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Preface

This book presents the results of 3 years of co-operation between 13 institutes from nine European countries. A major objective was the collation of data from studies of ammonia emission from applied animal manure and the construction of a database containing the collated data. The group also described technologies for the reduction of ammonia emission from applied animal manure and conducted an analysis of the cost of using the abatement techniques. The report presents R& D requirements, principles for designing models and decision support systems and also a protocol for measuring ammonia emission from manure applied in the field.

Achieving these objectives led the group through the challenging, but fruitful process. This was particularly true for the process of developing a common data format for the results database. For practical reasons, not all factors recorded in all experiments could be stored, and yet the database had to store data from experiments with very different objectives in a way that still enabled meaningful analysis and interpretation. This focused minds on the fundamental processes determining ammonia volatilisation from field-applied manure. This in turn led to the development of a common conceptual model describing major mechanisms determining ammonia loss from field applied manure and the development of common guidelines for carrying out research. This will contribute to more multi-purpose research and lead to enhanced collaboration between institutes in Europe. The collation of data and the economic assessment have also given an insight in differences in concepts of research and governance within Europe.

The combination of data in a common database enables a more rigorous comparison of results and measurement methods than is possible by examination of the scientific papers reporting the results of individual experiments. This means that errors, inconsistencies and systematic differences can be more readily identified.

The database and web pages (www.alfam.dk) will be maintained after the end of the project. We believe this will continue to encourage collaboration between institutes from different European countries and encourage researchers to design experiments that improve our understanding of the fundamental processes involved in ammonia volatilisation in addition to solving specific problems that are restricted to particular geographic areas. We also believe that the databases will be useful for testing effects of new regulations, for developing consensus for new EU-regulations and for the development of management and decision support models dedicated to different agro-ecological zones in Europe.

The project has been carried out with financial support from the Commission of the European Communities under the work programme FAIR No 4057. The project does not necessarily reflect the Commission's views, and in no way does it anticipate its future policy within this area.

Summary

Animal production is the main source of atmospheric ammonia in Europe. The volatilisation or emission of ammonia from field-applied animal manures, which accounts for approximately 40% of the total agricultural emissions, not only represents a significant financial cost to the European farmers in terms of the loss of plant available nitrogen but also results in environmental pollution. The negative impact of ammonia deposition on sensitive ecosystems forms the background to the United Nations Economic Commission for Europe's 1999 protocol to abate acidification, eutrophication and ground level ozone that was adopted in Gothenburg and to the proposed EU directive on emission ceilings.

The potential for ammonia emissions from field-applied manures varies greatly, depending on a variety of factors, including the application technique and the weather. Consequently, it is difficult to reliably predict the expected contribution of manure nitrogen for crop production. It is not surprising that farmers generally tend to ignore or underestimate the fertiliser value of manure nitrogen and, consequently, to apply inorganic nitrogen at rates higher than those required to ensure that their crop production targets are achieved. The resulting excess nitrogen in the soil system increases the potential for other nitrogen loss pathways with negative environmental impacts including nitrate leaching and nitrous oxide emissions.

The European expertise in ammonia emissions from field-applied manure is confined to a relatively small group of researchers from the different member states. The measurement of ammonia emissions requires significant resources and limits the number of experiments that are conducted by each group. The ALFAM (Ammonia Losses from Field-applied Animal Manure) Concerted Action (FAIR6-PL98-4057) was conceived to improve the co-operation between the various national research groups and to share the research results, so that their potential can be fully exploited.

The main objectives of the ALFAM Concerted Action were to:

- Collate and analyse European data on ammonia emission from field-applied manure and to describe the relationships between agricultural practice, weather and ammonia losses
- Promote the comparison of data collected in different European countries by describing standard techniques for measuring ammonia emissions and agreeing measurement protocols
- Evaluate both the efficiency of the techniques available to reduce ammonia emissions from field-applied manure and their economic implications
- Identify the requirements for future research and development
- Disseminate the findings to policymakers and advisors.

A summary of the output from the work is presented below.

Emission Database and Data Analysis
An ammonia emission database was developed by using data from manure application experiments conducted in eight countries that represented the broad range of European climatic conditions. This database is now accessible via the Internet (http://www.alfam.dk/). The database contains data (currently *ca.* 7000 records) collected under a wide variety of environmental conditions and manure handling systems. Most of the data are for pig and cattle slurry applied to grassland and fallow soils. The records in the database were primarily obtained by using the wind tunnel or dynamic chamber technique; techniques that are most useful for testing the effect of one variable whilst keeping all others constant.

A statistical analysis of the data showed that emissions could be described mathematically by a Michaelis-Menten-type equation with the loss rate as the output variable (R^2 = 80%). The analysis also gave meaningful parameter estimates that are supported by theoretical considerations of the effect of input variables. The variables significantly affecting ammonia emissions are soil moisture content, air temperature, wind speed, manure type, dry matter content of manure, ammoniacal nitrogen content of manure, application method (including soil incorporation) and application rate. Table (0.1) presents the ammonia emission in relation to each of the parameters included in the statistical analysis. An interactive model of ammonia emission, based on the statistical analysis, is available for downloading from the website (www.alfam.dk).

Table 0.1 Factors affecting ammonia volatilisation from animal slurry applied to fields

Experimental factor	Effect on NH_3 volatilisation
Soil moisture	Wet soil 10% higher than dry soil
Air temperature	+2% per °C
Wind speed	+4% per m/sec
Slurry type	Pig slurry 14% less than cattle slurry
Dry matter content	+11% per % DM
TAN content	-17% per g N/kg
Application method:	
Band spreader/trailing hose	42% less than broadcast spreader
Open slot injection	72% less than broadcast spreader
Manure incorporation.	No incorporation 11 times higher than shallow cult.
Measurement tech.	No significant difference between wind tunnel and micrometeorological method. Lennart boxes about 81% higher than these two.

Methods for Measuring Ammonia Emissions
A number of methods have been developed and are used for the measurement of ammonia emissions from field-applied animal manure at both small plots and on a field scale. Current methods include soil nitrogen balance, micrometeorological methods, enclosures and controlled gas release methods. In addition, a number of models exist for the estimation of am-

monia emissions without direct measurement. Methodological development is a continuing process, with current emphasis on establishing a non-intrusive measurement method for emissions from small plots. A range of sensors exists for the measurement of ammonia concentration in air or fluxes. Flux samplers have the advantage of giving a more accurate integrated measurement of flux than that derived from independent measurements of concentration and wind speed. The importance of additional measurements to characterise the manure, crop, soil and environmental conditions at the time of manure application was highlighted, so that the usefulness of emission data, particularly for model development, can be increased.

The Economic Implications of New Application Techniques

New manure application techniques have been developed to reduce ammonia emissions over the last decade. However, they increase application costs. A model has been developed to calculate the costs of manure application. The input data includes field size (or area to be spread), spreader work-rate, work pattern, incorporation method and application rate. The output is the cost of manure application by different application techniques. Input data for the model was collected from a number of European countries, and the cost implications of the new application techniques were calculated.

Application costs varied from 1.7 to 13.0 Euro per m^3 manure applied. The observed variations in costs was attributable to the differences in circumstances between countries and between farms (field size, work pattern, *etc.*) within a country. Farm circumstances and the cost and type of the application techniques used have large impact on costs. Variation in costs within countries was at least as large as the variation between countries.

Analysing the factors that affect the costs was difficult due to the large variation and the small sample survey. Therefore, scenarios for a range of "typical" farm circumstances were simulated to explore the cost components of manure application and allow comparisons. The costs were calculated based on the assumption that the farmer owned the manure application machinery. The results showed that for farms with an annual manure production of 1000 to 3000 m^3, the costs of manure application by trailing hose, trailing foot, shallow injector and arable land injector were on average *ca* 2 Euro per m^3 higher than for surface spreading. The difference in costs between surface spreading and the other application techniques became smaller, *ca* 1.4 Euro per m^3, on farms with higher annual manure production, but higher, *ca* 2 Euro per m^3, on farms with an annual manure production of only *ca* 500 m^3. Manure application by a contractor may be less expensive, because a contractor may be able to use the machinery more efficiently (more working hours per year), particularly on the smaller farms.

Future Research and Development Requirements

The following research and development requirements were identified:

1. The low emission manure application techniques that have been shown to work well under controlled experimental conditions require evaluation under the less controlled conditions on farms.

2. The low emission manure application techniques that are currently available require evaluation to assess what is technically feasible under differing country (or region) specific conditions.

3. Research is required on ammonia emissions from field-applied solid manures and on the possible abatement technologies.

4. There is a requirement to develop application techniques, so that manure can be utilised on more mature crops without causing damage.

5. A European analysis of the likely cost of implementation of the different emission reduction techniques is required, with additional data collected by surveying. This analysis should consider the options available to reduce the cost of low emission technology to farmers in general and to small farmers in particular.

6. There is a requirement for cross-calibration of the different measurement methods to enable the comparison of results from different experiments. A standard range of soil, climate and manure measurements should be made in all ammonia emission experiments.

7. There is a requirement to improve the effectiveness of the transfer of existing information on ammonia emission from manures to farmers, particularly through the use of decision support systems.

8. There is a requirement to construct a decision support system to allow policy makers to assess the cost and impact on ammonia emission of different abatement scenarios.

9. There is a requirement for reliable statistical information on the spatial variation in manure handling systems and management practices within the EU and neighbouring countries to construct reliable inventories and to improve the ability of policymakers to assess the cost effectiveness of abatement measures.

10. There is a requirement to develop and adopt a standardised method for estimating the detailed emission factors for ammonia emission from field-applied manure.

11. There is a requirement to establish a European network of atmospheric ammonia monitoring as a check on the validity of emission inventory techniques.

12. There is a requirement for more integrated farm scale studies to focus on the consequences of changes in feeding systems, animal housing and manure storage on subsequent losses after field application.

13. There is a need to study the consequences of changes in manure application practice on other gaseous emissions (*e.g*. methane, nitrous oxide) and on leaching and crop uptake.

Dissemination of Findings

The dissemination of the ALFAM results has hitherto been made via the website (www.alfam.dk). This website gives an overview of the participants in the project, presents the results achieved and allows the ALFAM results database to be interrogated for different types of data. These data can be downloaded by participants (and by other researchers by agreement). The statistical model developed by using the results in the database can also be downloaded. This final report will be made available on the website in PDF format.

The website will continue to be maintained after the funding of the project has ceased. It will act as a useful portal into the European research, and this could be of particular use to US researchers, as Europe currently has a significant lead in this area.

Policy Implications – An Example

The statistical model developed as part of the ALFAM project was used to estimate ammonia emissions from standard slurry applied in Italy, England, Norway and Denmark. The objective was to examine how the rate of loss varied depending on climate and farming practice. This is a relevant question when, for example, formulating regulations on the maximum time that may elapse between broadcast spread application and ploughing. Significant variations might require that specified time be adjusted to account for these differences in such a way that a particular maximum percentage of the ammonium in the manure is lost in all areas of Europe. Climate observations from three periods (1, one week before the normal sowing time for spring crops; 2, by the time of a mid-season grass cut; and 3, one week after the harvesting of the spring-sown crop) in year 2000 were used.

The emissions were higher from cattle slurry than from pig slurry, due to the higher dry matter content and the higher viscosity of cattle slurry. The predictions showed that there was little difference in the total ammonia emissions from slurry applied in the three periods, because of interaction between soil moisture and air temperature, *etc*. Surprisingly, the time from application until 10% of the slurry has been lost is almost the same for countries in the south and in the north of Europe. This was due to an interaction between climatic variables. The low wind speeds in the Italian data (Po Valley) were balanced by the high air temperatures, while in North European countries, the situation was the reverse.

The conclusions from this analysis is that in order to reduce ammonia emissions, the slurry should be incorporated faster in mid and late seasons than in early spring, due to generally increasing air temperatures during the growing season. On the basis of these scenarios, there appears to be little justification for requiring the maximum time before incorporation to be adjusted for climate. However, it should be emphasised that such a conclusion would need to be confirmed by using weather data from a wider range of locations.

1. Introduction

Ammonia volatilisation from field applied animal manure represents a major source of atmospheric pollution and can result in a substantial reduction in the nitrogen fertiliser value of the manure for crop production. Ammonia losses vary greatly with application technique and weather, reducing its reliability as a source of nitrogen for crop production. As a consequence, many European farmers do not rely on manure nitrogen to make a contribution to crop nitrogen requirements. Consequently, many European farmers do not rely on manure to supply the nitrogen requirements of their crops. They tend instead to apply the crop's nitrogen fertiliser in the form of inorganic nitrogen and to ignore most or all contributions from the manure. The potential exists under such circumstances for the surplus nitrogen in the soil system to contribute to increased losses through nitrate leaching to water and nitrous oxide emissions to air.

Concern for the impact of atmospheric ammonia forms the background to ongoing international negotiations for reduction of national ammonia emissions. In addition, existing and proposed EU and national legislation relating to other undesired effects from manure application (*e.g.* nitrate leaching) will increasingly restrict organic and inorganic nitrogen applications to agricultural land. This will encourage farmers to adopt strategies and technologies that will reduce emissions of nitrogen and increase the use of manure nitrogen by the crop.

The mechanisms of ammonia volatilisation from field applied animal manure and the development of abatement strategies has been the subject of research in many European countries. Measuring ammonia volatilisation following land application is labour demanding and expensive. Therefore, the number of studies undertaken within a given country has been curtailed to a limited range of soil and climatic conditions, and generally, each experiment was designed to examine one or two of the most important contributory factors. In contrast, the users of the data generated often require an evaluation of ammonia volatilisation under a wide range of climatic, soil and agronomic situations, even within a single region. Alone, the individual European studies do not provide an adequate basis for providing the level of information currently required by policy makers and agronomists or for the development of national guidelines for best agricultural practice.

The value of national research data relating to ammonia emissions could be greatly increased if the country specific data were combined with those from other European countries into a single database. A primary objective of this Concerted Action Project was the development of such a database – the ALFAM database. The ALFAM database provides a basis for a more complete description of the factors determining ammonia loss than is possible with the results from national research alone. This ALFAM database will also be useful both for developing mechanistic models, designed to encapsulate the state of knowledge relating to ammonia losses from fields applied manure and in the construction of decision support systems (DSS).

This report provides a description of the new database of results from studies of ammonia emission from the field application of manure. An analysis of the loss pattern in relation to measurement technique and manure types is presented. A model linking ammonia volatilisation to climate data is tested and used to estimate the efficiency of abatement techniques at a regional level. The report also presents a review of the abatement techniques and an economic analysis of the costs of introducing abatement techniques in European countries. Finally, protocols for data collection during ammonia emission studies, the principles for developing DSS and a conceptual model of ammonia emission from applied manure are described.

2. Ammonia losses from field applied manure – A conceptual model

2.1 Introduction

The twin objectives of the conceptual model were to provide a conceptual framework:

- for structure for the ALFAM database
- the specification of a decision support system (DSS) that could be used to assist in either predicting the fertiliser nitrogen value of a manure application or the ammonia emission reduction achieved by adoption of certain technologies losses from field-applied animal manure.

The purpose of this chapter is to provide a brief overview of the major factors affecting ammonia volatilisation from field applied animal manure. The subject has been reviewed in more detail by others.

2.2 The conceptual model

Field-applied manure can be considered as a volume of water containing ammonium that covers a certain area. This concept is readily applicable to liquid manure or slurry, but can also be used to describe the wet surface of solid manure particles.

Ammonia volatilisation is a surface phenomenon; the current (instantaneous) volatilisation is a product of the volatilisation per unit area of solution and the area of solution exposed to the atmosphere. The total volatilisation from a particular manure application is the instantaneous volatilisation integrated over the duration of emission event. The total volatilisation can be considered largely to be the result of the competition for ammonium between the processes driving volatilisation and those determining its removal to other parts of the plant/soil system.

The ammonium originates either from the transformation of the urea in animal urine or the uric acid in poultry droppings or from the mineralisation of the more stable organic nitrogen compounds by micro-organisms. The hydrolysis of urea or uric acid to ammonium is sufficiently rapid that the process will generally be complete before manure is taken to the field. The dry matter of poultry manure determines the rate of hydrolysis of uric acid with lower rates occurring in high dry matter manure. The extent to which more stable organic nitrogen will mineralise during storage varies widely. Organic nitrogen is generally mineralised very slowly in liquid manure or slurry, whilst substantial amounts can be transformed in solid manure if it composts during storage.

2.3 The instantaneous volatilisation rate

Following field application, the ammonium in solution dissociates reversibly to ammonia (also in solution). The balance is determined by the chemical composition of the manure, especially the pH, and the extent of the interaction between the applied manure and the soil. The ammonium at the surface of the solution is in dynamic equilibrium with ammonia gas in the air, the balance being determined by the temperature. The ammonia at the surface of the solution is depleted by volatilisation, immobilisation by incorporation into organic matter, nitrification and plant uptake. Ammonia in the solution can be replenished from the mass of manure/soil by diffusion, by mass (capillary) flow, hydrolysis of uric acid, mineralisation of organic matter or by evaporation of water causing the surface to retreat downwards (slurry) or inwards (solid manure). The factors and the direction in which they effect the concentration of ammonia at the liquid surface are shown in Table 2.1.

Table 2.1 Primary and secondary factors affecting the concentration of ammonia at the liquid surface of manure with an indication of the direction of the impact

Primary factors		Secondary factors	
+	Ammonium concentration	+	Urea/uric acid decomposition
		+	Mineralisation of manure organic matter
		+	Evaporation of water
		-	Volatilisation
		-	Rain/irrigation/dilution
		-	Immobilisation/nitrification
		-	Binding to soil
+	pH	+	Urea/uric acid decomposition
		-	Carbon dioxide flux
		-	Acidification
+	°C	+	Solar radiation
		+/-	Interaction with soil
		-	Evaporation of water
		-	Rain/snow
		-	Turbulent heat transport in atmosphere

Note: + = positive effect on volatilisation *i.e.* increases the rate: - = negative effect on volatilisation *i.e.* reduces the rate.

This ammonia gas is transported from the solution surface, either vertically, by turbulence, or horizontally, by the advection of air from upwind of the area to which manure has been applied. The factors affecting and the direction in which they affect the ammonia transport in the atmosphere are shown in Table 2.2.

The area of solution exposed to the atmosphere depends on the size of the area to which manure is applied and the cover ratio. The latter is a measure of the extent to which the manure covers the area to which it has been applied (area covered by manure per unit ground area). If the spreading is done by use of a method that deposits manure in clumps or strips, the cover ratio will be less than 1. Alternatively, if the surface of the field is rough or a crop canopy intercepts some of the manure, the cover ratio may exceed 1. The factors that determine the area of solution exposed to the atmosphere and the nature of their effect are shown in Table 2.3.

Table 2.2 The primary and secondary factors affecting the ammonia transport in the atmosphere with an indication of the direction of the impact

Primary factors		Secondary factors	
+	Surface roughness	+/-	Crop cover and complexity
		+	Topography
+	Windspeed		
+	Solar radiation		
-	Crust formation	+/-	Quality of dry matter
		+	Quantity of dry matter
		+	Application rate

Note: + = positive effect on ammonia transport *i.e.* increases transport rate, - = negative effect on ammonia transport *i.e.* reduces the transport rate.

Table 2.3 The primary and secondary factors affecting the area of liquid exposed to the atmosphere with an indication of the direction of the impact

Primary factors		Secondary factors	
+	Total area to which liquid is applied		
+	Foliar contamination	+	Crop canopy cover and complexity
		+/-	Quality of dry matter in liquid
		+	Quantity of dry matter in liquid
		-	Rainfall
-	Band spreading, injection		

Note: + = positive effect on the area of liquid exposed *i.e.* increases the area of liquid exposed, - = negative effect on the area of liquid exposed *i.e.* reduces the area of liquid exposed.

2.4 Duration of volatilisation

Ammonia volatilisation continues for as long as there is ammonium in solution that is exposed to the atmosphere. In theory, volatilisation could be stopped by the depletion of all the ammonium. In practice, although depletion normally causes volatilisation to fall rapidly from an initially high rate, it is normally ended by factors other than depletion. These can be that the surface dries out, the remaining solution infiltrates into the soil or the manure is ploughed in. The factors affecting the duration of volatilisation are shown in Table 2.4.

2.5. Outline of decision support systems (DSS)

Two primary users of DSS for ammonia emissions were identified. These are the farmer/advisor and the policy maker. There is a requirement for two DSSs, as the objectives for and outputs from both are different.

2.5.1 Farmer/advisor

The objective for this DSS is to provide a tool for use by individual farmers or advisors that would predict the fertiliser nitrogen equivalent of a given animal manure applied at a given rate, by using a given application method and under given weather, crop and soil conditions. A validated DSS of this nature could increase the farmer's confidence in animal manure as a nitrogen fertiliser source for crop production. If successful, it would encourage a greater reliance on manure nitrogen for crop production than is currently the case. The DSS could also be used either during the annual nutrient management process, by estimating the nitrogen fertiliser value of a manure application, or as an educational tool to enable the potential benefits of different manure application techniques to be illustrated.

Table 2.4 Primary and secondary factors affecting the duration of volatilisation with an indication of the direction of the impact

Primary factors		Secondary factors	
+	Application rate		
-	Burial of manure	+	Post application tillage
-	Infiltration rate	+	Sand versus clay
		+	Pre application tillage
		+	Rainfall
		-	Frozen soil
		+/-	Soil moisture content
		+/-	Quality of dry matter in liquid
		-	Quantity of dry matter in liquid
-	Evaporation rate	+	Surface roughness, wind speed, solar radiation

Note: + = positive effect on the duration of volatilisation *i.e.* increases the volatilisation period, - = negative effect on the duration of volatilisation *i.e.* reduces the volatilisation period.

