would increase to 407 t NO_3 eqv.

<u>Combined improvements</u> - The GWP benefits of the CAD Plant can be achieved without any acidification, eutrophication and nutrification disbenefits. Firstly, enrolled farmers need to use at least 85% of the theoretically available nitrogen in the digestate to replace inorganic fertiliser. Secondly, land spreading techniques need to abate 70% of the ammonia emissions compared with low trajectory splash plate applicators. Thirdly, ammonia emissions from digestate stores must be 50% of those

from uncovered tanks. There is a trade-off between land spreading and storage abatement. With 60% storage abatement, the minimum spreading abatement falls to 65%. Conversely, 35% storage abatement would require 80% land spreading abatement.

The establishment of further CAD systems in the UK might be guided by the following issues.

Changes in the transport distances for slurry and digestate would have relatively small impacts, e.g. tripling the transport distances at the Holsworthy CAD Plant would reduce the net GWP benefit by 6%, increase net acidification and nutrification by 3% and 2%, respectively, and leave eutrophication largely unaffected. Therefore, there is some flexibility when considering new CAD Plant locations, although the costs of road transport and other effects on local infrastructure would impose greater constraints than the isolated implications of environmental emissions.

Use of digestate in arable systems - Grassland systems and spring sown arable crops, such as maize, provide good opportunities to utilise the highly available nitrogen in the digestate. However, most arable crops are winter-sown, so arable use of digestate brings disbenefits. For example, 4.8% of the Holsworthy digestate goes to arable land, if this increased to 80%; the eutrophication benefit of 47 t would become a 240 t additional NO_3 eqv burden. Concurrently, the other environmental burdens would deteriorate by 4 to 5%.

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Summary

Efficient use of agricultural residues and imported waste materials within agriculture is increasingly viewed from a whole-farm perspective.

The 12th International Conference of the Ramiran network in Aarhus, Denmark (11-13 September 2006) considered effects of management and technology on environmental impact and nutrient value of animal manure and other residues recycled within agriculture. Methods to describe and quantify effects of a given strategy or treatment practice at the farm level were discussed.

The Network on Recycling of Agricultural, Municipal and Industrial Residues in Agriculture (RAMIRAN) is a voluntary, FAO-based research cooperation among scientists involved in research in food or agriculture across Europe.

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