A framework for such a decision support tool is proposed and outlined in the box below.

Decision Support System Framework for farmers/advisors

DSS Inputs

1. *Manure characteristics:* Manure analysis (dry matter, total and ammoniacal nitrogen content, phosphorus and potassium concentrations. In absence of this data the following information is required **a)** type (*e.g.* slurry, solid manure); **b)** animal type; **c)** type of animal housing and manure storage.

2. *Application details:* **a)** method and rate of application; **b)** type and timing of cultivation (if any); **c)** crop type and height; **d)** distance to field.

3. *Soil and weather conditions:* **a)** soil type and initial soil moisture status (wet, dry);
   b) expected weather for next 24 hours (temperature, wind speed, rainfall).

DSS Outputs

1. An estimate of the nitrogen fertiliser value of the manure application (kg ha^{-1}).
2. An estimate of the quantity of ammonia lost from the manure application (kg ha^{-1}).
3. The percentage of total ammonia applied in the manure application lost to volatilisation.
4. The cost of the application.

DSS Format

Probably an Internet based system in which a core model resides on one server, and this is then linked to a single server in each of the participating countries. These country-based servers would host an interface in the local language.

2.5.2 Policymakers

The objective of a DSS for policymakers is to assist the evaluation of ammonia emission abatement scenarios for the various regions within Europe, both in terms of the reductions achieved and the cost implications for farmers.

A framework for such a policy maker's DSS is proposed and outlined in the box below. The choice of inputs is based on the results of the data analysis and the conceptual model. Note that much of the input data need only be entered once.

Decision support system framework for policymakers

DSS Inputs (For each region of interest, the following would be required)

1. *Agroclimatic data:* **a)** Crops (% distribution by area); **b)** Crop management (ploughing, sowing and fertilisation dates); **c)** Climatic data (temperature, rainfall, windspeed); **d)** Soil type (including texture and stoniness); **e)** Topographic data (mainly slope); Limitations of each application technique (*e.g.* slope, stoniness).

2. *Animal data:* **a)** Census data (number of dairy cattle, pigs *etc*); **b)** Excretion data (manure produced and nitrogen excretion rate for each animal type); **c)** Proportion of solid or liquid manure for each animal type.

3. *Economic data:* **a)** Operating costs of machinery (labour, machine costs, fixed and variable costs); **b)** Time required for spreading operations (tanker filling time, travel time to and from field, *etc*)

4. *Manure application strategy:* **a)** Manure application rate for each crop type; **b)** Manure application method

DSS Outputs

Possible outputs could be produced for each region:

1. An estimate of the total ammonia emission for the region.

2. The percentage of total manure nitrogen available for crop utilisation.

3. An estimate of the cost per hectare of the particular strategy both on a regional and farm type basis.

3 Ammonia losses from field applied manure – The ALFAM databases

3.1 Introduction

Two databases were constructed during the ALFAM project. The first was a database of current and recent research activities, and the second was a database of measurements from experiments of ammonia volatilisation from field-applied manure.

3.2 ALFAM research activity database

Research activities are described in terms of projects, where each project will typically describe a series of field experiments or other similar research activities. The aim was to minimise the amount of information required – demanding extensive details was felt likely to be a disincentive.

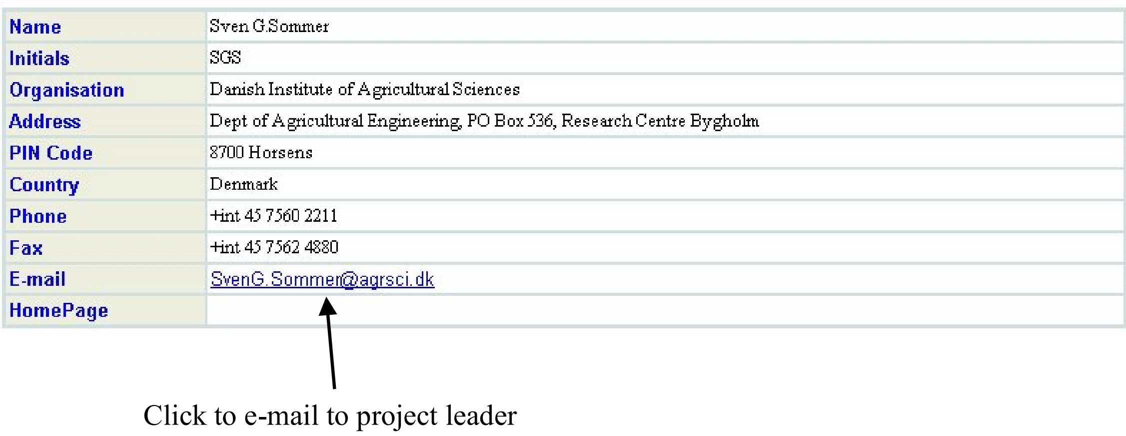
To enter details of a project, a user must first register as a project leader. Registration involves providing the details that will be displayed to users who are interested in the project(s). After registration, the project leader will receive a password by e-mail. The purpose of this password is to prevent anyone but the project leader from altering the relevant data. The ALFAM webmaster is also notified of the registration. Once registered, the project leader can add details of one or more projects. If, at a later date, the project leader wishes to edit or delete details of a project, he first has to log in by using his password. If a project leader forgets his password, he can request that a copy be e-mailed to his registered e-mail address.

The ALFAM activity database has been fully functional since August 2000, and thus far it contains details of 14 research projects. The website enables users to list all the projects or to search for key words in the project description and list only those projects. From the listing, it is possible to obtain further details of the project and the project leader or link to the project home page (Figure 3.1 and 3.2).

The project leader details are as shown below:

Project Name	Begin	End	Web URL	Man Months	Project Leader	Project Details
ALFAM	01/04/99	31/03/01	http://www.alfam.dk	36	Sven G.Sommer	Show

Figure 3.1 Example of project leader information (summary level)



Name	Sven G.Sommer
Initials	SGS
Organisation	Danish Institute of Agricultural Sciences
Address	Dept of Agricultural Engineering, PO Box 536, Research Centre Bygholm
PIN Code	8700 Horsens
Country	Denmark
Phone	+int 45 7560 2211
Fax	+int 45 7562 4880
E-mail	SvenG.Sommer@agrsci.dk
HomePage	

Figure 3.2 Example of project leader information (detailed level)

3.3 ALFAM results database

The process of collating and presenting the data to be stored in the ALFAM database developed from two workshops involving all participants, as well as individual contact between the project co-ordinators and participants. This initial phase was extremely valuable and important. It ensured that the data eventually collated represented the range of variables that effect ammonia emissions across Europe. Participants were obliged to focus on the type of data (variables and units) that would be included and, equally importantly, those that should be excluded. Participants were also able to agree upon the format for data to be submitted to the co-ordinators for entry into the database. A blank data entry file (Microsoft Excel format) can be downloaded from www.alfam.dk.

From an early stage, the participants agreed that the ALFAM database should be accessible on the World Wide Web. It was also decided that a summary of each dataset should be available to all users, but that access to the raw data should be restricted to those researchers that had contributed with data (in practice, this means mainly the ALFAM participants). Microsoft Access was chosen as the database format, as it was to be made accessible on the Internet via a Microsoft NT web server. However, data is downloaded in comma-delineated format, as this can be imported into many database and spreadsheet applications.

A summary of the institutes that provided data for inclusion in the ALFAM database (*ca.* 5900 records) before February 2001 are listed in Table 3.1. This data set formed the basis of the statistical analysis presented in Chapter 4. Further data (*ca.* 1400 records) have been included in the ALFAM database since then. The most recent update of the ALFAM database can be found at www.alfam.dk.

3.4 Description of the ALFAM database website

The ALFAM website includes the ALFAM database and a data catalogue that contains a summary of the information pertaining to each study contained in the database. This catalogue includes the following information:

1. Data ownership
2. Study objective (30 words)
3. Where the data is published
4. Link to Project Leader (e-mail address)
5. The number of records in the dataset
6. Variables measured
7. Manure type used
8. Application technique

The website user can search the ALFAM database by using these criteria, but only the ALFAM participants can request that the raw data within these data sets be downloaded. The participants may only use the data if they obtain permission from the scientist and the institute responsible for the generation of the original data.

Table 3.1 A summary of the Institutes that provided data for the ALFAM database (prior to February 2001) with a short description of each experiment

Institute & Country*	Manure type	Application technique	Measurement technique	Short description of data
Danish Institute of Agricultural Sciences, Denmark	Slurry	Splash plate & Trailing hose	Wind tunnel (1) Micrometeorological technique (2)	1) Effect of harrowing before slurry application (Wind tunnel). 2)Trail hose and splash plate application, effect of climate and trail hose application, effect of crop height and acidification (Micrometeorological technique)
Nutrient Management Institute & Institute of Agricultural and Environmental Engineering (IMAG). The Netherlands	Slurry	Injection – shallow & deep; Trailing shoe Splash plate.	Micrometeorological technique (2)	Effect of application method, rate of slurry application and climate.
Swiss Federal Research Stations for Agro-ecology & Agriculture and for Agricultural Economics and Engineering; Swiss College of Agriculture. Switzerland	Slurry Solid Sewage sludge	Splash plate	Wind tunnel (1) Micrometeorological technique (2)	Manure types, surface applied (one experiment with incorporation), climate, grass and stubble (wind tunnels and micrometeorological techniques)
Agricultural University of Norway, Norway	Slurry	Injection (Pressure) Trailing hose Splash plate	Micrometeorological technique (2)	Application technique (incl. pressurised injection), manure type, grassland and fallow soil
Institute of, Grassland and Environmental Research (IGER) England	Slurry Solid	Splash plate; Trailing hose & shoe; Injection – shallow & deep.	Wind tunnel (1) Micro- meteorological technique (2)	Study of effect of manure composition (solid and slurry), climate and grass (wind tunnel); Study of effect of application technique - band spreading, trailing shoe and injection. Grassland (micrometeorological technique)
Centro Ricerche Produzioni Animali, Italy	Slurry	Splash plate; Trailing hose; Injection – shallow	Wind tunnel (1)	Emission from pig slurry on different crops. Application rate, crop or fallow, application technique.
Swedish Institute of Agricultural and Environmental Engineering, Sweden	Slurry Urine Solid	Splash plate; Trailing hose & shoe; Injection - shallow & deep	JTI equilibrium concentration method (3)	Application of animal slurry and urine to grass and arable land, effect of different application technique. Application of solid manure to fallow soil – effect of incorporation
ADAS, England	Slurry	Splash plate	Wind tunnel (1)	Study of the effect of wind speed and slurry composition.

(1) Wind tunnel. A portable dynamic chamber. (Ryden, and Lockyer, 1985). (3) JTI equilibrium concentration method. (Svensson and Ferm, 1993)

(2) Meteorological mass balance technique. (Ryden and McNeill, 1984).

4 The data

The ALFAM database currently contains results (7400 records as of September 2001) from studies where ammonia volatilisation was measured with micrometeorological mass balance techniques or wind tunnels. Each measurement of volatilisation was formatted as one record, giving information on volatilisation rate, climatic conditions, soil properties, *etc.* during the measurement period. The data can therefore be easily transferred to statistical analysis programs such as SAS or SPSS. Since each experiment had different objectives, some variables are missing in a number of the records. The ALFAM database variables, their units and a summary of these are given in Table 3.2. The definitions of the categories used in Table 3.2 are presented in Table 3.3.

4.1 Climatic zones represented by data

The countries that submitted data and the quantity of data supplied for the ALFAM database are shown in Figure 3.3. The Italian data derives from work conducted in the northern part of the country (Po Valley). There are currently no data from Southern or Mediterranean Europe, Germany or Eastern Europe.

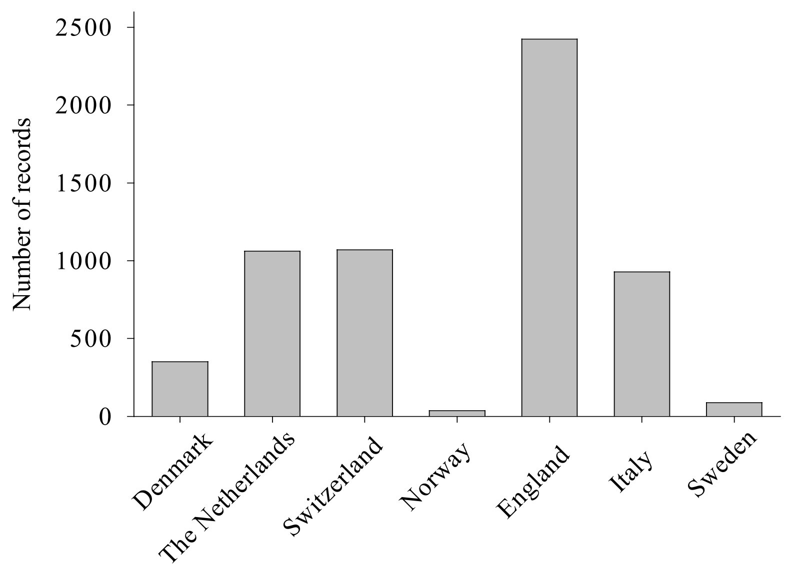


Figure 3.3 The European countries that contributed with ammonia emissions data to the ALFAM database with their respective number of records submitted.

Table 3.2 The ALFAM database variables, their units and a summary of the data available (5900 records) in February 2001.

Measurement	Variable	Unit or category range	Average	SD	Max	Min	Number of observations
Total NH_3 volatilised during the experiment	Volatilised*	kg N ha^{-1}	25.7	29.0	206.1	0	744
		% of TAN	43	26	99	0	744
Soil information	Soil type**	(1-4*)			4	1	679
	Clay**	%	31	19	62	7	407
	Silt**	%	29	12	60	11	303
	Sand**	%	39	28	80	7	321
	OM**	% C	3.1	1.8	6.5	1.0	352
	Soil water**	% v/v	26.3	15.2	67	0.005	212
	Soil moisture**	1-2*			2	1	567
	pH**		6.9	0.9	8.2	4.9	544
Climate/ Meteorology	Air temperature	^{o} Celcius	14.7	7.4	36.5	-5.7	4717
	Air temperature	height (cm)	89.6	252	1000	10	3652
	Soil temperature	^{o} Celcius	12.92	6.2	28.8	-0.1	2373
	Soil temperature	Depth (cm)	4.4	3.2	15.0	0.5	1647
	Radiation	joules $m^{-2}s^{-1}$	228.0	219.8	1042.6	0	4361
	Wind speed	m s^{-1}	2.2	1.5	9.0	0	5769
	Wind speed	height (cm)	78	89	200	0.25	5757
	Rain during	shift (mm)	0.43	1.92	19.4	0	2257
Manure information	Type**	1-9*			9	1	800
	Treatment**	0-4*			4	0	569
	Bedding**	0-3*			2	0	760
	Dry matter**	%	5.9	4.9	62.2	0.6	800
	Nitrogen-total**	g N kg^{-1}	4.3	4.8	31.9	0.2	727
	Total ammoniacal nitrogen**	g N kg^{-1}	1.9	1.3	11.8	0.1	800
	Uric acid**	g N kg^{-1}					0
	pH**		7.5	0.5	9.2	6.6	611
Manure application	Method**	0-5*			5	0	800
	Rate**	(t or m^3 ha^{-1}	38	27	315	6.6	800
	Incorporation**	0-2*			2	0	681
	Incorp. speed**	hours			4	0	80
Crop information	Type**	1-4*			4	1	800
	Height**	cm	8.4	5.1	65	0	565
Measurement	Technique**	1-4*			3	1	800
Soil treatment	Pretreatment**	1 = harrow 0 = unworked			1	0	16

* Class variables given an index number referring to the categories in Table 3.3.

** The number of observations for variables that were measured at the start of an experiment only will always be lower than the total number of records made during the experimental period (about 10% of the total number of records).

Notes:

- In a few experiments total ammonia volatilisation was higher than the initial total amomoniacal nitrogen (TAN) content of applied manure. These experiments have been omitted from the statistical analysis. The number of manure type records is therefore larger than the number of total ammonia volatilisation records.
- The average pH is the calculated without transforming pH to [H+] .
- The ammonia emission rate from one manure application will have been determined several times during the period of measurement. For each manure application, the cumulative emission was calculated by fitting an exponential model to the ammonia loss rates measured during the emission event (*cf.* Chapter 4). As a consequence, the number of ammonia loss records are lower than the number of air temperature records, as air temperature was usually determined each time the ammonia emission rate was measured (Table 3.2).

Table 3.3 Description of measurement group variables used in the ALFAM database, with the category range and index numbers for each category value

Measurement group	Variable	Category range	Index
Soil	Soil type**	1-4*	1 = Sand 2 = Clay 3 = Loan 4 = Organic
	Soil moisture	1-2	1 = Wet 2 = Dry
Manure information	Type	1-9	1 = Pig slurry 2 = Cattle slurry 3 = Pig-solid 4 = Cattle solid 5 = Poultry slurry 6 = Poultry solid 7 = Sewage sludge 8 = Liquid manure 9 = Mixed solid manure (horse, pig and poultry)
	Treatment	0-4	0 = Not separated 1 = In house separation (includes effluent from weeping wall) 2 = Mechanical separation 3 = Aerobic treatment 4 = Anaerobic treatment
	Bedding	0-3	0 = None 1 = Straw 2 = Sawdust/wood chips 3 = Paper
Manure application	Method	0-5	0 = Splash plate or broad spread (liquid & solid) 1 = Band spread/trailing hose 2 = Trailing shoe 3 = Open slot – injection 4 = closed slot – injection
	Incorporation	0-2	0 = None 1 = Plough 2 = Shallow cultivation
	Incorporation Speed	Hours	Hours between application and incorporation
Crop	Type	1-4	1 = Grass 2 = Stubble 3 = Bare soil 4 = Growing crop
Measurement	Method	1-4	1 = Wind tunnel 2 = Micrometeorological mass balance technique 3 = JTI or equilibrium concentration method
Soil treatment	Pre-treatment	0-1	0 = Unworked 1 = Harrow

4.2 Soil and climate characteristics

The manure characteristics and environmental variables recorded in the ALFAM database are summarised in Table 3.2. The ALFAM database contains data for a wide range of soil textures, pH and water contents. These data have all been measured or quantified at the beginning of the experiment, only, and it should be noted that the initial soil water and topsoil pH may not be reliable indicators of the water content and pH during the course of the experimental period (Sommer & Olesen, 2000). By contrast, records of ammonia volatilisation data are available for all periods of all experiments.

The data also covers a wide range of weather conditions. Most climate variables were determined for each period of ammonia emission measurements during the experiments. The air temperature, soil temperature and wind speed have been measured at different heights/depths above or below the soil surface. These variables ideally need to be standardised to the same height/depth before they are used in comparative statistical analysis. Weather data was not always available at the experimental site. In these circumstances, the weather data was obtained from a nearby weather station. This highlighted the need for a measurement protocol applicable to all field application of manure experiments where ammonia volatilisation is to be measured (*cf.* Chapter 5).

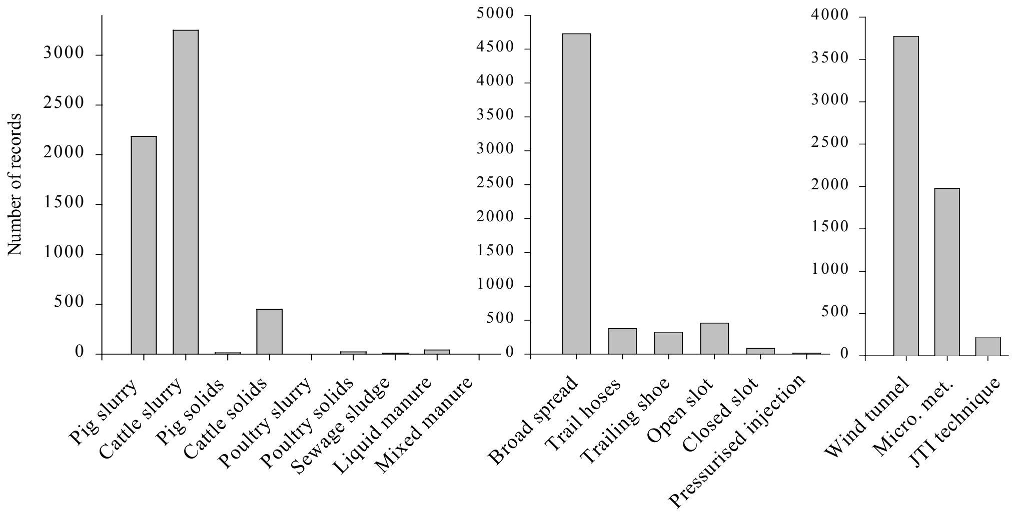


Figure 3.4 Record frequency distribution for manure types, spreading methods and measuring techniques used in the ammonia emission experiments contained in the ALFAM database.

4.3 Manure type and application technique used

The type of manure applied in the experiments is recorded for all studies in the ALFAM database. Most data are for pig and cattle slurry (Figure 3.4), with relatively few data sets for ammonia volatilisation from solid manure (pig, cattle or poultry). The pig slurry used in the Dutch experiments had a relatively high dry matter and total ammoniacal nitrogen content (Table 3.4) compared with the pig manure used in other countries. Otherwise, the ranges for these two manure variables were relatively similar for the different countries. In most studies, slurry was applied with tankers mounted with a splash plate, although experiments where injection and trailing hose application techniques were used are also included (Figure 3.4).

4.4 Measuring technique

The Dutch and Norwegian institutes tended to use the micrometeorological mass balance technique for the measurement of ammonia volatilisation from field applied manure (Table 3.1). The Danish, Swiss and English institutes used wind tunnels in addition to the micrometeorological mass balance technique. The Italian data was generated from wind tunnel experiments. Data from Sweden was generated by using the JTI equilibrium concentration method. The micrometeorological technique will give results reflecting ambient weather conditions during measurements, whereas the wind tunnel may create a microclimate within the tunnel that does not reflect ambient conditions. For example, the wind profile in the tunnel will differ from that in the open, and the tunnel will provide shelter from rain. The JTI technique is a micrometeorological technique in which a chamber is used to obtain an equilibrium concentration. The chamber will therefore provide shelter from rain during measurements, but will not affect wind speed and temperature (*cf.* Chapter 5).

Table 3.4 Summary of the type, dry matter content and total ammoniacal nitrogen concentration of the manure in the ALFAM database

Manure type	**Country**	**Dry Matter** (%)	***Total Ammoniacal Nitrogen*** g N kg^{-1}
Cattle slurry	Denmark	6.49	2.42
	Italy	ND	ND
	The Netherlands	7.76	2.18
	Norway	4.02	1.29
	Sweden	6.38	1.54
	Switzerland	3.6	0.96
	England	4.32	0.97
Pig slurry	Denmark	3.82	2.96
	Italy	3.75	1.92
	The Netherlands	10.32	5.6
	Norway	ND	ND
	Sweden	6.2	2.94
	Switzerland	3.8	1.67
	England	5.67	3.83
Cattle solid manure	Switzerland	18.8	0.96
	England	19.67	0.62
Pig solid manure	Switzerland	22.1	1

Note: ND = No data provided

4.5 Analysis of data

The strength of the data in the database is that it represents a large number of measurements taken under a wide range of conditions. However, caution must also be applied, because the data was generated under different husbandry systems and by different research groups, often with different experimental objectives. For example, there may be confounding factors between the composition of pig slurry and environmental conditions in the Dutch experiments. Furthermore, the use of different measuring techniques in different experiments should be taken into account in any statistical analysis of the data. Gaps in the data due to variables not being measured in particular experiments will often prevent the inclusion of all data in the database in a statistical analysis.

4. Ammonia volatilisation from field applied animal manure – The ALFAM statistical model

4.1 Introduction

This chapter describes the development and validation of an empirical model to predict ammonia losses from field applied manure for a range of weather, soil and management conditions. The value of the model to predict ammonia losses is also considered and discussed.

4.2 Methods

The background to and description of the ALFAM database is provided in Chapter 3 of this volume.

4.2.1 The model

The data in the ALFAM database was analysed by fitting a model to the ammonia volatilisation (loss rates). The ALFAM model used was based on the Michaelis-Menten-type equation presented by Sommer & Ersbøll (1994):

$$N(t) = N_{\max} \frac{t}{t + K_m} \qquad (1)$$

where $N(t)$ is the cumulative ammonia volatilisation at time t from the start of the experiment, expressed as a fraction of the total ammoniacal N (TAN) applied, N_{max} is the total loss of ammonia (fraction of TAN applied) as time approaches infinity, and the parameter K_m (h) is the time when $N(t) = ½ N_{max}$ (Figure 4.1). The loss rate (loss per time unit), which is also illustrated in Figure 4.1, is defined as the derivative of the function in Equation 1.

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

The initial loss rate (loss rate at $t = 0$) is N_{max}/K_m. This fact will be utilised when discussing the results later in this chapter.

Sommer & Ersbøll (1994) fitted this type of model directly to cumulative loss data. However, to minimise the serial correlation between successive measurement and thereby achieving a more reliable statistical analysis, it was more appropriate to model loss rates (loss per time interval) rather than cumulative losses. Another advantage of this approach is that missing loss rate observations during an experiment are permissible, since calculation of cumulative loss values will not be needed. However, to develop the ammonia loss rate model or the ALFAM model, it was necessary to reformulate Equation 1.

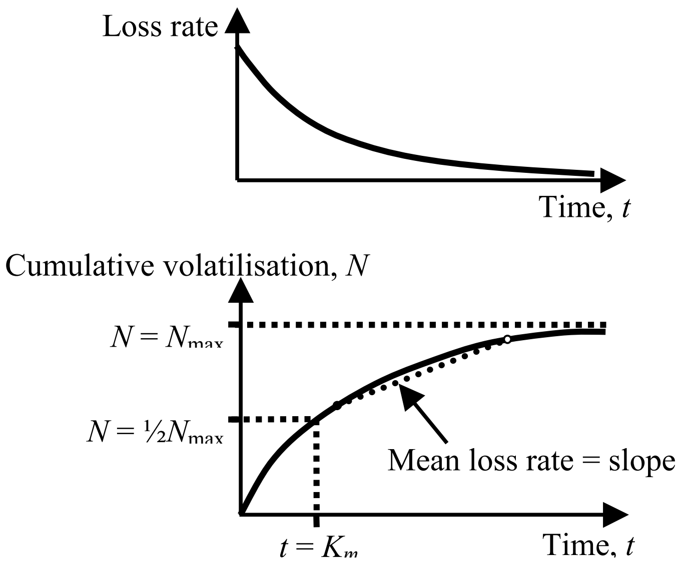


Figure 4.1 Schematic presentation of ammonia loss rate in relation to time following the field application of slurry (above) and the cumulative volatilisation loss from a slurry application (below). N_{max} and K_m are the parameters used in the Michaelis-Menten-type model of the rate of ammonia loss.

Loss rates will normally be recorded as mean rates over finite time periods. Assuming that the ammonia loss has been measured over the time period from t to $t + \Delta t$, the mean loss rate over this time period can be expressed as:

$$\overline{N}_{\text{rate}}(t,\Delta t) = \frac{N(t+\Delta t) - N(t)}{\Delta t} = \frac{N_{\max}\dfrac{t+\Delta t}{t+\Delta t+K_m} - N_{\max}\dfrac{t}{t+K_m}}{\Delta t}$$

or

$$\overline{N}_{\text{rate}}(t,\Delta t) = N_{\max}\frac{K_m}{(t+K_m)(t+\Delta t+K_m)} \qquad (3)$$

The model in Equation 3 was used for the analysis of ammonia loss rates as a function of time. Since the loss pattern over time will depend on climate, manure composition, soil conditions, application method, *etc.*, the parameters, N_{max} and K_m, were modelled as functions of such explanatory variables. Sommer & Ersbøll (1994) applied linear (additive) functions of the following type:

$$N_{max} = a'_0 + a'_1 x_1 + \cdots + a'_m x_m \quad (4)$$

$$K_m = b'_0 + b'_1 x_1 + \cdots + b'_m x_m \quad (5)$$

where a'_0, …, a'_m and b'_0, …, b'_m are model parameters to be estimated by statistical analysis, and x_1, …, x_m are the explanatory variables.

Since N_{max} and K_m should only take non-negative values, and the expressions in Equations 4 and 5 can take any real values, the following relationships may be more appropriate:

$$N_{max} = \exp(a_0 + a_1 x_1 + \cdots + a_m x_m) \quad (6)$$

$$K_m = \exp(b_0 + b_1 x_1 + \cdots + b_m x_m) \quad (7)$$

where a_0, …, a_m and b_0, …, b_m are model parameters to be estimated. By rewriting these expressions it can be seen that Equations 6 and 7 correspond to multiplicative relationships with the exponentials of the explanatory variables as factors:

$$N_{max} = A_0 \times A_1^{x_1} \times \cdots \times A_m^{x_m}\ , \quad \text{where } A_i = e^{a_i}, i = 0, \ldots, m \quad (8)$$

$$K_m = B_0 \times B_1^{x_1} \times \cdots \times B_m^{x_m}\ , \quad \text{where } B_i = e^{b_i},\ i = 0, \ldots, m \quad (9)$$

The explanatory variables selected for the model analysis are listed in Table 4.1 (m = 15). Most of the variables are so-called indicator variables, which can only take values of zero and one. These variables have been introduced in order to represent different states of categorical factors (*e.g.* application method).

Table 4.1 Explanatory variables used for modelling ammonia volatilisation loss rates

Experimental factor	Explanatory variable(s) Symbol	Range	Comment
Moisture content of soil	x_1	[0, 1]	$x_1 = 1$ if wet soil; $x_1 = 0$ if dry soil.
Air temperature	x_2	[–5.6, 36.0]	Unit: °C.
Wind speed	x_3	[0.0, 9.0]	Unit: m s^{-1}.
Manure type	x_4	[0, 1]	$x_4 = 1$ if pig slurry; $x_4 = 0$ if cattle slurry (only pig and cattle slurry have been included in the analysis).
Dry matter content of manure	x_5	[0.8, 11.0]	Unit: %.
TAN content of manure	x_6	[0.2, 4.0]	Unit: g N kg^{-1} .
Application method	x_7	[0, 1]	$x_7 = 1$ if band spread/trailing hose; $x_7 = 0$ otherwise.
	x_8	[0, 1]	$x_8 = 1$ if trailing shoe; $x_8 = 0$ otherwise.
	x_9	[0, 1]	$x_9 = 1$ if open slot injection; $x_9 = 0$ otherwise.
	x_{10}	[0, 1]	$x_{10} = 1$ if closed slot injection; $x_{10} = 0$ otherwise.
	x_{11}	[0, 1]	$x_{11} = 1$ if pressurised injection; $x_{11} = 0$ otherwise. (The application method "broad spreading" will be represented by the variable constellation $x_7 = x_8 = x_9 = x_{10} = x_{11} = 0$).
Application rate of manure	x_{12}	[9.6, 99.3]	Unit: t ha^{-1} or m^3 ha^{-1}.
Manure incorporation	x_{13}	[0, 1]	$x_{13} = 1$ if no incorporation; $x_{13} = 0$ if shallow cultivation.
Technique for ammonia loss measurement	x_{14}	[0, 1]	$x_{14} = 1$ if wind tunnel; $x_{14} = 0$ otherwise.
	x_{15}	[0, 1]	$x_{15} = 1$ if micro met; $x_{15} = 0$ otherwise. (The measuring technique "Lennart boxes (dynamic chambers)" will be represented by the variable constellation $x_{14} = x_{15} = 0$).

The set of explanatory variables that were selected for modelling reflects a compromise between two important considerations. Firstly, priority was given to variables which, on theoretical grounds, should have a major influence on the ammonia volatilisation. Secondly, it was an objective to utilise as many records from the ALFAM database as possible. However, in most of the records, only subsets of all the possible variables were recorded. It was, therefore, necessary to find a compromise between excluding variables from the model and excluding database records. For example, the manure pH is considered to have an important effect on ammonia volatilisation, but is not included. In general, the pH was not measured in a large number of experiments included in the ALFAM database, and consequently, inclusion of this factor in the model would reduce the number of useful records by about 45%.

The ammonia loss model in Equation 3 was analysed with both additive and multiplicative models for N_{max} and K_m (Equations 4 & 5 and 6 & 7, respectively). In both cases a power transformation was introduced in Equation 3 to ensure that the residuals will be approximately Gaussian distributed:

$$[N_{\text{rate}}(t,\Delta t)]^{\lambda} = \left[N_{\max} \frac{K_m}{(t+K_m)(t+\Delta t+K_m)} \right]^{\lambda} \qquad (10)$$

The value of the exponent, λ, was chosen to give the best approximation to a Gaussian distribution. Both sides of the equation were raised to the power of λ to ensure that the original interpretations of N_{max} and K_m were still valid.

The analysis was undertaken by using the non-linear regression procedure (proc nlin) in the SAS System for Windows, Release 8.00.

To restrict the range of conditions covered by the model, and to avoid extreme and unreliable observations, only records in the database satisfying certain criteria were used for the analysis. The following selection criteria were applied:

- Only data from application of cattle and pig slurry were used.
- Only data from experiments on bare soil, stubble or crop height less than 15 cm were used.
- The observed loss rate of ammonia was non-negative.
- The first ammonia loss rate in a measuring series was greater than the second one. If not, the first one was discarded.
- The application rate of manure did not exceed 100 t ha^{-1}.
- All the selected explanatory variables were measured.

These selection criteria left 2481 data records for the model analysis.

4.3 Results and Discussion

4.3.1 Manure composition and environment

The characteristics of the pig and cattle slurry from the ALFAM database used in the ALFAM model analysis are summarised in Table 4.2. As noted in Chapter 3, there was a high dry matter and total ammoniacal nitrogen content in the pig slurries used in the Dutch experiments. The ranges for both these variables in the remaining slurries were similar to that from most ammonia volatilisation experiments. The data also represent a wide variation of soil textures, pH, water contents, air temperatures, soil temperatures and wind velocities. Five different application methods were used to apply the animal slurry at different rates, and in one study the effect of incorporation into the soil was included. Furthermore, the ammonia volatilisation was measured by using three different techniques, *i.e.* wind tunnels, the micrometeorological mass balance technique and the JTI equilibrium concentration method (Ryden and Lockyer 1985; Ryden and McNeill, 1984; Svensson and Ferm, 1993).

Table 4.2 Averages of pH, dry matter (DM) percentage, total nitrogen (N) and total ammoniacal nitrogen (TAN) for the slurry data, from the ALFAM database, used in the model analysis. Standard deviations and numbers of observations are given in parenthesis as (standard deviation; number of observations)

Slurry type	pH	DM, %	N Total, g N/kg	TAN, g N/kg
Cattle slurry	7.34 (0.38; 162)	4.34 (1.75; 250)	2.30 (1.97; 230)	1.05 (0.50; 250)
Pig slurry	7.55 (0.35; 83)	4.04 (2.41; 115)	3.67 (1.32; 97)	2.54 (0.99; 115)

4.3.2 Fitting the model to volatilisation data

Loss rates are high immediately after slurry application (Figure 4.1). The high initial loss rate is related both to the initial high concentration of total ammoniacal nitrogen in the surface of the mixture of soil and slurry and to the rise in pH in the surface of newly applied slurry (Sommer & Sherlock, 1996). One to two days later, ammonia volatilisation rates are generally lower, because the dissolved total ammoniacal nitrogen in the soil surface will decrease rapidly due to volatilisation, infiltration and nitrification (Molen *et al.*, 1990). Emission peaks will occur during daytime, because of increases in temperature (Bless *et al.*, 1991; Brunke *et al.*, 1988). In most studies, cumulative ammonia volatilisation has reached 50% of its maximum within the first 4 to 12 hours after slurry application with splash plate (Pain *et al.*, 1989; Moal *et al.*, 1995).

The model with multiplicative submodels for N_{max} and K_m produced a somewhat better fit than the one with additive submodels (R^2 values of 0.80 and 0.77, respectively). These results together with the non-negativity constraints for N_{max} and K_m suggest that the multiplicative approach is more appropriate than the additive submodel. Only results from the former approach are presented here.

The estimates of the A and B parameters are given in Table 4.3. It should be noted that five parameters with very low statistical significance levels ($P > 0.4$) have been fixed at 1. This will minimise the variance when the model is used to predict ammonia losses for situations (values of the explanatory variables) not covered by the data set used for modelling.

N_{max} and K_m can be estimated by use of Equations (8) and (9) and from the parameter estimates in Table 4.3. However, one may also attribute a direct interpretation to each individual parameter value in Table 4.3. For example, the total ammonia volatilisation (N_{max}) from a wet soil has been estimated to be about 10.2% higher than from a dry soil as the multiplicative factor is $A_1 = 1.102$.

Table 4.3 Parameter estimates and confidence limits for the ALFAM model of ammonia loss with multiplicative submodels (*cf.* Equations 8 and 9)

Parameters related to N_{max} (see Eq. (8))

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

Parameters related to K_m (see Eq. (9))

Experimental factor	Interpretation of the corresponding parameter (as a multiplicative factor)	Parameter estimate	Approximate 95% confidence limits	
None	Common factor	$B_0 = 1.038$	0.606	1.776
Moisture content of soil	Wet soil (compared to dry soil)	$B_1 = 1.102$	0.967	1.256
Air temperature	Decrease per °C	$B_2 = 0.960$	0.951	0.969
Wind speed	Decrease per m s^{-1}	$B_3 = 0.950$	0.913	0.988
Manure type	Pig slurry (compared to cattle slurry)	$B_4 = 3.88$	3.18	4.74
Dry matter content of manure	Increase per % dry matter	$B_5 = 1.175$	1.134	1.218
TAN content of manure	Increase per g N kg^{-1}	$B_6 = 1.106$	1.004	1.219
Application method	Band spread/trailing hose	$B_7 = 1^{*)}$	-	-
	Trailing shoe	$B_8 = 1^{*)}$	-	-
	Open slot injection	$B_9 = 1^{*)}$	-	-
	Closed slot injection	$B_{10} = 1^{*)}$	-	-
	Pressurised injection	$B_{11} = 1^{*)}$	-	-
	(Compared to broad spreading)			
Application rate of manure	Increase per t ha^{-1} or m^3 ha^{-1}	$B_{12} = 1.0177$	1.0127	1.0227
Manure incorporation	No incorporation (*versus* shallow cult.)	$B_{13} = 1^{*)}$	-	-
Technique for ammonia loss measurement	Wind tunnel	$B_{14} = 1.48$	1.04	2.08
	Micromet (*versus* JTI Equilibrium concentration method)	$B_{15} = 2.02$	1.38	2.94

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

Care is required when interpreting the A_6 and A_{12} parameters. Both are less than 1, indicating that the total ammonia volatilisation, N_{max}, will decrease if the total ammoniacal nitrogen content of the slurry increases or if the manure application rate is increased. However, since the total ammonia loss is defined as a fraction of the total ammoniacal nitrogen applied, this will only be true in a relative sense. It can be proved that the actual amount of ammonia lost (g N ha^{-1}) will increase when the total ammoniacal nitrogen content or the manure application rate is increased.

4.3.3 Effect of slurry composition, application technology and environment on N_{max}

The ALFAM model predicts that the cumulative ammonia loss (N_{max}) increases with air temperature and wind speed (A_2 and $A_3 > 1$ in Table 4.3). This analysis confirms the results from studies showing that ammonia volatilisation during the initial 4 to 6 hours increases with increasing air temperatures or incident solar radiation (Brunke *et al.*, 1988; Moal *et al.*, 1995; Braschkat *et al.*, 1997; Sommer *et al.*, 1997). The increase with incident global radiation is due to the energy requirement for the endothermic volatilisation process to take place. The model will, in general, predict lower total ammonia volatilisation from slurry applied early in the morning than from slurry applied in the afternoon as a consequence of the dependency on air temperature.

Other experiments indicate that cumulative ammonia volatilisation after 7 days from slurry applied with splash plates to crops will be related to wind speed (*e.g.* Sommer *et al.*, 1997). The effect of wind speed has not been found in all studies (Beauchamp *et al.*, 1978; Bussink *et al.*, 1994), probably because the wind speed is generally high enough for the gas phase resistance to be negligible.

This study has shown that the ammonia volatilisation (ammonia volatilisation in real terms, not relative to the total ammoniacal nitrogen applied) will increase with increasing slurry dry matter and total ammoniacal nitrogen concentrations. Furthermore, the volatilisation will be lower from pig slurry than from cattle slurry ($A_4 < 1$ in Table 4.3). In the study of Bussink *et al.* (1994), it was shown that the ammonia volatilisation during the first 24 hours was related to dissolved ammonia, which was calculated from slurry total ammoniacal nitrogen, slurry pH and temperature. Thus, total ammoniacal nitrogen is an important factor when modelling volatilisation, as indeed is slurry pH following the application (Sommer & Olesen, 2000). The dry matter content has been shown to affect the ammonia volatilisation significantly, and field studies have shown that the ammonia volatilisation tends to be linearly or sigmoidally related to the dry matter content (Braschkat *et al.*, 1997; Moal *et al.*, 1995; Sommer & Olesen, 1991). Higher dry matter contents have shown to cause higher ammmonia volatilisation from cattle slurry than from pig slurry (Pain *et al.*, 1990).

The model output confirms the findings of a number of previous studies showing that the ammonia volatilisation will be relatively low, if slurry is applied on dry soil ($A_1 > 1$), even if the air or soil surface temperature is high (Sommer *et al.*, 1991). Previous studies have shown that ammonia losses will increase if the infiltration is reduced because of a high soil water content (Donovan and Logan, 1983). In a laboratory study, it was shown that the ammonia volatilisation from slurry applied on a dry soil (0.01 g H_2O g^{-1} of soil) was 70% of the volatilisation from slurry applied to soil at more than 0.8 g of H_2O g^{-1} of soil (Sommer & Jacobsen, 1999).

The model indicates that application of slurry by means of band spreading or injection methods will the reduce ammonia volatilisation, compared to splash plate or broad spreading (A_7, A_8, A_9, A_{10} and $A_{11} < 1$). In all cases, except from trailing shoe application, the reductions are significant ($P < 0.05$). The confidence intervals for trailing shoe application, closed slot injection and pressurised injection are relatively wide (upper limit between 3 and 6 times higher than lower limit, *cf.* Table 4.3). This is because the model is based on very few observations representing these application methods (less than 40 ammonia volatilisation measurements). The trailing shoe application and closed slot injection methods are well-represented in the original ALFAM database (more than 400 ammonia volatilisation measurements), but the major part of the data records could not be used for modelling, due to missing observations of soil moisture, air temperature or wind speed.

The analysis indicated that volatilisation measured with the wind tunnels and micrometeorological mass balance technique was much lower than volatilisation measured with the JTI technique (A_{14} and $A_{15} < 1$). At IGER in the UK, similar differences between measuring techniques were found. The JTI technique is a micrometeorological mass balance technique where a chamber is used to derive the equilibrium concentration, and thus, the chamber will provide shelter from rain during measurements, but it will not affect the wind and the temperature. The sheltering effect may have caused high losses from the applied slurry, compared to the estimates measured with the micrometeorological mass balance technique, which will reflect the weather during the measurement period. The tunnel will provide shelter from rain, but the adjusted wind speed during the experiments may have been low and the estimated volatilisation have, therefore, been lower than volatilisation measured with the JTI technique (*cf.* Chapter 5).

4.3.4 Effect of slurry composition and environment on K_m and the initial loss rate

A low value of K_m indicates that a higher proportion of the overall ammonia loss takes place quickly after slurry application, and a high K_m value indicates the reverse. The initial loss rate (at time t = 0) can be computed as N_{max}/K_m (*cf.* Section 4.2.1). This means that if an experimental factor changes N_{max} by a multiplicative factor A and changes K_m by a multiplicative factor B, then the initial loss rate will be changed by the multiplicative factor A/B. Thus, the ratio between corresponding A and B parameters in Table 4.3 can be used to assess how a given experimental factor will affect the initial ammonia loss. It appears that the initial loss rate will increase with increasing air temperature and wind speed ($A_2/B_2 = 1.06 > 1$ and $A_3/B_3 = 1.10 > 1$) (Table 4.3).

Despite the fact that a high dry matter content will result in a low initial loss rate ($A_5/B_5 = 0.94 < 1$), this loss rate will only decline very slowly, because of the high value of K_m ($B_5 > 1$). The high volatilisation rates in the days after slurry application may be due to TAN being retained in the slurry dry matter on the soil surface, rather than infiltrating into the soil.

The predicted loss rate will be lower for pig slurry than for cattle slurry, both initially ($A_4/B_4 = 0.22 < 1$) and in the long run ($A_4 < 1$). This may be attributed to a lower viscosity in the pig slurry, so that TAN infiltrates more easily into the soil.

4.3.5 Prediction of ammonia volatilisation for an independent data set

The ability of the model to predict ammonia losses was tested by using an independent data set produced in a recent experiment, reported by Hansen *et al.* (2001). In this study, slurry was applied by trailhoses or injected (open slot) to a grass field (*ca.* 10 cm high). The slurry slot volume ratio was measured to determine the injection efficiency. A ratio greater than 1 indicates that the slot has been overfilled with slurry. Ammonia volatilisation was measured for 7 to 10 days after application. The relation between predicted and measured ammonia volatilisation is shown in Figure 4.2. The vertical distances between the symbols and the 1:1 line indicate prediction errors. The model overestimates ammonia volatilisation from trailing hose applied slurry, as it can not account for crop height. A 10 cm grass height may reduce volatilisation (Sommer *et al.*, 1997; Chapter 6). This observation suggests that the model could be improved by including an experimental factor for crop cover (grass, stubble, bare soil or growing crop).

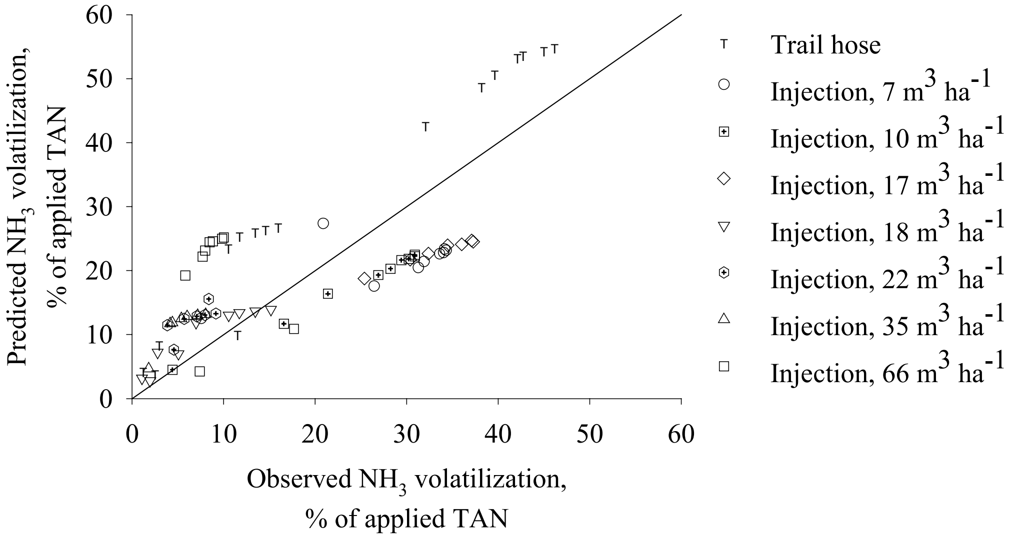


Figure 4.2 The relation between the statistical model prediction of ammonia volatilisation compared with the measured (Hansen *et al,*. 2001) ammonia volatilisation from slurry applied to a grass field.

The model overestimates the ammonia volatilisation from slurry injected at high rates and underestimates it from slurry injected at low rates. This is because the model does not differentiate for the slurry slot volume ratio. Hansen *et al.*, (2001) have shown that ammonia loss is significantly related to slurry slot volume. Therefore, injection efficiency should be included as an experimental factor in volatilisation models. Measurement of injection efficiency should be included in future studies of ammonia volatilisation where this spreading technique is being evaluated.

4.3.6 Prediction of ammonia volatilisation for selected cases

The climatic conditions, the manure composition and the spreading method application practice will vary considerably throughout Europe. It was therefore decided to predict the effect of air temperature, wind speed and soil water content on volatilisation from surface applied cattle and pig slurry surfaces at three different times of the year. The following typical application times were selected:

1. One week before normal sowing time for spring crops
2. At the time of a mid-season grass cut (if relevant)
3. One week after harvesting of the spring-sown crop.

Seven days of observed weather and soil water content from the year 2000, corresponding to the three application times for Italy (Po Valley), Southern England (North Wyke region), Southern Norway (Follo area) and Denmark (Western Jutland), were used in this modelling exercise. The predictions are examples of the most important European climate and soil conditions (Table 3.2) and most widely used slurry application technique. The compositions of the slurries used for the calculations were the mean values for pig and cattle slurry given in Table 4.2. Slurry was applied at a rate equivalent to 100 kg TAN ha^{-1} by use of broadcast spreading in the three periods, resulting in application rates of 95.2 t ha^{-1} for cattle slurry and 39.4 t ha^{-1} for pig slurry. The predictions were calculated on the basis of the micrometeorological mass balance technique.

The model prediction of the cumulative ammonia loss in average of seven applications are shown in Figure 4.3. It was assumed the initial application took place at 07.00 hours on day 1. The results assume no slurry incorporation.

The model predicted that there would not be large differences in ammonia volatilisation between the four countries (Figure 4.3). The ammonia losses were higher from cattle slurry than from pig slurry. The volatilisation patterns throughout the periods are similar for the two slurry types. The expected increase in volatilisation between periods 1 to 2 and 3, as a consequence of the increase in air temperature (Table 4.5), did not occur, because changes in soil moisture and wind speed confounded the temperature response in all countries. Wet soil and high wind speeds led to higher losses in period 1 in the UK and Denmark. The wind speeds are generally low in Italy and the variations in ammonia losses are primarily caused by variations in air temperature. Higher ammonia loss as a consequence of higher temperatures in Italy would be expected. However, volatilisation in Italy (Po Valley) was not higher than in cooler parts of Northern Europe, due to the lower wind speeds here (Table 4.4). This meant that in these cases, wind speed and soil moisture rather than temperature are major determinants in ammonia volatilisation from field applied manure.

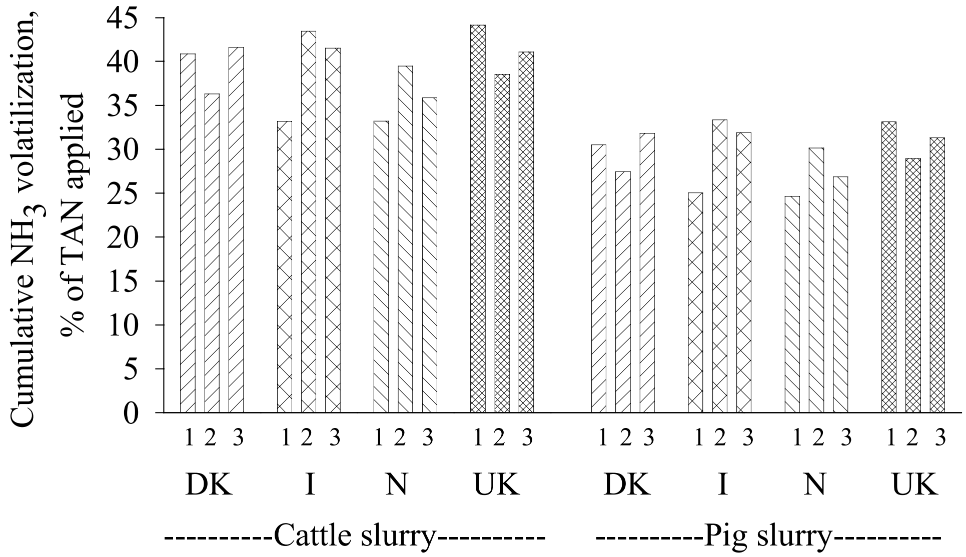


Figure 4.3. Predicted cumulative ammonia volatilisation from a surface application of a standard cattle and pig slurry applied during three periods (1. before sowing barley, 2. mid summer, and 3. after harvest) in West Jutland in Denmark (DK), Po Valley of Italy (I), South Norway (N) and South England (UK).

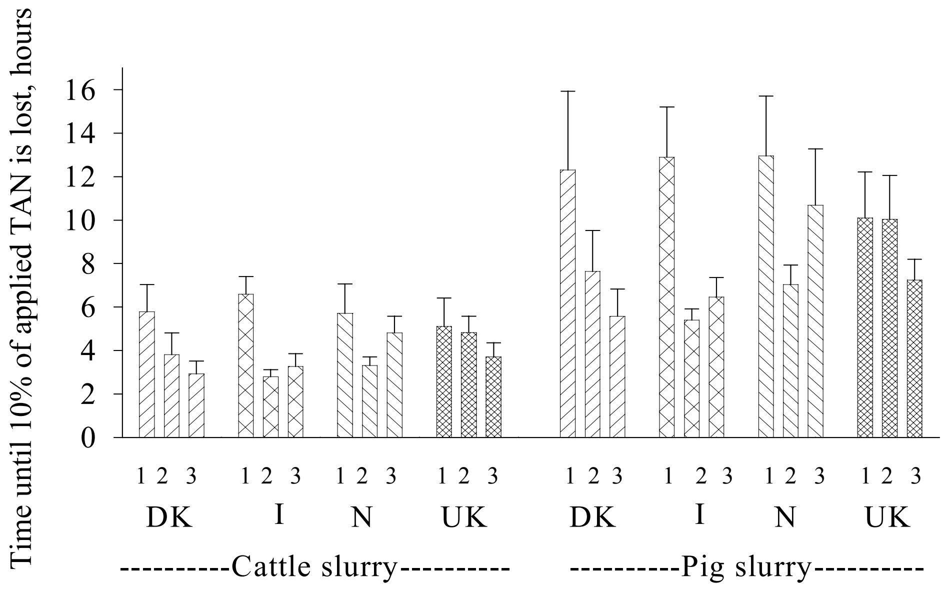


Figure 4.4 Time elapsed until 10% of the total ammoniacal nitrogen in applied slurry is lost. The results are based on simulation of slurry application during three periods (1. before sowing barley, 2. mid summer, and 3. after harvest) in Western Jutland in Denmark (DK), Po Valley of Italy (I), Southern Norway (N) and Southern England (UK).

It is interesting, both from a farmer's and a policy maker's perspective, to determine how soon after application slurry would have to be incorporated to limit the losses to a given proportion of the TAN applied. Furthermore, it is interesting to determine to what extent this time interval varies between the regions. The model was used to estimate the length of the period needed if losses of TAN from the two example slurries were to be restricted to 10% of the

Table 4.4. Weather and soil moisture data used for prediction of ammonia volatilisation for the three periods (of the year) in each of the four countries. Section A refers to the average for the first 24 hours of each period and Section B refers to the average for each seven-day period

A. First day (24 h) of each period

	Period 1			Period 2			Period 3		
	Average air temp. (°C)	Average wind speed (m s^{-1})	Soil moisture	Average air temp. (°C)	Average wind speed (m s^{-1})	Soil moisture	Average air temp. (°C)	Average wind speed (m s^{-1})	Soil moisture
Denmark	6.0	5.8	Wet	11.1	1.8	Dry	13.4	3.0	Dry
Italy	6.5	0.9	Wet	19.1	0.9	Dry	18.2	1.0	Dry
Norway	4.6	1.1	Wet	12.0	3.1	Dry	8.0	0.6	Wet
England	10.4	5.3	Wet	10.1	3.5	Dry	14.1	3.6	Dry

B. Statistics for each period (week)

	Period 1			Period 2			Period 3		
	Average air temp. (°C)	Average wind speed (m s^{-1})	Soil moisture	Average air temp. (°C)	Average wind speed (m s^{-1})	Soil moisture	Average air temp. (°C)	Average wind speed (m s^{-1})	Soil moisture
Denmark	5.0	4.1	Wet	11.1	3.7	Dry	14.1	4.4	Dry
Italy	8.1	1.3	Wet	20.7	0.8	Dry	18.6	1.0	Wet/dry
Norway	4.5	2.1	Wet	13.0	2.5	Wet/dry	8.6	0.7	Wet
England	8.5	3.5	Wet	9.5	2.7	Dry	14.4	2.7	Dry

TAN applied. It was assumed that the application was made at 07.00 hours at the start of the seven-day period for each of the three application times. The results are shown in Figure 4.4. The time interval was calculated seven times for each combination of slurry type (cattle/pig), country and period. Therefore, each bar in Figure 4.4 represents an average of seven values. The standard deviations are shown as error bars. The weather and the soil moisture status during the first 24 hours after slurry application are the main determinants of the ammonia loss in Figure 4.4. The average air temperature, the average wind speed and the moisture state of the soil for the first 24 hours and the corresponding data for the entire seven-day period are shown in Table 4.4. The data in Table 4.4 is relevant to the results in Figure 4.4, since the results are averages corresponding to slurry application each day in the periods

The results show that soil moisture has little influence on the time for 10% of the applied total ammoniacal nitrogen to be volatilised, so differences in wind speed and temperature can be used to explain the predicted differences between countries (Figure 4.4). The mean wind speed varies little between periods within a country (Table 4.4). Therefore, the wind speed is unlikely to explain the differences within a country in the time elapsed for the loss of 10% of the applied total ammoniacal nitrogen shown in Figure 4.4. On the other hand, the mean air temperatures for different periods within countries varied considerably with lower temperatures for period 1 compared with periods 2 and 3. This difference can explain the longer time interval from spreading for the loss of 10% of the applied total ammoniacal nitrogen in period 1, compared with the other two periods (Figure 4.4). It is worth noting that the relatively higher temperature in Italy does not result in a shorter time for the loss of 10% of the applied total ammoniacal nitrogen because of lower wind speeds, when compared with the other countries.

Local experiments should always be made, as there may be additional local specific factors that have great importance and which are not included in the model. Also one has to be careful not to over-interpret the case studies – they are for a few sites in just one year!

4.3.7 The model – a wider perspective

The analysis has shown that the results from different European countries are generally comparable. This suggests that the results from the ALFAM database can be used throughout countries, thus reducing the requirements for further extensive experimentation. Furthermore, the ALFAM model can be used to calculate rough estimates of the ammonia volatilisation under combinations of conditions, which have not been examined through experiments. Countries that have not yet conducted ammonia emission experiments from field-applied manure can perform searches of the ALFAM database to check if the national circumstances for slurry application are present. Additional experiments will only be necessary where this is not the case.

The ALFAM model can predict total ammonia volatilisation for a given set of weather, soil and some management conditions. This information together with knowledge on the quantities of ammoniacal and organic nitrogen applied will enable calculation of the fertiliser nitrogen value of the slurry for crop production. The potential to use the outputs of the model will provide an important input to any farmer or nutrient adviser. To make this information accessible to farmers requires the development of a decision support model (*rf.* Chapter 2).

4.4 Conclusion

The ALFAM model development and testing has shown that data from European experiments can be collated in one database and used to create a model with meaningful outputs. The outputs from the data modelling are supported by theoretical considerations of effects of weather, soil characteristics and manure composition impacts on ammonia volatilisation from field applied animal slurries to low crops or fallow land. A Michaelis-Menten-type exponential equation provides a good fit for the measured ammonia loss rates. The analysis indicates that the loss rates rather than cumulative ammonia losses should be used in modelling exercises. The use of a multiplicative submodel rather than an additive submodel in the model improved its overall predictive accuracy. The model was used to calculate the ammonia losses from standard slurry applied in four different European countries. Unexpectedly, total volatilisation was similar in Italy (Po Valley) and North European countries, because the effect of differences in temperature and wind speed between the two locations counteracted one another. Likewise, the maximum time between surface application with a splash-plate and incorporation that is necessary to achieve a specified emissions target was similar between countries.

5. Ammonia losses from field applied manure – A Review of the Measurement Techniques

5.1 Introduction

Ammonia is a highly reactive gas that is very soluble in water, and which readily combines with a proton to form an ammonium ion. It is normally present in air at trace concentrations, only. Ammonia is therefore difficult to measure, especially from diffuse sources such as those arising from the field application of manure. Although fluxes per unit area may be small, the need to multiply by large areas can result in large total emissions, with a corresponding potential for large errors. It may also be necessary to distinguish between gaseous and particulate forms. Ammonia will be adsorbed onto materials (*e.g.* stainless steel, some plastics) commonly used in the construction of tubing, *etc.,* for sampling or trapping gases as well as onto wet surfaces. Care must be taken in the selection of these materials, with glass and Teflon being most suitable, and provision must be made to ensure that they are kept free of condensation.

Despite these difficulties, considerable advances have been made over the past 20 years in the measurement of ammonia emissions from land, and new, improved techniques continue to be developed. Initially, measurements of emissions from mineral nitrogen fertilisers were based on a soil balance without direct measurement of ammonia losses. Ammonia and ammonium readily react with acid. Early direct measurements employed dishes of acid placed near the ground and equated the rates at which the dishes absorbed ammonia and ammonium with fluxes from the atmosphere. In practice, this gave a measure of concentration rather than of flux. At this time, interest was more concerned with deposition than with emission of ammonia, until the importance of animal sources on emission was highlighted by, for example, Healy *et al.* (1970). Later, it was established that emission from spreading manures on to land was a very significant source, often accounting for 30-40% of the total national emission (ECETOC, 1994). Consequently, a considerable number of measurements have been made of ammonia emissions arising from spreading manures in different countries under wide-ranging conditions. Data resulting from such measurements are needed for the construction of national emission inventories, to assess the impact on the environment and to develop and evaluate methods for abatement.

This chapter reviews the methods available for measuring ammonia emissions from land, to identify those that are most suitable for particular purposes and to produce detailed protocols for a series of "standard" methods. The latter are based on the results of a questionnaire sent to all members of this concerted action project and other interested parties, requesting information on the most commonly used methods and the procedures followed. The chapter is subdivided into three main sections: i) a review of methods and their applicability; ii) a review of sensors for measuring ammonia concentration in air or ammonia flux; iii) detailed descriptions of the most commonly used methods.

5.2 Review of methods used for measuring ammonia emissions from field applied manure

Early methods were based on constructing a soil balance because no reliable techniques for direct measurement of ammonia were available. Most, but not all, methods used in recent years fall into two main types of micrometeorological and enclosure methods. In general, both types require a sensing device for sampling and measuring the ammonia concentration in air and a system for measuring and recording air flow or wind speed, so that a flux can be calculated. Micrometeorological methods are usually most suitable for use in the measurement of integrated fluxes or emissions from relatively large areas of land under undisturbed conditions. Enclosure methods are commonly used on small plots and are useful for comparative studies, for example, in investigating factors that control rates of emission.

5.2.1 Soil balance or N recovery method

This involves sampling soil after application of fertiliser or manure to measure changes in soil nitrogen (*e.g.* Denmead *et al.*, 1977). It is an unsatisfactory method, because the NH_3 losses are very small in relation to the size of the soil nitrogen pool, so the resolution of measurements is insufficient to estimate losses reliably. Increased precision may be obtained by the use of ^{15}N labelled manure (*e.g.* Moal *et al.*, 1994; Morvan *et al.*, 1997).

5.2.2 Micrometeorological methods

In these methods, the horizontal NH_3 transported to and from an experimental area is measured, and then the flux is calculated as emission or deposition from integration of the difference in the flux to and from the plot. They have the advantages of being non-intrusive and of integrating the heterogeneity of the experimental areas. These techniques include the following:

- Mass balance (Integrated horizontal flux, Single profile method or theoretical profile shape, Theoretical profile shape – Philip's solution and Perimeter profile method)
- Eddy correlation and relaxed eddy accumulation
- Gradient methods (Aerodynamic approach and Bowen ratio-Energy balance)
- Backward Lagrangian stochastic model
- Equilibrium concentration technique (JTI method).

5.2.2.1 Mass balance

The mass balance method equates the vertical flux from the edge of the field to the measurement mast to the horizontal flux through a vertical plane located at the measurement mast (Denmead *et al.*, 1977). The horizontal flux is normally derived as the product of wind speed and net NH_3 concentration (i.e. leeward NH_3 concentration minus NH_3 concentrations upwind of the plot) divided by the fetch, i.e. the distance between upwind and downwind edges. They include the following:

Integrated horizontal flux (IHF): This is the commonly used method for measuring NH_3 emissions from spread manures and involves the measurement of ammonia flux at the upwind and downwind edges of an area of land. The method, discussed by Denmead (1983), assumes that the vertically integrated product of wind speed and NH_3 concentration, divided by the fetch, is equal to the NH_3 flux F from the surface. Thus, in theory

$$F = \frac{1}{x} \int_{z_0}^{z_p} \overline{u\chi_l} - \overline{u\chi_w} . dz \tag{1}$$

where u is the wind speed and χ_l and χ_w the downwind and upwind NH_3-concentrations, respectively, and x is the fetch. The integration limit z_p is the height at which the NH_3-concentration is at background level, and z_0 is the height at which the wind speed falls to zero.

In practice, a mass balance is determined from the difference in the amount of NH_3 driven across the upwind and downwind edges of the experimental area, estimated from a vertical profile that is normally obtained from measurements at 5-6 heights.

For practical use, the following equation is often used:

$$F = \frac{1}{x} \int_{Z1}^{Zn} \overline{u}(\overline{X}_l - \overline{X}_w) dz \tag{2}$$

where $\overline{u}$ is the mean wind speed and $\overline{\chi}_l$ and X_w are the mean downwind and upwind NH_3 concentrations, respectively, over a particular measurement period. This simplification leads to the neglecting of the turbulent terms of the flux, and thus, the flux is overestimated by about 10-15% (Raupach and Legg, 1984; Leuning *et al.*, 1985; Wilson and Shum, 1992). This can be overcome by the use of integrated samplers.

Certain conditions are required for the methods viz a flat, homogeneous area surrounding the experimental area and low concentrations of ammonia in the background air. For square, rectangular or irregularly shaped experimental areas, the fetch will vary with the wind direction. It is, therefore, necessary to record the wind direction and calculate the fetch for each period of measurement. This can be avoided by using circular experimental areas (*e.g.* Pain *et al.*, 1989) with a mast supporting an array of sensors mounted in the centre of the circle.

Single profile method or theoretical profile shape (TPS): The emission can be inferred from measurements of ammonia concentration in air and wind speed at a single height above the ground termed Z_{inst}, providing certain conditions are met relating to the size and uniformity of the land surrounding the experimental area (Wilson *et al.*, 1983). The flux *F* is calculated from:

$$F = \frac{\bar{u}.\bar{\chi}}{\left(\overline{u\chi} / F \right)} \tag{3}$$

Where $\bar{u}$ and $\bar{\chi}$ are mean wind speed and ammonia concentration, respectively, measured at height Z_{inst}. The term $\overline{u\chi}/F$ is a dimensionless ratio, values for which are given in Wilson *et al.* (1982). The height for Z_{inst} is normally between 0.9 and 1.5 m. The method offers savings in labour and equipment, with results comparable to those obtained to the IHF method outlined above.

Theoretical profile shape – Philip's solution (PTPS): A power law curve is fitted to wind speed and transfer coefficient profiles for this method (Philip, 1959; McInnes *et al.*, 1985; Sommer *et al.*, 1995). The resulting multiplier and power coefficients are used together with the measure of atmospheric stability to solve for a gas concentration with unit flux. This theoretical concentration with a gradient concentration measurement is used to determine the gas flux from the surface. The method can be used when wind speed measurements can be made at several or at only one height above the surface. For the latter, there is a method for estimating the wind speed profile, but extra measurements of temperature are required.

Perimeter profile method: This method employs four masts placed perpendicular to each other around the boundary of an experimental area (Schjørring *et al.*, 1992). Arrays of flux samplers (Ferm, 1991) are mounted in pairs on masts around the boundary of a circular experimental area with one orifice pointing towards the test area and one towards the background. The horizontal flux of the inward and outward pointing tubes are determined separately for each of several heights on each mast. The vertical flux of ammonia is then determined by stepwise summation of the difference between the inward and outward facing horizontal fluxes. Although relatively simple to use in the field, a considerable labour input is needed for sampler preparation and analysis in the laboratory. Problems with condensation in the samplers under wet conditions may also occur.

The following micrometeorological methods are used to measure the vertical flux from fields. These methods do not affect the microclimatic conditions, which influence volatilisation. The problem is that fields of one to several hectares are needed for the measurements. Thus, comparative studies would require several large fields that are very similar, and this condition may be difficult to fulfil in practice.

5.2.2.2 Eddy correlation and relaxed eddy accumulation

In this method, flux is calculated by averaging the product of fluctuations of gas concentration and of the vertical wind speed (Lenschow, 1995). It requires instantaneous measurements of ammonia concentration in air, limiting the application of this method to those gases for which appropriate sensors are currently available (*e.g.* methane and carbon dioxide). The relaxed eddy accumulation method (Businger and Oncley, 1990) eliminates the need for high response sensors. Net vertical flux (F) is given as the product of the difference in average concentration between the upward (χ+) and downward moving (χ-) eddies and the standard deviation in vertical wind speed (σ_w) and an empirical constant (b):

$$F = b\sigma_w\left(\chi^+ - \chi^-\right) \tag{4}$$

However, it is still necessary to separate the up and down draughts in the sampling system, requiring valve and switching equipment sufficiently rapid to fully separate and sample from each of them. Fowler *et al.* (1995) verified the relaxed eddy accumulation method against eddy correlation for methane fluxes over peat wetlands. The method, as yet, has not been evaluated for ammonia.

5.2.2.3 Aerodynamic method or gradient diffusion method

With the aerodynamic method, the vertical turbulent diffusion coefficient depends on the wind speed and air temperature gradient (Denmead, 1983; Itier, 1981). It takes into account the effect of the stratification of the low atmospheric layers by using stability corrections derived from the Monin-Obukhov theory (1954).

$$F = K_z \, {}^{d\chi}\!/_{dz} \tag{5}$$

The method requires measurement of NH_3 concentration, wind speed and temperature at several heights. It has been used to measure NH_3 fluxes from natural vegetation (*e.g.* Harper *et al.*, 1983; Sutton *et al.*, 1992), but also from manures (Genermont *et al.*, 1998) and in combination with FTIR (Griffith & Galle, 2000). This method has been compared to the ^{15}N recovery method: both methods gave similar results for ammonia volatilisation from slurry applied on bare soil (Génermont *et al.*, 1996).

5.2.2.4 Bowen ratio – Energy balance

This method requires measurement of the gradient concentration of temperature, water vapour and ammonia, together with fluxes of net radiation and soil heat (Monteith, 1973). It is difficult to use. Application to ammonia may be feasible if concentration gradients can be determined over a few hours. Errors, however, may be large where sensible heat flux is small, especially during cloudy weather and in the night-time.

5.2.2.5 Backward Lagrangian stochastic model

The model is used to determine the backward trajectories of a gas, starting at the sensor and moving to the surface source (Flesch *et al.*, 1995). The flux is determined from

$$F = \frac{u\chi}{\eta} \tag{6}$$

where η is a constant calculated from the number of trajectories touching down on the treated surface. The model requires prior knowledge of the sensor height, roughness length and atmospheric stability, defined by the Monin-Obukhov stability length. Preliminary studies indicate that this technique is not as sensitive to emissions from manure-treated soils as the integrated horizontal flux method (McGinn and Pradhan, 1997).

5.2.2.6 Equilibrium concentration technique (JTI method)

This method was developed by the Swedish Institute of Agricultural Engineering (JTI) (Svensson, 1994). It is a micrometeorological method suitable for measuring ammonia emissions from small plots. It involves sampling close to the soil surface to measure the driving force for volatilisation and the aerodynamic resistance to flux. The latter is obtained from the ammonia concentration difference between the soil/manure surface and the air at a specific height above the surface. To calculate the emission rate, the equilibrium ammonia concentration (χ_{eq}) is determined by using a ventilated chamber together with the concentration in the ambient air ($\chi_{a,z}$) at height z. In addition, measurement is required of the mass transfer velocity for the distance from the soil surface to the height at which ambient air concentration is measured ($K_{z,a}$). The flux is calculated from the following relationship, derived from the law of aerodynamic resistance:

$$F = (\chi_{eq} - \chi_{a,z})K_{z,a} \tag{7}$$

The method has recently been verified against the IHF method for applications of urea fertiliser and manure to large plots (Misselbrook & Hansen, 2001).

5.2.3 Enclosure methods – chamber methods

These methods are useful when measurements of emission are required over defined areas (*e.g.* small plots) or spatial scales below the resolution are possible with micrometeorological methods. The function and use of enclosure methods for measurement of trace gases has been discussed by Livingston and Hutchinson (1995). Enclosures function by restricting the volume of air available for exchange across the covered surface, so that emission can be measured as a change in concentration. It is important that the enclosure does not significantly affect the production or absorption of the gas or the transport processes that control the flux, *e.g.*

temperature. They must be manufactured from materials that do not act as a sink, or the source of ammonia must be kept free of condensation.

There are two main types of enclosure viz closed (or non-steady-state or static) chambers and open (or steady state or dynamic) chambers.

5.2.3.1 Closed enclosure methods – static chambers

The flux is calculated from the speed of the increase of the gas concentration in the enclosure just after the system has been closed. However, the gas concentration gradient from the emitting surface to the air beneath the enclosure decreases as the concentration in the air increases. The design of the enclosure and the measurement period must be carefully selected to minimise negative feedback on the rate of diffusion of the gas. These restrictions pose practical difficulties for measuring ammonia emission from manure, because the rate of release of the gas is often very high immediately after application to land.

5.2.3.2 Open enclosure methods – dynamic chambers

In steady-state methods, the gas concentration gradient is assumed to be constant after an initial period following deployment. Steady state is maintained by using a constant flow of external air to sweep the enclosed volume, so that the gas concentration gradient can be regulated.

The wind tunnel system described by Lockyer (1984) is an example of a steady-state method representing the method most commonly used for measuring ammonia emissions from manures. These employ a fan to draw air over a 1 m^2 area treated with manure and samplers to measure inlet and outlet ammonia concentrations. Emission (E) from the area is calculated from:

$$E = (\chi_o - \chi_i)u \qquad (8)$$

where χ_o and χ_i are the ammonia concentrations in the outlet and inlet air, respectively, and u is the volume of air flowing through the tunnel over the sampling period. Alternative designs employ a filter to remove ammonia from the inlet air, which increases the costs, but avoids the need to measure χ_i.

Evaluations of wind tunnel systems (Sommer *et al.*, 1991; van der Weerden *et al.*, 1996) indicated that NH_3 recovery averaged 74 or between 86 and 90%, respectively. It was suggested that the ammonia trapping efficiency of wind tunnel systems should be checked on a regular basis to avoid errors.

5.2.4 Controlled gas release ratio methods

Vandre and Kaupenjohann (1998) describe a method, which they termed the standard comparison method, whereby the transfer factor of ammonia from source to a passive sampler on experimental plots is determined by means of releasing ammonia at a known rate *via* a cylinder and tubing on standard comparison plots. The transfer factor is then applied to passive sampler measurements of concentration from manure-treated plots to determine ammonia release rate (*i.e.* flux) from treated plots. A prerequisite of the method is that all factors influencing the transfer factor (micrometeorological conditions) are equal across all plots.

Galle *et al.* (2000) included tracer gas (SF_6) release in the gradient diffusion method to negate the requirement to estimate K_z (Equation 5). In this way, the flux of ammonia (F_{NH3}) can be derived from knowledge of the tracer gas flux (F_t) and the ratio of the measured concentration gradients of ammonia and the tracer gas.

$$F_{NH3} = F_t \frac{d\chi_{NH3}}{d\chi_t} \tag{9}$$

5.2.5 Modelling

5.2.5.1 Dispersion models

Amon *et al.* (1997) and Schafer *et al.* (1998) described methods for using open-path FTIR in combination with dispersion modelling for measuring ammonia emissions following manure application. Measurement of temperature, wind speed and direction and solar radiation, which should not change significantly during the measurement period, are required to determine air turbulence class and dispersion coefficients for the Gaussian dispersion model used. The accuracy of this method is not high, so it is more suitable for making comparisons (*e.g.* from different methods of application) than for measuring absolute emission rates.

5.2.6 Measuring ammonia emissions during field application

5.2.6.1 By difference

Ammonia losses during field application of manure have been calculated previously by comparing the ammoniacal nitrogen content of slurry before and after spreading (Genermont *et al.*, 1998; Safley *et al.*, 1992; Sharpe and Harper, 1997). Samples following application are collected in trays placed over the spreading area. A small volume of acid in the trays prior to sampling minimises the ammonia volatilisation from collected liquid. This technique is not very reliable, because losses are small, compared to the total ammoniacal content of the applied manure.

5.2.6.2 Micrometeorological measurement

A micrometeorological technique for measuring ammonia emission during slurry application was developed by Pain *et al.* (1989). A frame, constructed of tubular metal, with a cross sectional area of 40 m^2 was mounted on the front of a 4-wheel drive vehicle. The frame supported ammonia absorption flasks at 16 points. During spreading, the vehicle followed behind the spreading machine in the wake of the slurry plume at the minimum distance necessary to ensure that the absorption flasks were not contaminated by droplets of slurry. Pumping and air-flow control equipment used to draw air through the absorption flasks at 5 l min^{-1} were housed at the back of the vehicle. By always spreading exactly upwind, the volume of air flowing through the sampling frame could be calculated from ((speed of travel + ambient wind speed) × spreading time × cross-sectional area of frame). The product of air volume and mean ammonia concentration gave the total amount of ammonia lost during spreading.

5.2.7 Future methodological developments

There is still much activity in methodological development, particularly in the development of a non-intrusive method applicable to small plot measurements. Denmead *et al.* (1998) describe a mass-balance method which they used on a 24 × 24 m square plot for measuring carbon dioxide, methane and nitrous oxide fluxes, but the method could equally well be used for ammonia if used with a detector of sufficient sensitivity. Concentration measurements were made at heights of up to 3.5 m along each of the four boundaries. Gas concentrations were multiplied by the appropriate vector winds to yield horizontal fluxes at each height on the boundary. The difference between these fluxes integrated over downwind and upwind boundaries represents the net emission. The method was reported to be unreliable in conditions of light wind or variable wind direction. However, it represents a non-intrusive method, unaffected by atmospheric stability and appropriate for use in situations where conventional micrometeorological techniques cannot be used (*e.g.* small plots, elevated point sources, heterogeneous surface sources). The method was further developed by Magliulo *et al.* (2000), who used it on 4 × 4 m plots.

5.3 Sensors for measuring ammonia concentrations in air or ammonia flux

Sensors can be broadly divided into those measuring concentration and those measuring flux.

5.3.1 Ammonia concentration sensors

Samplers for measuring ammonia concentration can be divided into those measuring real-time concentrations in the gaseous phase and those giving time-averaged concentrations, involving adsorption of ammonia to a specific substrate (solid or aqueous).

5.3.1.1 Gaseous phase measurement

The principle of ammonia gas analysis is absorption or emission of a specific wavelength by ammonia or a derivative of ammonia. Maximum absorption of ammonia is at wavelength in the range 200-300 nm (infrared). Non-destructive instruments that have been used to measure ammonia concentrations are: FTIR, differential optical absorption spectroscopy (DOAS), tuneable diode laser (TDL), optical derivative spectroscopy, photofragmentation, laser photothermal techniques and photoacoustic sensors (Galle *et al.*, 2000; Heise *et al.*, 2001; Mennen *et al.*, 1996; Sommer *et al.*, 1995; Vogt *et al.*, 1999; Warland *et al.*, 2001). A chemoluminescence method has also been developed, where NH_3 is converted to NO_x, and the NH_3 is determined as the difference in luminiscence before and after NH_3 oxidation (Mennen *et al.*, 1996). The advantages of such instrumentation are that analyses are made in-situ, continuously and are automated. There can also be great advantages to having real-time measurements. Disadvantages include high cost, lack of sensitivity and problems with adsorption of NH_3 on tubing or in measurement or reaction chambers.

5.3.1.2 Adsorption techniques

Adsorption techniques rely on contact between sampled air and a substrate that has a great affinity for ammonia (usually an acid, as ammonia is a weak alkaline species). One of the most commonly used adsorption techniques is the adsorption flask (or bubbler) containing acid solution (typically orthophosphoric, sulphuric or boric) through which sampled air is drawn at a known flow rate (*e.g.* Lockyer, 1984). Other adsorption techniques include denuder tubes and badges. Denuder tubes consist of glass tubes that are coated internally with an acid (*e.g.* oxalic acid) through which air is drawn at a known rate (Ferm, 1979). Badges consist of cellulose filters impregnated with citric or oxalic acid (Svensson, 1994) that are exposed directly to the atmosphere. Following exposure, denuders and badges are washed, so that adsorbed ammonium can be measured in aqueous solution. A continuous flow denuder for measuring ammonia concentrations has been developed and is used routinely in the Dutch monitoring programme (Mennen *et al.* 1996).

Advantages of adsorption samplers are that they are relatively inexpensive and adaptable to a large range in atmospheric ammonia concentration. Disadvantages include that they are not automated, they may be labour intensive and give mean concentrations for a measurement period, and concentrations are only known some time after the event. There is also the possibility of interference of other nitrogen containing species (*e.g.* volatile amines) that may be adsorbed, although such interference is generally accepted to be insignificant when measuring high ammonia concentrations (such as following a field application of manure).

5.3.2 Ammonia flux samplers

Integrated samplers that sample ammonia proportional to wind speed and give a direct measure of flux rather than concentration, have been developed by Leuning *et al.* (1985) and Schjoerring *et al.* (1992). These samplers are passive, requiring no air to be drawn through them and, therefore, no electrical power at the measurement site. The sampler designed by (Leuning *et al.*, 1985) is coupled to a wind vane, so that the intake of the sampler always points upwind. The horizontal flux of ammonia is determined as the amount of ammonia adsorbed divided by the effective opening area of the sampler and sampling time. The samplers of (Schjoerring *et al.*, 1992) are of fixed orientation, but the ventilation rate inside the tube is the product of the average wind speed and the cosine of the angle between the sampler axis and the wind direction. An added advantage of flux samplers is that the integrated measurement of $\overline{u\chi}$ is more accurate (*Eq.* 1) than independent measurements of mean wind speed ($\bar{u}$) and concentration ($\bar{\chi}$) (Wilson & Shum, 1992).

5.4 Descriptions of the most commonly used methods

Of the methods outlined in Section 5.2, the ones most commonly used for measuring ammonia emissions from manure application are:

- Micrometeorological mass balance (IHF)
- Wind tunnels
- Equilibrium concentration (JTI).

Selection of the most appropriate technique will depend on the objectives of a particular experiment and the resources available. If measurements of absolute emission are required, then the micrometeorological mass balance technique is most appropriate in that it is non-intrusive and integrates the emission rate over a large area (thereby accounting for any variations in source strength across the plot). Disadvantages are the requirement for large uniform areas of land (plots 0.1 ha or greater, with sufficient spatial separation to avoid cross-contamination) that may limit the number of plots (and hence replication).

For comparative studies, small plot techniques may be used. Both the wind tunnel and the equilibrium concentration technique are suitable for use on small plots. Both methods interfere (the wind tunnels to a greater extent) with the microclimate in the plot. In addition, emission rates are likely to be greater from small plots, due to the much greater edge effect, than in a field-scale plot. There are also the disadvantages of high capital costs and the requirement for an adequate electricity supply. The advantage of the wind tunnel system is that it is relatively simple to operate. The equilibrium concentration technique has the advantages of low capital costs and portability, and an accumulator, only is needed to provide electric power. The main disadvantages of the technique are the requirements of high labour input and very clean laboratory procedures.

5.4.1 Micrometeorological mass-balance (IHF)

The most commonly used micrometeorological method for measurement of ammonia emissions from field applied manure (large plots > 0.1 ha) is IHF (*cf.* Section 5.2.2.1 and Equation 2) (Plate 5.1 and Appendix 5, Diagram 1).

Plate 5.1 The micrometeorological mast used for the measurement of ammonia emissions from field applied manure [Photo Courtesy of IGER, North Wyke].

Ideally, a circular area with a radius of 20-30 m is spread with manure, with a centrally placed mast supporting ammonia concentration or flux samplers, so that the fetch length is constant, regardless of the wind direction. In practice, accurate spreading of a circular plot is difficult to achieve, but as shown in Appendix 5, a pseudo-circle may be achieved by using farm-scale machinery. The first area to be spread should be the central strip, in line with the wind direction, so that the central mast can be placed in position and measurement can begin immediately. The remaining strips, decreasing in length, are spread thereafter. Alternatively, a mast can be placed at the centre of a square plot or downwind of a manure treated strip of land (with wind direction at 90^{o} to the strip length). In both of these cases, wind direction must be monitored frequently (*e.g.* every two minutes) and used in combination with plot geometry to calculate the fetch. Manure application rates to plots can be determined by weighing spreading machinery before and after application or by placing a series of trays over the spreading area that can then be weighed after application.

It is common for ammonia concentration or flux samplers to be mounted at five or six heights on the central (or downwind) mast (Plate 5.1) and at three heights on the upwind mast to determine the background horizontal flux. Typical sampler heights for a fetch length of 25-30 m would be 0.2, 0.5, 1.0, 2.0 and 3.3 m for the downwind mast (a rule of thumb is for the downwind mast height to be at least one tenth of the fetch length). Assuming the background flux is relatively constant with height, upwind sampler heights would typically be 0.5, 1.5 and 3.0 m.

The duration of sampling periods depends to a certain extent on the manure being applied, as well as on the application method and the environmental conditions. Typically, for splash-plate or other broadcast spread surface applications, samplers should be changed twice on the day of application, daily for the next three days and then every two to three days for a total period of 7-10 days. For application techniques designed to reduce ammonia emissions, sampling periods may need to be of longer duration.

A choice exists between using ammonia concentration or flux samplers. If the concentration is measured (typically by using absorption flasks), then the wind speed profile is also required. According to Ryden and McNeill (1984), the wind speed (u) and ammonia concentration (χ) profiles are related linearly to the logarithm of height (z)

$$\bar{u} = D \ln z + E \tag{10}$$

$$\bar{\chi} = -A \ln z + B \tag{11}$$

These can be substituted into Eq. (1), which, on integration, yields

$$F = \frac{1}{x}\Big[-AD\{z(\ln z)^2 - 2z\ln z + 2z\} + (BD - AE)\{z(\ln z - 1)\} + EBz - \overline{\chi_1}D\{z(\ln z - 1)\} - \chi_1 Ez\Big]_{Zo}^{Zp} \tag{12}$$

where χ_1 is the mean background concentration, z_o is determined from Equation (10) by setting u to zero, and z_p is determined from Equation (11) by setting χ to the mean background concentration.

The use of passive flux samplers (Leuning *et al.*, 1985) simplifies the practical application of the method in the field, negating the requirement for electrical power and the need to measure the wind speed profile. The internal surfaces of the samplers are coated with oxalic acid and allowed to dry. This is generally achieved by using 30 ml of a 3 % solution of oxalic acid in methanol (use of an ion exchange resin, such as Dowex 50/10W at 5 g l^{-1}, in the methanol eliminates ammonia contamination of the methanol, thus reducing blank values). Following exposure in the field, samples are extracted from the passive flux samplers by washing through with 40 ml deionised water that is subsequently analysed for ammoniacal-nitrogen. The horizontal flux measured at each sampling position is determined from the amount of ammonia collected in the sampler (M) according to

$$\overline{u\chi} = \frac{(M - M_b)}{A' t} \tag{13}$$

where M_b is the blank value (*i.e.* the amount of ammonium-nitrogen on an unexposed sampler), t is the duration of the sampling period and A' the effective cross-sectional area sampled. From (Leuning *et al.*, 1985)

$$A' = CAC_d^{0.5} \tag{14}$$

where C is the discharge coefficient, A is the cross-sectional area of the outlet orifice of the sampler, and C_d is the drag coefficient. For the samplers described by (Leuning *et al.*, 1985), C, A and C_d have values of 3.85×10^{-5} m^2, 0.62 and 1.0, respectively. The vertical flux from the plot can then be determined from the upwind and downwind horizontal flux measurements according to Equation (2).

The TPS method, described in Section 5.2.2.1, can be used as an alternative to the IHF method, with the advantage of much reduced sampler numbers, as the flux profile is determined from measurements at a single height only. Although the method has been verified against IHF on several occasions, it is not widely used, possibly due to the heavy reliance on a single sampler for a plot measurement. Sommer *et al.* (1995) reported it to be a robust method, with small deviations from the z_{inst} measurement height or in fetch length having relatively little effect on calculated emission. More consideration should be given to the use of this method in the future, particularly as replicated measurements of flux at the z_{inst} height may improve the accuracy of flux profile determinations compared with single samplers at each height, as used in the IHF method.

5.4.2 Wind tunnels

The most common design of wind tunnel used is that described by Lockyer (1984) (Plate 5.2 and Appendix 5, Diagram 2).

Plate 5.2 Typical wind tunnels used to measure ammonia emissions [Photo: Courtesy of IGER, North Wyke].

Differences exist in the equipment used to control and log the airflow through the tunnel. Briefly, the wind tunnels consist of two parts. The first part is a transparent section formed from a polycarbonate sheet (*e.g.* $2.0 \times 1.2 \times 0.002$ m) that is flexed and pinned to the ground along each 2 m edge to form a tunnel covering an area of 1 m^2 (0.5×2 m). The second part is a circular steel duct containing a co-axial fan to draw air through the transparent section. A flow "straightener" (anti-swirl device) is positioned at the rear of the duct. The fan is fitted with a speed control, and the air speed is measured by a vane anemometer mounted in the steel duct. To prevent wind from gusting through the tunnels, a large container (*e.g.* a dustbin) is fitted at the back of the duct, leaving a space of *ca.* 5 cm for air exit. Alternatively, a re-curved duct can be fitted to the duct outlet, so that inlet and outlet are both facing forward and will be subject to the same air pressure. The loss of ammonia from the area covered by each tunnel is the product of the volume of air flowing through each tunnel and the difference between the concentration of ammonia in air entering and leaving the tunnel.

There are few significant departures from the basic design described above. A system used in the Netherlands included a scrubbing system at the tunnel inlets to remove ammonia from inlet air. The tunnels, designed and constructed by the University of Hohenheim (Reitz and Kutzbach, 1997), differ more significantly. Inlet air is drawn in through a 4 m high chimney, and the tunnel is much longer, the first section being closed at the base. The measurement area is 4×0.5 m. To ensure a uniform ammonia concentration profile, outlet air is sampled following a mixing chamber.

Loubet *et al.* (1999) found that the position and design of the ammonia sampling point were shown important. Their measurements demonstrated non-uniformity of both the ammonia concentration and the wind speed profile within the tunnel. By modelling these, they simulated the recovery efficiency of a range of different systems for sampling outlet ammonia concentration. A single point sampler located at the centre of the duct gave 61% recovery, whereas a four-branched sampler with 20 sampling points spaced quadratically along the branches gave a simulated recovery efficiency of 100%. Smaller errors were associated with the positioning of the air flow measurement, but these errors could be reduced by modifying the duct section in order to stabilise the wind profile (Loubet *et al.*, 1999).

As the measurement area is relatively small, it is important that the manure application is uniform. This can be achieved either by using a specially designed small-plot applicator, or, with care, by the use of calibrated buckets or watering cans. Wind tunnels should be placed over the treated area, and measurement should be started immediately after application. Plots should be arranged in such a way that there will be no ammonia sources in front of wind tunnels. Ideally, plots should be in a line at 90° to the prevailing wind direction, with wind tunnels positioned with inlets facing the wind.

The duration of measurement will depend on the manure being applied, but, typically, it would be 1 week for liquid cattle or pig manure, 2 weeks for solid cattle or pig and up to 4 weeks for poultry manure (assuming that manure is not incorporated into the soil). The frequency of sampler changes will depend on whether detailed emission rate measurements or simply a cumulative total emission are required. Typically, more frequent changes should be made in the first two days after application, with daily changes from then on. If the intention is to measure emission under, as far as possible, ambient conditions, then the wind tunnels should be moved to a new part of the treated plot after rainfall events. Airflow through all wind tunnels should be continually logged, ideally in a format, which can easily be downloaded to a spreadsheet. Gas volume meters and airflow meters should be used to record and adjust airflow rates through the ammonia concentration samplers, taking care to ensure that readings are not affected by any pressure differences which may exist in the sampling air lines.

Ammonia emission is calculated according to Equation (8). Tunnel inlet and outlet air ammonia concentrations can be determined by the use of absorption flasks (typically) or active denuder tubes. The use of critical orifices in air sampling lines to ensure flow rates through samplers can greatly improve the accuracy of measurements (S. Sommer, *pers. comm.*).

5.4.3 Equilibrium concentration (JTI) method

This method is described briefly in Section 5.2.6 and relies on the determination of the equilibrium concentration of ammonia in air at the emitting surface, the concentration of ammonia in air just above the surface and the coefficient of mass transfer between the two (Plate 5.3 and Appendix 5).

Plate 5.3 The Equilibrium concentration technique, *i.e.* JTI method [Photo: Courtesy of IGER, North Wyke].

Passive diffusion samplers (PDS) are used to measure ammonia concentration in the air. Two types of PDS, which differ in the length of the diffusion path (Appendix 5, Diagram 3a), are used. For the L-type, the ammonia-absorbing filter is directly exposed to the ambient air. The C-type has the ammonia-absorbing filter placed 10 mm below a Teflon membrane filter. The amount of ammonia collected by the PDS-C type (X) and PDS-L type (Y) is given by

$$X = D\chi t \frac{A}{(L_R + L_{LBL})} \tag{15}$$

and

$$Y = D\chi t \frac{A}{L_{LBL}} \tag{16}$$

respectively, where D is the diffusion coefficient for ammonia in air, χ is the concentration of ammonia in the air, t is the exposure time for the PDS, A is the exposed area of the filter, L_R is the distance between the Teflon membrane filter and the ammonia absorbing filter for PDS-C type, and L_{LBL} is the laminar boundary layer above the top of the PDS. By combining Equations (14) and (15), an expression for L_{LBL} can be derived.

$$L_{LBL} = \frac{XL_R}{(Y - X)} \tag{17}$$

The mass transfer coefficient can then be derived from the relationship

$$K_{z,a} = \frac{D}{L_{LBL}} \tag{18}$$

Concentration of ammonia in the air can be determined by combining Equations 15 and 16.

$$\chi = \frac{XYL_R}{DtA(Y - X)} \tag{19}$$

By using the two types of PDS close to the surface of a treated plot (*ca.* 2 cm above the soil surface), both the concentration of ammonia in the air just above the emitting surface ($\chi_{a,z}$) and the mass transfer coefficient ($K_{z,a}$), required in Equation 7 can be determined from Equations (16) to (18). The diffusion coefficient is derived from the relationship

$$D = T^{1.5} \times 4.59 \times 10^{-9} \tag{20}$$

where T is the absolute temperature. The other parameter required in Equation (7), χ_{eq}, is determined by use of a ventilated chamber (Plate 5.3 and Appendix 5, Diagram 3b). A fan *via* a small inlet ventilates the chamber with outlet openings ensuring that condensation does not form on the internal walls. Inlet air is assumed to have an ammonia concentration $\chi_{a,z}$, as measured by the PDS outside the chamber. Ammonia flux from the area covered by the chamber can be calculated according to Equation (7) as

$$F_{ch} = (\chi_{eq} - \chi_{ch})K_{ch} \tag{21}$$

where χ_{ch} is the ammonia concentration of air inside the chamber, and K_{ch} *is* the mass transfer coefficient for ammonia inside the chamber (which should be constant for a given flow rate and given surface conditions). Flux from the chamber can also be calculated by mass balance

$$F_{ch} = (\chi_{ch} - \chi_{a,z})\,{}^{U}\!/_{A} \tag{22}$$

where U is the air flow rate through the chamber, and A *is* the area of emitting surface covered by the chamber. Combining Equations (20) and (21) gives an expression for the equilibrium concentration at the emitting surface

$$\chi_{eq} = \chi_{ch}\left(1 + \frac{{}^{U}\!/_{A}}{K_{ch}}\right) - \chi_{a,z}\left(\frac{{}^{U}\!/_{A}}{K_{ch}}\right) \tag{23}$$

By exposing both types of PDS within the chamber, values can be derived for χ_{ch} and K_{ch}, as they were for $\chi_{a,z}$ and $K_{z,a}$, using Equations (16) to (18), enabling derivation of χ_{eq} from Equation (22). This value can then be used in Equation (7) together with the determined values for $\chi_{a,z}$ and $K_{z,a}$, to derive the flux from the treated area for the measurement period.

The PDS is charged by soaking the cellulose filters in 3% solution of oxalic (or tartaric) acid in methanol and allowing them to dry in an ammonia-free air stream. It is important that blank values (*i.e.* ammonia-nitrogen measured on unexposed filters) are both low and consistent, to ensure maximum sensitivity of the method. Efficiency of absorption of ammonia nitrogen is linear up to 250 µg N per filter, after which saturation effects become significant. Extraction of PDS is by soaking the filters in 4 ml deionised water for a period of several hours, after which the extract is decanted to a clean vial for analysis.

Choice of appropriate sampling period duration is, perhaps, one of the most problematic aspects of this technique, as it is important that the duration is sufficient to enable detection of ammonia by ambient C-type PDS, but not so long that chamber L-type PDS becomes saturated. Typically, 3-6 measurements are made after manure application covering a period of 3 to 5 days, depending on manure type, application method, *etc.* On the first day following application exposure, periods of between 1 and 3 hours are typical, increasing to 4-8 hours on subsequent days and extending to 24 hours towards the end of the period. Chambers should be moved to a new position on the plot for each sampling period, otherwise, the cumulative effect of the chamber covering one section of the plot will become more significant. For application methods that do not result in a complete surface cover of manure (*e.g.* injection, band spreading), it is important to ensure that the area covered by a chamber is representative of the whole plot in terms of proportional slurry cover.

In conclusion, the advantage of this technique is that it measures ammonia loss from small plots. It should be noted that emission from small plots will be much higher than from large plots (Genermont & Cellier, 1997). The technique demands a high quality of the laboratory in which the absorbers are coated and ammonium concentration in the extract from the filters determined. Contamination with ambient ammonia will ruin the measurements. Furthermore, the technique measures losses from a very small area (*ca.* 30×35 cm^2). Thus, to get a reasonable estimate of the emission from a plot amended with manure, more than three replicates of each measurement are required. In addition, there is uncertainty regarding the accuracy of the method when used in crops with a height of >10 cm (Ferm *et al.*, 1999). The technique is not appropriate for measurements during rainfall, as the chambers will exclude rain, leading to probable overestimation of χ_{eq} and, therefore, emission.

5.4.4 Additional measurements

Measurements other than ammonia emission made during the course of an experiment will be important both to identify the experimental conditions and to enable interpretation of results.

Publication of such additional measurements will also aid comparison of results between research groups and model development, as illustrated in Chapter 4. Previous research has identified a large number of factors that may influence emission, and some of these are discussed in Chapters 2 and 4, particularly in relation to model development. Which additional measurements should be made will, to a certain extent, depend on the objectives of the experiment and the resources available. However, on the basis of the experience gained during this project, a protocol for measurements is presented in Table 5.1.

Table 5.1. Measurement protocol for ammonia emission experiments determining losses from field applied manure

Parameter	Desirability	Frequency
Manure		
Dry matter	1	At application
Total ammoniacal nitrogen	1	At application
pH	1	At application
Flowability[‡]	3	At application
Weather		
Wind speed	1	At frequent intervals (*e.g.* 5 min) throughout the experiment
Wind direction	2	
Air temperature	1	
Relative humidity	3	
Soil surface temperature	1	
Rain	1	
Soil		
Moisture content	1	At application
Texture	1	At application
pH	2	At application
CEC	3	At application
Compaction	2	At application
Infiltration rate	2	At application
Buffer capacity	3	At application
Surface pH	3	At intervals throughout the experiment, more frequent at the beginning
Total ammoniacal nitrogen	3	At application
Crop		
Type	1	At application
Height	1	At application
Biomass	3	At application
Development stage	2	At application
Leaf area index	2	At application
Machine		
Injection slot volume	1	At application for injectors only
Slurry band width	1	At application for low emission surface spreaders (*e.g.* trailing shoe, trailing hose)

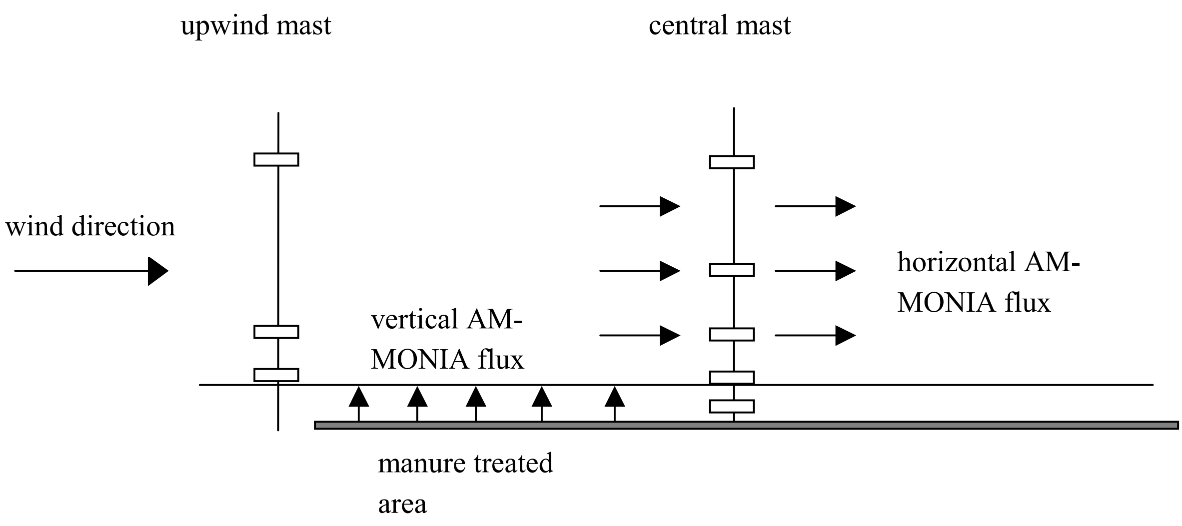
[†] 1, essential; 2, desirable; 3, useful

[‡] after Malgeryd and Wetterberg, (1996)

APPENDIX (Chapter 5)

Diagram 1. Diagrammatic representation of the micrometeorological mass balance technique for measuring ammonia emissions

a) side view



b) plan view

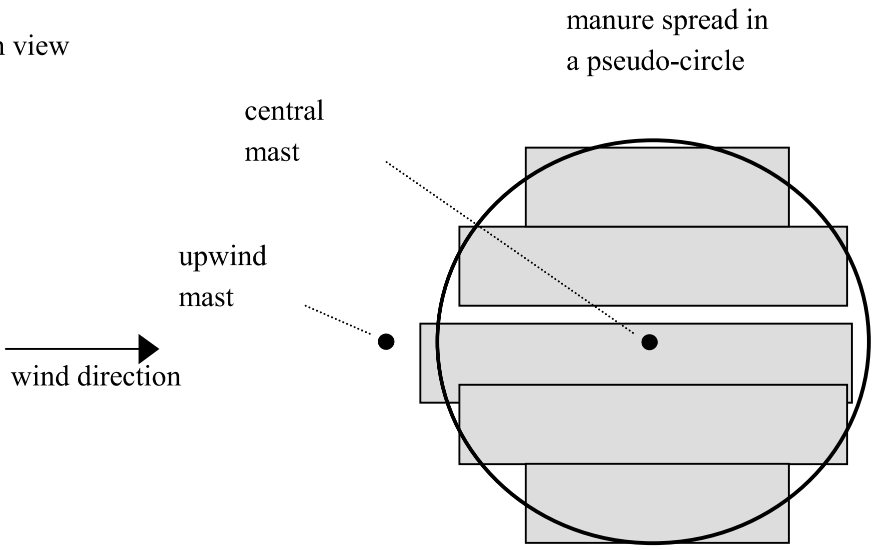
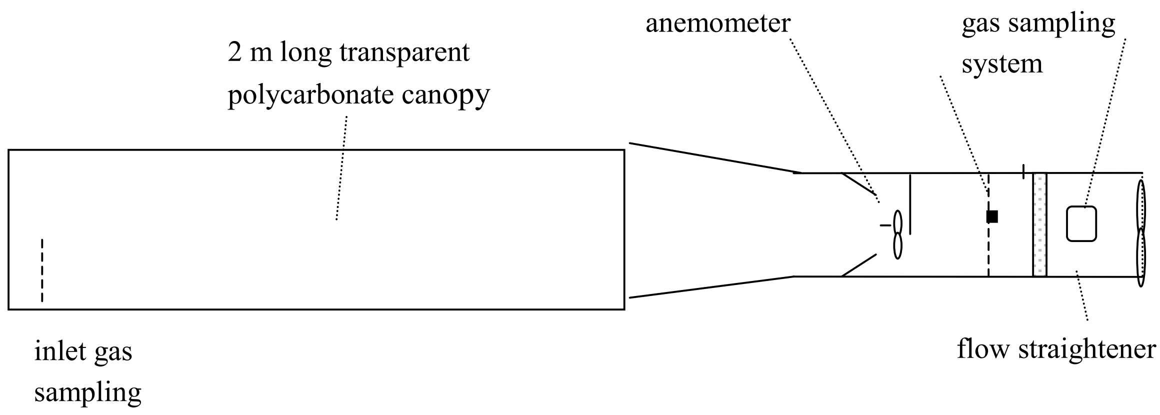
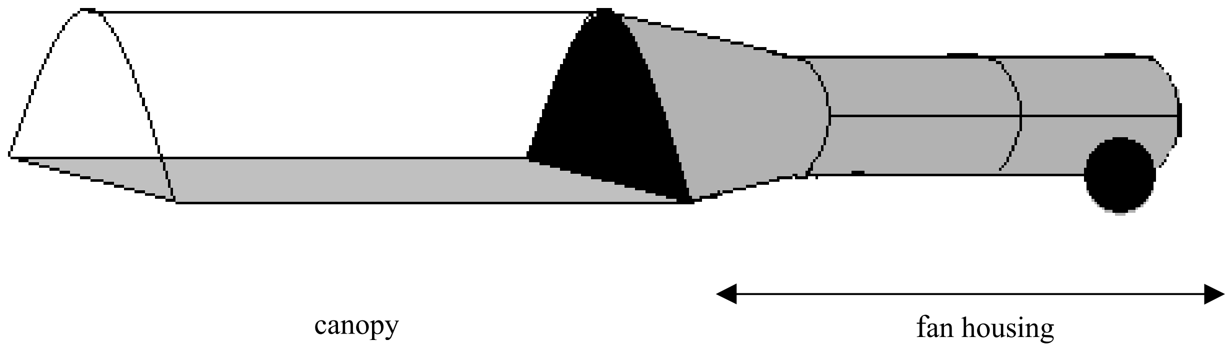


Diagram 2. Diagrammatic representation of a wind tunnel for measuring ammonia emissions

a) schematic diagram



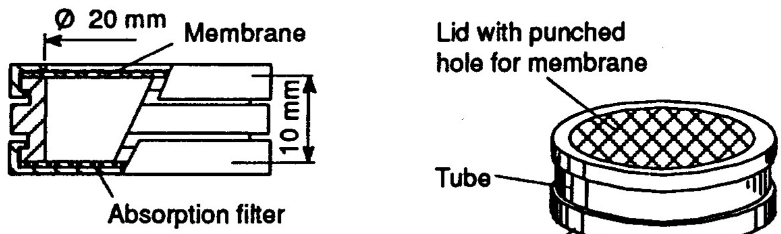
b) pictorial representation



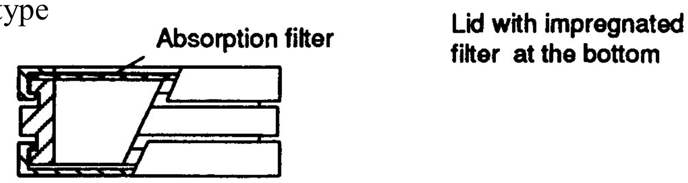
Diagrams 3a & 3b. Diagrammatic representation Passive diffusion samplers (a) and dynamic chamber (b) used in the equilibrium concentration technique (from Svensson, 1994)

(a)

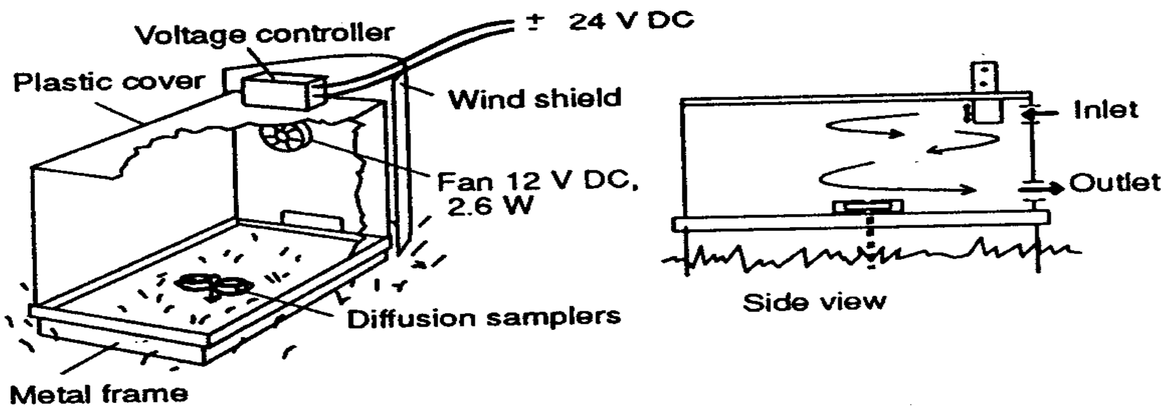
C-type



L-type



(b)



6. Ammonia losses from field applied manure – An overview of the abatement strategies

6.1 Introduction

Sommer & Hutchings (1995) reviewed the various abatement techniques and strategies for ammonia emission from agriculture, including those applicable to animal houses, manure storage and field applied manure. The authors highlighted that the abatement strategies at the various stages of livestock production systems are interdependent, and combinations of measures are not simply additive in terms of their combined emission reduction potential.

The ammonia abatement technologies and strategies are constantly under review and new research will continue to provide guidance in terms of improved abatement strategies. In this context it is interesting to note that the Guidance Notes, prepared by a Working Group on Ammonia Abatement Techniques under the auspices of the UN/ECE (1999), address the known potential abatement measures. They group them into three categories: (a) techniques considered to be practical and reliable in terms of existing quantitative data; (b) techniques that are promising but still need further development; and (c) techniques that are considered as ineffective or unpractical.

This chapter presents a summary of the two main approaches, generally adopted in terms of abatement strategies for reducing ammonia losses from field applied animal manure:

- Dietary measures
- Manure treatment
- Amendments, adsorbents and commercial additives
- Spreading techniques

6.2 Dietary measures

Dietary measures have an indirect influence on emissions. They can either reduce the quantity of nitrogen excreted and thus, the subsequent ammonia emission potential, or change the characteristics of the excreta (ammonium content, pH, *etc.*), so thatthe emission potential is reduced. Measures that reduce the nitrogen excretion in animal manure limit the total agricultural nitrogen flux, thereby reducing all forms of nitrogen emissions to air and water. By changing the emission potential of the manure, dietary measures can influence all the stages where ammonia emissions from manure occur, i.e. housing, manure-storage, manure application and grazing. Therefore, dietary measures are pivotal, if the overall nitrogen efficiency of animal based agricultural systems is to be improved.

The greatest potential for avoiding nitrogen losses and for increasing the efficiency of its conversion into animal product lies in improving the protein quality and minimising the nitrogen intake (Kirchgessner *et al.*, 1994). Many authors have indicated the various approaches to achieving dietary control of nitrogen content in excreta and reduction in ammonia loss (*e.g.* Stadelmann *et al.*, 1998; Menzi *et al.*, 1997). These include

- reduction of the excess crude protein in the ration
- reduction of the ration crude protein concentration by optimising the amino acid supply
- increase in the efficiency of nitrogen utilisation by improved animal feed conversion efficiency.
- use of feed additives to change the nitrogen composition of manure

The dietary measures to reduce nitrogen excretion by cattle (ruminants), pigs and poultry are discussed briefly in the following sections.

6.2.1 Dietary measures for cattle

Tamminga (1990, 1992) and Castillo *et al.* (2000) identified a number of strategies to reduce ammonia losses from dairy cows through dietary measures:

- Reduction of the nitrogen intake by adjusting the ration supply to reflect changes in the animals' different physiological requirements (dry cow versus milking cows),
- Improvement of the feed quality
- Increase in the level of production
- Matching or synchronising the nitrogen availability and energy supply in the rumen to improve the efficiency of nitrogen capture in the rumen through microbial protein synthesis
- Shifting digestion of protein and starch from the rumen to the small intestine.

Rations with excess protein and deficiency in energy lead to higher nitrogen excretions (mainly water-soluble nitrogen) and lower carbon concentrations in the slurry (Sutter and Kreuzer, 1995; Kirchgessner and Kreuzer, 1986). For each percentage point increase in the crude protein content of a dairy cow ration, nitrogen emissions increased within the range of 10 to 20% (Smits *et al.*, 1995; Paul *et al.*, 1998; Kröber *et al.*, 2000; Külling *et al.*, 2001a and b). James *et al.* (1999) reported similar results for heifers. Castillo *et al.* (2000) suggested that reducing the crude protein content of a cattle diet from 200 to 150 g kg^{-1} dry matter would reduce the annual nitrogen excretion in faeces by 21% and, more importantly, in urine by 66%.

Nitrogen emissions from ruminants can be reduced by supplementation or partial replacement of grass and grass silage rations with low-protein feeds, such as maize (Vurren van *et al.*, 1993a), hay (Külling *et al.*, 2001b), sugar beet pulp (Vurren van, 1993a), cereal-based concentrates, or starch concentrates (Vurren van, 1993a and b). The urinary nitrogen excretion of ruminants was reduced by 30 to 40% when grass (ryegrass receiving high rates of nitrogen fertiliser) was partially replaced (Vurren van *et al.*, 1993a). Menzi *et al.* (1997) estimated the emission reduction potential of supplementing high protein rations with maize, hay or beets at 10 to20%.

It is worth noting that reduction of the nitrogen excretion will also reduce the potential for nitrous oxide emissions, a powerful “greenhouse” gas, from manure (Oenema *et al.*, 1997), but may increase methane emissions.

A reduction in nitrogen excretion of dairy cows by improving animal production performance will contribute to reduced potential for ammonia loss. Menzi *et al.* (1997) estimated that an increase in milk yield of 1000 kg $year^{-1}$ per cow would reduce the nitrogen excretion and thus the emissions per unit milk production by 10-15%.

6.2.2 Dietary measures for pigs

Several authors (Lenis and Jongbloed, 1999; Dourmad *et al.*, 1999a and b; Peet Schwering van der, *et al.*, 1999 a and b) report that reductions in nitrogen excretion and ammonia emissions from pig production can be achieved by:

- phase feeding (changes in the ration’s nitrogen supply to reflect more accurately thechanges in the animal’s physiological requirements) and/or supplementation of diet with amino acids
- including additional non-starch polysaccharides in the diet
- adding acidifying salts to the diet.

In addition, to improve the performance of fattening pigs (faster growth with less feed) will improve the nitrogen efficiency of the system (Aarnink *et al.*, 1997), and thus, the potential for reducing overall ammonia emissions from pig enterprises.

6.2.2.1 Lowering the protein content of pig feed

In the past, standard diets often met the nutritional needs of the pigs for the most limiting amino acids, but exceeded the requirements for other amino acids. Furthermore, the same feed was often used during the entire fattening period or for gestating and lactating sows. This feed had to meet the maximum demand of the animals, which resulted in a surplus in the nitrogen supply for a significant part of the production period. By using different feed for different growth stages, the feed conversion efficiency is improved, and excretion of nitrogen and minerals can be reduced without compromising production. Nitrogen excretions can also be reduced by supplementing the diet with essential amino acids in pure form, according to the needs of the animals. The amino acids that have to be supplemented depend on the basic components of the feed. Those most commonly used ones are lysine, methionine, thrionine and tryptophan. The effect of different feeding strategies is shown in Table 6.1. In a field study Misselbrook *et al.* (1998) measured a 60% reduction in ammonia emissions in the five days following the application of manure produced by pigs fed on a diet with a 14% rather than a 20.5% crude protein content. In addition, the same authors reported a reduction in nitrous oxide emissions of 73% over 51 days after application for the lower crude protein diet.

The ammonium content of the slurry, and thus, the emissions are also influenced by the water:feed ratio. In the experiments of Aarnink *et al.* (1997) a reduction in this ratio from 2.4 to 2.2 resulted in an increase of the ammonium content of the slurry of 11%.

Table 6.1 Reduction in N excretion through changes in feeding practises

Feeding strategy	Reduction in %		Reference
	N excreted	Ammonia loss	
Low N-diet		20-30	Dourmad et al (1999b); Menzi *et al.* (1997)
Phase feeding (3 phases)	15		Spiekers *et al.* (1990)
Phase feeding (3 phases) + amino acids	40		Spiekers *et al.* (1990)
Adding lysine	40		Kirchgessner *et al.* (1994)
Change from 18% crude protein to 10% + essential amino acids	>40		Sutton *et al.* (1997)
Reduced N in feed		46 (growing) and 46 (finishing)	Kay *et al.* (1997)
N crude protein reduction		10-12.5% reduction per % decrease in dietary crude protein (interval 16.5-12.5% crude protein)	Canh *et al.* (1998a)

6.2.2.2 Including additional non-starch polysaccharides in pig diets

Fermentable carbohydrates in pig rations lead to the formation of volatile fatty acids in the small intestine and in the manure. This lowers the manure pH, which reduces the ammonia volatilisation potential. Canh *et al.* (1998c) found that for each 100-g increase in the intake of dietary non-starch protein in replacement of corn starch, the manure pH decreased by approximately 0.12 units and the ammonia emission from slurry decreased by 5.4%. In these experiments, the addition of soyabean hulls to the diet had a significant effect on manure pH, with similar effects from additions of sugar beet pulp and coconut expeller. Canh *et al.* (1997) reported that the pH of manure decreased by 0.4 to 0.5 units, and the ammonia emission decreased by approximately 15% for each 5% increase of sugar beet pulp in the ration. Kreuzer *et al.* (1998) reported that feeds high in pectin (citrus pulp) and hemicellulose (beet pulp) were the most effective treatments to reduce gaseous nitrogen losses. The effects of fermentable non-starch protein supply were further enhanced by a reduction in dietary protein. The synergistic effect of low protein diets and the addition of low starch poly-saccharides were also demonstrated by Sutton *et al.* (1997), who used a low-level addition of an oligosaccharide and cellulose. Work by Kreuzer *et al.* (1998) also indicated that diets high in bacterially fermentable substrate are effective in reducing nitrogen emissions by reducing the urinary nitrogen excretion. The ratio of urinary to totally excreted nitrogen decreased by 0.07 to 0.18 percentage units for each g of additional fermentable non-starch proteins kg^{-1} dietary DM in the range of 160 to190 fermentable non-starch proteins kg^{-1} dry matter.

6.2.3 Dietary measures for poultry

Theoretically, measures reported for pigs, such as phase feeding and supplementing limiting amino acids, can also be used for poultry (Elwinger and Svenson, 1996). Considerably less work on diet manipulation to reduce nitrogen excretion for poultry has been reported in the literature, compared with that reported for pigs. This may reflect the higher protein or nitrogen conversion efficiency for poultry. In addition, the scope for strategies like phase feeding is more difficult for poultry than for pigs, because of the large numbers of animals involved. Menzi *et al.* (1997) estimated that under Swiss conditions, the maximum reduction in poultry nitrogen excretion through phase feeding would be 5-8% for layers and less than 5% for broilers. The lower value for broilers is due to the variability in growth rate between individual birds. These values appear to be somewhat lower than those given by Priesmann and Petersen (1995) and Mennicken (2000). These authors estimated that phase feeding of laying birds based on the weight and health of the animals and the climatic conditions could reduce ammonia emissions by 8-12% per 1% (10 g) reduction of the average crude protein content of the feed. Ferguson *et al.* (1998) suggested that for every percentage point of reduction in dietary crude protein (and supplementation with amino acids), there will be a corresponding 7% reduction in the nitrogen content of the litter within the acceptable range of crude protein in commercial broiler grower diets (days 22-42; 19-21% crude protein).

6.3 Manure treatments

A reduction in the viscosity of slurry will reduce ammonia losses, although it will not be effective if the infiltration rate is limited by other factors, such as soil moisture content (Rubæk *et al.*, 1996). The viscosity of the slurry may be reduced by dilution with water (*cf.*, Section 6.4.2.1), by reducing the fibrous fraction by mechanical separation (Sommer & Olesen, 1991; Frost, 1994), or by anaerobic digestion (Rubæk *et al.*, 1996). However, the effect of such treatments in terms of ammonia emission abatement may be difficult to predict.

6.4 Amendments, adsorbents and commercial additives

6.4.1 Background and evolved mechanisms

Over the last three decades, a large number of manure amendments, absorbents and additives have come and gone from the market. Generally, these products claim to modify the composition of manure by increasing the biological/chemical stability or by improving flow properties of liquid manure. Specifically, many people claim to reduce odour emissions and more recently, ammonia emissions from manure. The materials used in these products include water, bacterial enzyme preparations, plant extracts, oxidising agents, disinfectants, urease inhibitors, masking agents, acidifying compounds and substances such as peat and clay minerals that act as adsorbents. Very few have been subjected to independent, evaluation, so their effectiveness in terms of the claimed benefits remains uncertain. Furthermore, the product evaluations to date have generally involved their impact on stored slurry or the impact on emissions arising from diet amendments. Very few of these evaluations addressed the use of these products in terms of their impact on ammonia emissions following field application of manure.

6.4.2 Types of additives

The two most commonly used additives for reduction of ammonia emissions from field application of manure are water and inorganic acids.

6.4.2.1 Water

Sommer & Olesen (1991) reported a linear relationship between slurry dry matter content and ammonia emissions. The more diluted the slurry, the higher the soil infiltration potential with consequential reductions in ammonia emissions. Dilution with water is only appropriate for slurries with initially high dry matter contents. Frost (1994) demonstrated that dilution of slurry by 90-120% by volume lowered the dry matter content from 10.3 to 5.5% and the ammonia emissions by 50% compared with the undiluted slurry. The abatement efficiency of the strategy is also dependent on soil conditions. Rapid infiltration is inhibited for waterlogged or compacted soils. Dilution of slurry requires that an adequate supply of water is available and has the disadvantage of increasing the volume of slurry that has to be transported and applied in the field.

6.4.2.2 Inorganic acids

The equilibrium between ammonium and ammonia in solution is pH dependent. Reduction of the pH of slurry to 5.5-6.0 by adding sulphuric acid has shown to reduce ammonia losses by 30 to 95% following surface application (Stevens *et al.*, 1989; Pain *et al.*, 1990). Nitric acid has also been used (*e.g.* Bussink *et al.*, 1994; Stevens *et al.*, 1997) and has the advantage of increasing the nitrogen content of slurry, thereby resulting in a more balanced fertiliser (in terms of its nitrogen, phosphorus and potassium content). Two disadvantages of slurry acidification have discouraged its adoption at farm scale. Firstly, the use of strong acids on farms is hazardous, and specialist equipment is required, thus increasing the costs. Secondly, acidified slurry, especially slurries treated with sulphuric acid, will result in acidification of the soil. Addition of nitric acid to anaerobic and carbon-rich slurry may also promote losses of nitrogen through denitrification (Oenema & Velthof 1993).

6.4.2.3 Commercial additives

Commercial additives based on saponins may be extracted from the sap of yucca plants. Saponins are high-molecular-weight glycosides, consisting of a sugar part linked to a triterpene or steroid aglycone. The mechanism by which saponins conserve ammonia is unclear, but it was suggested that they act as a binding or converting agent for ammonium (Amon *et al.*, 1995). Products of this type are mainly used as feed additives.

Heber *et al.* (1997) reported the results of a field trial in which a reduction in mean ammonia emissions from 5.9 to 1.8g day^{-1} pig^{-1} was achieved by using a commercial manure additive. Martinez *et al.* (1997) evaluated five commercial additives in terms of their potential to reduce ammonia emissions. They reported that two of the five additives achieved reductions in ammonia emissions of up to 40 to 50% – both during storage and following field application.

6.4.2.4 Acidifying compounds

The addition of soluble magnesium and calcium salts has been shown to significantly reduce ammonia volatilisation. The proposed reaction mechanism involves the precipitation of carbonate formed during urea hydrolysis as calcium or magnesium carbonate. Witter (1991) showed that an addition of $CaCl_2$ to chicken slurry (at the rate of 36 mg Ca g^{-1} dry wt) reduced the ammonia volatilisation by 73%, compared to untreated control. In laboratory and field trials Vandré & Clemens (1997) found that acidification of manure additives would potentially reduce ammonia losses. They suggested to use slurry amendment with ground gypsum ($CaSO_4$) at moderate rates (0.05 to 0.1 mol kg^{-1}) prior to application to reduce ammonia losses.

The addition of acidifying salts instead of $CaCO_3$ to pig feed may reduce the pH of the urine and the slurry and thus, ammonia emissions (Aarnink *et al.*, 1997; Canh *et al.*, 1998b; Peet Schwering van der *et al.*, 1999a and b). Hendriks *et al.* (1997) achieved a 87% reduction of the ammonia emission from pig houses by replacing the chalk ($CaCO_3$) in the diet with a mixture of acid salts. In the experiments of Canh *et al.* (1998b), the use of $CaCl_2$, $CaSO_4$ and Calcium benzoate in pig diets reduced the ammonia emission by 30, 33 and 54%, respectively.

6.4.2.5 Absorbents

Reductions in ammonia emissions can be achieved by amending manure – particularly solid manure – with organic materials such as straw and sphagnum peat moss. Amendments will be effective, if they either adsorb the ammonium or ammonia, reduce the manure pH, or promote the microbial immobilisation of the nitrogen.

Peat is a natural material of variable composition capable of absorbing large amounts of gases and liquids. Peat can be considered as a complex of polycarboxylic acids exchanging protons with cationic species on an equivalent basis. Peat – in particular *Sphagnum fuscum* peat – appears to show a high adsorptive capacity. This ability is derived from the physical and chemical properties of dead leaves and stems. Leaves offer a very large surface area and are no thicker than a single layer of cells, and are highly porous.

According to Subair *et al.* (1999), paper products have potential as an amendment to reduce ammonia losses from manure, because their high carbon to nitrogen ratio would be expected to promote the immobilisation of the mineral nitrogen fraction. Compared to the control slurry, an increase in the addition of various paper products to pig slurry from 2.5 to 5%, reduced ammonia volatilisation by 29 and 47% depending on the product.

Zeolites are naturally occurring alumino silicate minerals with high cation exchange capacities. There are many different types of natural zeolites differing in their selectivity towards different cations. For example, clinoptilolite has a specific affinity for ammonium ions. Miner (1984) found that the application of 1 to 4% (w/v) finely ground clinoptilolite to dairy slurry, immediately before spreading through a sprinkler system, reduced the ammonia emission rates by up to 60%.

The advantages of using either clinoptilolite or peat for conservation of manure nitrogen are that they are non-hazardous and have soil conditioning properties.

6.5 Low emission manure spreading techniques

Ammonia emission following the field application of manure can account for up to 40% of the total ammonia emission from agriculture. Strategies and technologies to reduce ammonia losses following application can make an important contribution to reducing overall losses.

Infiltration of manure to the soil, or incorporation of manure into the soil matrix, will reduce the quantity of ammonia available for volatilisation. The effectiveness of the soil/manure interaction depends on the pH and cation exchange capacity of the soil. The direct incorporation of manure following application or the direct injection of the manure into the soil is effective for reduction of ammonia emissions.

The two following types of application technique can be identified: manure application on the surface of the soil or crop (surface spreading), and placement of manure in the soil matrix (injection). Manure can be incorporated into the soil matrix, either by ploughing or by other suitable cultivation following its application to arable land or grassland that is to be ploughed. There is a large range of manure application systems for grassland designed to reduce the surface area of slurry exposed to the air and/or to increase the rate of infiltration into the soil. Many published reports address the potential of these techniques to reduce ammonia volatilisation following field application of manure. A summary of these results is presented in Figure 6.1.

6.5.1 Low emission spreading techniques

A range of low emission manure spreading techniques are available to reduce emissions (Chadwick, 1997). The choice of technique used by the farmer will vary, depending on the relevant national and EU legislation, the climate, the restrictions imposed by the soil, the topography, and the farming system and cost (see Chapters 7 and 8). A brief review of the available low emission techniques is presented here. The techniqus are divided into the three groups of surface application, soil injection and incorporation.

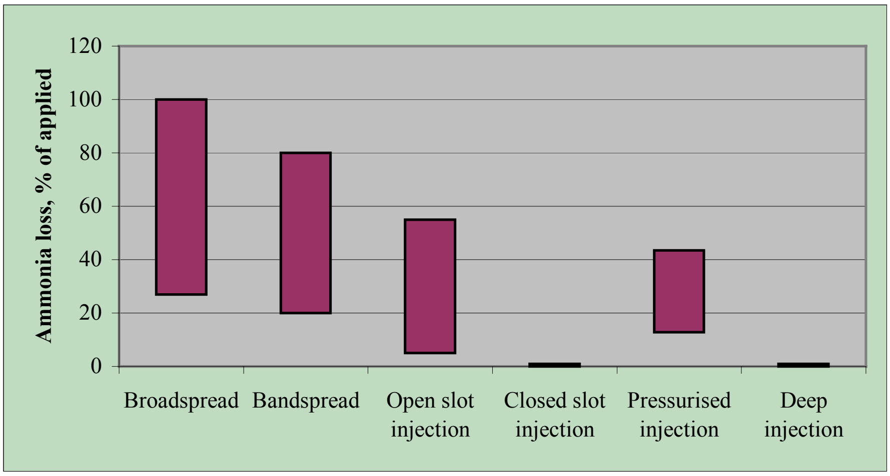


Figure 6.1 Summary of the ammonia loss (the percentage loss of total ammoniacal nitrogen applied) from field applied manure, using a range of application methods. [Bandspread = trailing hose & trailing shoe] (Summary of the findings of; Bless *et al.*, 1991; Dosch and Guter, 1996; Frick and Menzi, 1997; Frost, 1994; Hol and Huijsmans, 1998; Huijsmans and Bussink, 1990; Huijsmans, 1998; Morken and Sakshaug, 1998; Rodhe and Karlson, 2000; Rodhe *et al.*, 1997; Rubæk *et al.*, 1996, Sommer *et al.*, 1997; Steffens and Lorenz, 2000; Vandre *et al.*, 1997; Weslien *et al.*, 1998).

6.5.1.1 Surface application

Placement of the manure in bands beneath a crop will reduce emission compared to broad spread/splashplate applied manure (Plate 6.1) (Figure 6.1). Band spacing is 25-30 cm, depending on the manufacturer.

Plate 6.1 Broadcast spreader; still the most common method in most European countries [Photo: Courtesy of J.F.M. Huijsmans, IMAG].

The reduction occurs because the crop canopy changes the microclimate near the soil surface, *i.e.* lower wind speeds, temperature and radiation, and increased relative humidity (Thompson *et al.*, 1990a and 1990b). The extent of the reduction reported varies between authors, but this can be explained in terms of the differences in crop height and leaf area index. Furthermore, ammonia emissions from the slurry can be absorbed by the leaves of the crop (Sommer *et al.*, 1997). Some reduction is achieved in the absence of a crop, due to the reduced area of manure exposed to air.

Plate 6.2 Trailing hose slurry spreader. It places the slurry on the soil in bands in the rows between the plants and is particularly suitable for use in tillage crops [Photo: Courtesy of J.F.M. Huijsmans, IMAG].

There are two types of low emission surface spreading systems, namely trailing hose and trailing shoe systems.

Trailing hose system: The trailing hose system applies the slurry onto the soil between rows of plants (Bless *et al.*, 1991; Döhler, 1991) (Plate 6.2). The reduction in cumulative ammonia emission from manure applied with the trailing hose system is consequently related to the crop cover with lower losses in places, where the cover is high (Sommer *et al.*, 1997; Sommer & Olesen, 2000).

Trailing shoe system: The trailing shoe system is a development of the band spreader (Plate 6.3). On grassland, the trailing shoe system allows placement of the slurry beneath the grass canopy rather than on the surface. Some trailing shoe machines also cut a shallow slot into the soil, so that the rate of infiltration of the slurry into the soil is increased.

Plate 6.3 Trailing shoe slurry spreader. It places the slurry beneath the canopy thereby reducing the potential for ammonia emission compared with surface applied slurry bands [Photo: Courtesy of J.F.M. Huijsmans, IMAG].

6.5.1.2 Soil injection

The injection of manure into the soil has the double advantage in terms of controlling ammonia emissions, because injection reduces the exposed slurry surface area and promotes infiltration into the soil. There are three basic designs of the injector tool. The first one uses two disc coulters. The coulters have opposite horizontal angles, and when pressed into the soil, they form a V-shaped slit into which the manure is placed. The second design uses a disc coulter to cut the soil, and this is closely followed by a V-shaped knife, which opens a slot beneath the surface. The third design uses wheels of increasing thickness towards the centre and its effect is similar to the two-disc coulter system. When the wheel is pressed into the soil, a V-shaped slit is formed.

Many different types of machine are commercially available, and they can be grouped into the following basic designs:

Shallow injection: An array of injector units is normally fitted to a tool bar that is either mounted on a tanker or directly on the tractor (Plate 6.4). A distribution system, often incorporating a chopping system to reduce blockages, distributes slurry to each injector unit via pipes. Each injector cuts a slot (*ca.* 5 cm deep) into which the slurry is placed. Some machines are fitted with wheels behind each injector unit to close the slot.

Plate 6.4 Shallow injector. Slurry is placed in shallow slots (5 cm) cut into the soil by using either or both a disc and a coulter. The slots can be left open or closed [Photo: Courtesy of J.F.M. Huijsmans, IMAG].

Deep injection: This concept typically comprises a series of large tines fitted with lateral wings that place and disperse the slurry 10-20 cm beneath the soil surface. This often results in a greater reduction of ammonia emission, but also in higher equipment costs and a higher fuel consumption.

Pressurised injection: The pressurised injection concept involves the use of a high pressure pump which forces the slurry through a series of nozzles distributed along a boom (Plate 6.5).

Plate 6.5 The DGI or pressurised injection system [Photo: Courtesy of J. Morken, AUN].

The nozzles (10 to 15 mm in diameter, depending on the required application rate) are located in skis or shoes that slide on top of the soil. Rotating knives ensure that nozzles do not block and cause the slurry to jet out from the nozzles in pulses. The pulses are powerful enough to inject the slurry into porous soils in a series of elongated, discontinuous cavities.

6.5.2.3 Incorporation techniques

Incorporation of the manure into soil following field application will stop ammonia volatilisation, so where soils are to be tilled for cropping, manure can be incorporated as part of the cultivation procedures. The manure can be applied on grassland or cropland and ploughed in (Plate 6.6.). Alternatively, the manure can be worked into the soil following application by use of tines, dics or rotary harrows.

Plate 6.6 Ploughing in manure immediately after application will reduce ammonia emissions [Photo: Courtesy of J.F.M. Huijsmans, IMAG].

The efficiency of incorporation is not only dependent on the type of harrow used, but also on the management of the harrow, the soil texture and the soil conditions. Ammonia volatilisation will increase with time between application and incorporation. As soon as slurry or solid manure is incorporated, ammonia losses will be greatly reduced. The efficiency of the reduction is dependent on the method used.

6.5.2.4 Application strategy

The timing of the manure application can reduce the potential for ammonia losses. Sommer & Olesen (2000) estimated that by avoiding applications during the periods in the day when losses are highest, potential emissions could be reduced by half. However, this may not be practical for a number of reasons. These include a combination of the relatively short periods during the year when manure can be spread and the generally large quantities to be applied, the unpredictable occurrence of suitable soil and weather conditions, the legislative controls on spreading dates and other demands on labour and machinery. Furthermore, the efficiency of the incorporation in terms of reducing ammonia emissions depends on the soil conditions, the extent to which the manure is incorporated into the soil (efficiency of incorporation), and on the time between manure spreading and incorporation. Cultivation of the soil surface before the manure application can reduce ammonia losses by approximately 50%, compared with application to uncultivated soil. The reduction can be achieved through higher infiltration rate of the manure into the soil (Bless *et al.*, 1991, Sommer & Ersbøll, 1994).

6.6 Practical considerations

Effectiveness in controlling emissions, applicability and costs should be taken into account when selecting the most suitable techniques for reducing ammonia emissions. Band spreaders and injectors can be expected to reduce emissions from liquid manure by 30-80%, compared with the emission from application to the surface with a conventional broad spreading or splash plate spreader. Band spreaders and injectors are not suitable for use on steep slopes, and sub-surface injection techniques do not work well on very stony or compact soils.

Small, irregularly shaped fields present difficulties for large machines. Incorporation is restricted to land that is cultivated. Umbilical systems, where the applicator is mounted directly on the tractor and fed from a tank *via* a long hose, offer an alternative to mounting the applicator on a tractor drawn tanker. They have the advantage of higher work rates and of reducing the risk of soil damage.

A final consideration is that the low emission spreading techniques improve the evenness of the manure application. Liquid manure or slurry can vary in terms of viscosity, which makes it difficult to predict spreading width, evenness, and application rate when using a broad spreading or splash plate spreader. Band spreaders and injectors have fixed working widths, which are more independent of the slurry viscosity and the flow rate from the tanker. When the working width is fixed, it will enable adjustment of the flow rate according to the nutrient requirements of the crop.

A summary of the practical implications of the spreading techniques is presented in Table 6.2.

Table 6.2 Practical considerations in selecting spreading technique for ammonia abatement following field application of manure

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

6.7 Conclusions

There are three main strategies to address ammonia emissions from the land application of manure:

- Dietary measures
- Amendments, adsorbents and commercial additives
- Spreading techniques

Reduction of the nitrogen excretion of animals through dietary manipulation will not only reduce ammonia from field applied manure, but also from housed livestock and from manure stores. The efficiency of feeding strategies for reduction of ammonia losses from poultry production has not been assessed.

Amendments, adsorbents and commercial additives have been shown to control ammonia emissions from manure but are yet to gain widespread acceptance because of the costs, safety and practical considerations involved.

Control of the ammonia emission after field application of manure by using a range of spreading techniques has been the subject of much research in recent years. These techniques include band spreading (trailed hoses and trailed shoes), shallow injection (with or without closed slots), deep injection and pressurized injection. Ammonia losses with these techniques can be variable; from negligible to 80% of the ammonium applied, compared to 30 to 100% losses with conventional broadcast spreading. While the greatest reductions are achieved with injection systems, there are some practical disadvantages and limitations. Incorporation of manure prior to the tilling of the soil will also result in a reduction of ammonia emissions. Cultivating the soil surface before surface application of slurry can reduce ammonia losses to about 50%, compared with cultivation of uncultivated soil.

7. Costs of field application of manure in Europe

7.1 Introduction

The reduction of ammonia losses is a major concern and an important component of environmental pollution control strategies in many European countries. Recently, new manure application techniques have been developed reduction of ammonia emissions after field application of manure (*cf.* Chapter 6). However, these techniques generally require more draught force and require initial capital investment than conventional broadcast spreading with splash plate (Rodhe & Rammer, 2001). When large quantities of manure are handled, it may be profitable to invest in more environmental friendly technology like low trajectory systems, *e.g.* shallow injectors, (Brundin & Rodhe, 1994). Uncertainties about the complexity of manure spreading operations and the perceived high costs of implementing improved practice (*e.g.* extra storage capacity, new machinery) are among the factors thought to be responsible for poor farmer confidence in manure nutrients, particularly nitrogen, for crop production in some countries (Smith *et al.*, 2000).

The economics of manure handling and spreading are therefore of fundamental importance in encouraging the improved recycling and efficient utilisation of manure. The economics depend on the machine costs and the time required for field application. The machine costs depend on the operating costs, depreciation, interest on capital and insurance. The time required for field application depends on a number of important factors (including field area and layout, machine operational speed, machine width, distance to manure storage) and the work pattern. The costs of the field application of manure can be calculated, taking into account the hourly-based costs of labour and machine for a given piece of work/task. A model, CAESAR, (Computer simulation of the Ammonia Emission of Slurry application and incorporation on Arable land) developed for the analysis of manure application activities and efficiency (Huijsmans and De Mol, 1999) was used to simulate a range of manure application operations and calculate their associated costs.

In this chapter, the principal factors that effect machinery costs and time requirements for the field application of manure are considered. The costs of manure application in a number of European countries are calculated and the cost implications of adopting low emission manure spreading techniques are compared with broad spreading or splash plate application.

7.2 Assessment of the costs of manure application

Farm machinery costs are a substantial part of total farm costs and is an important element in the evaluation of alternative field operations, working methods and new machinery requirements. The cost is divided into the farm machinery operating cost (and indirect operating costs) and the labour cost for field operations.

7.2.1 Farm machinery operating costs

The direct farm machinery operating costs include costs for labour, ownership and machinery maintenance.

7.2.1.1 Labour

The cost of hired labour is the labour payment including charges for taxes, social and medical insurance, *etc.* Official tariffs can be applied for these costs. When farm labour is used, the opportunity cost must be charged (*i.e.* the price that can be obtained for alternative work). Even if alternative work is not available, the farmer is still willing to pay a price for free time.

7.2.1.2 Machine costs

The cost of durable assets like farm machinery can be divided into fixed and variable costs. Fixed costs are the costs that must be incurred, irrespective of whether or not the machine is used. They include depreciation, interest on capital, taxes, insurance and shelter. Variable costs are machine running costs (fuel, oil, repair and maintenance). The division between fixed costs and variable costs is not always clear. Depreciation depends on operating time for high-use farm machinery, while periodic maintenance is also required – even for infrequently used machinery. A similar division between fixed and variable costs is understood (described below) and has been applied in these calculations.

Fixed costs – depreciation: Depreciation is the loss in machine value due to age and use. It depends on the following factors:

- Wear with increasing age (to some extent independent of the intensity of machine use)
- Most wear and tear arising from machine use can be compensated for by timely repair. Over time, wear on major components will reduce the performance and reliability. Replacement of the main components, however, is often not economically viable
- Availability of new equipment with better performance
- Legislative requirements. Restrictions on operations, work practices or the use of machine types may be imposed, because of environmental damage or ethical considerations.

The period for which depreciation is budgeted is determined by the impact of these factors. Some of them are independent of use and determine the economic life of a machine. Estimation of the economic life of a machine is always difficult, because of the need to predict future developments. Machine use determines the 'effective' or 'practical' life, *i.e.* the maximum number of hours a machine can be used economically. The annual depreciation is determined by the economic life, the purchase price and the resale value of the machine as well as by the calculation method.

Fixed costs – interest, taxes, insurance, shelter: Interest must be calculated for the capital invested in the machinery. Taxes differ from country to country. In some countries, tractors and self-propelled machinery may have annual costs for registration plates. VAT and sales taxes are included in the purchase price. For tractors and self-propelled machinery, insurance is required to cover third party liability when driving on public roads. When storing machinery under cover, the costs of shelter (for the ground area required by the machinery) and the annual building costs per square metre can be calculated. For simplicity, a charge based on percentage of purchase price is used.

Variable costs – fuel and oil consumption, repair and maintenance: Fuel consumption during farm operations is a significant component of the operating costs and is an important element when calculating the energy efficiency of agricultural production. Fuel consumption depends on the energy requirements for the field operation, the efficiency of the transmission and tractive efficiency, the fuel efficiency of the power source and the type of fuel used. Engines require periodic replacement of crankcase oil. The cost of oil can be calculated as a percentage of the fuel costs (oil filters included). Repair and maintenance of machinery is necessary to ensure reliability and to guarantee work performance and quality. Repair and maintenance costs include costs for labour and replacement parts.

7.2.2 Time required for field application of manure

The time necessary for field application of manure depends on several operational variables such as field area and dimensions, working speed, working width, distance to manure store and work system (*e.g.* manure transport to the field in separate tanks). The CAESAR model enables calculation of the time requirement for the field application of manure by allocating the time spent on specific work components, *e.g.* spreading, turning, transport and loading.

Definitions, process description and parameters: In practice, manure spreaders are considered to apply the manure to the whole field until the complete area is covered. As soon as the spreader is empty, it is taken to the manure store for reloading. The manure store can be located nearby, or at some distance. To calculate the actual time for application, some activities and process parameters need to be defined.

In the model, a rectangular spreading area is considered (Figure 7.1). The manure application is performed in passes to and fro across the field with two successive passes forming a "round". The process of manure application is influenced by many factors. Technical factors include the dimensions of the covered area ("field"), the working forward speed, the working width, the manure application rate, and the payload of the spreader. The model considers three working methods:

I. Whole rounds – a new round (to and fro) is only started if there is enough manure in the tanker, otherwise the tanker is re filled
II. Whole passes – a new pass (to or fro) is only started if there is enough manure in the tanker, otherwise the tanker is re filled

III. Interrupted passes – application continues until the tanker is empty and, after reloading, the interrupted pass is continued from the same place and in the same direction as before the application stopped.

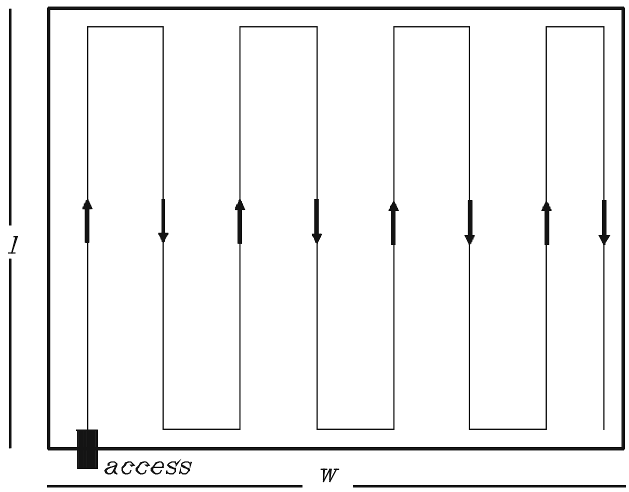


Figure 7.1 Layout of a plot (length *l* and width *w*) and the directions of passes of the manure spreader.

Simulation and calculation: The manure application time can be calculated by using the CAESAR model. Manure spreading (and incorporation) is a process interrupted by the specific and discrete requirements for turning, reloading or waiting. The model simulates discrete and continuous processes simultaneously. The model has an array of general input parameters and spreader or application machinery parameters (*cf.* Appendices 7.1 and 7.2).

The simulation begins with the full spreader at the field entrance or access point, which is designated to be at a corner of the plot (Figure 7.1). In the model, the spreader has a number of activities: working, waiting, driving across the field, transport to or from the manure store, reloading or turning. The main outputs generated by the model are the total time needed for application and the breakdown of this time over the different activities of the spreader.

7.2.3 Estimating European and standardised manure application costs and operating times

7.2.3.1 Data collection for cost calculations – European manure application costs

To provide an estimate of European manure application costs, CAESAR input data (machine operating costs and specific information on farm systems) for "typical" farm situations was collected for a number of the countries participating in the ALFAM project by use of questionnaires (Appendices 7.1 and 7.2).

The calculation of the machine operating costs was based on default values or on the specific input data provided by the individual countries as being representative of "typical" farms. Where the farmer does all the work himself and owns the machinery, the machine use in hours per year may be the outcome of the time required for the application of all the farm manure. When hiring a contractor, the labour hire charge has to be considered. The power requirement of the tractor is needed to determine fuel consumption.

The calculation of the operating time for the manure application process on the "typical" farms was based on the data received in response to the questionnaire. In order to determine the time required for spreading, it was assumed that:

- a known amount of manure was produced at the farm
- all the manure was applied on land within that farm
- not all fields may receive manure
- manure application may be carried out a few times per year on the same field
- manure may be applied separately to arable land and grassland.

7.2.3.2 Standardised calculation of costs

Calculations were conducted for a range of typical or standardised farms, in which the variation in farm situations and machine choices observed in Europe were simulated. These standardised calculations may be useful when attempting to explain the cost components of manure application and will allow systematic comparisons of cost components to be made. The standardised situations are representative of the range of farm scenarios received from participating countries (Appendices 7.1 and 7.2).

The machinery investment costs were assumed to be the same among the countries and were the average price for the machine type. The selected range of farm situations is given in Table .1. The machinery investment costs are given in Table 7.2. For each combination (farm situation), the costs of manure application were calculated, with costs expressed in terms of m^3 of manure applied. The costs of manure application for specific farm situations (or specific country situations) and changes in costs for manure application (*e.g.* when changing to low emission techniques) can then be derived.

Table 7.1. Farm situation variables used for the calculation of time and costs for the standardised situations (1920 possible combinations in total)

Variable		
Farm scale (manure production, m^3 per year)	500, 1000, 2000, 3000	
Application rate (m^3 ha^{-1})	10, 15, 20, 30, 40, 60	
Distance to storage (km)	0.5 and 2	
Average road speed (km h^{-1})	15 and 25	
Field size (ha)	2.4 and 5.4	
	Tanker size (m^3)	
	6	10
Application technique		
	working width (m)	
Broadcast spreading	12	12
Trailing hose	12	12
Trailing foot	4	5
Shallow injector	3	4
Arable land injector	3	4

Default values: labour costs 14 euro h^{-1}; working field speed 8 km h^{-1} (adjusted if pump capacity is not sufficient); idle field speed 10 km h^{-1}; maximum pump capacity 3 m^3 min^{-1}; interrupted passes; depreciation time tractor and implement 10 years, tanker 12 years; other default values see Appendix 7.1.

7.3 Results

7.3.1 European manure application costs

The time-related machine costs, spreading capacity and costs of applying the total farm manure output were calculated at individual farm level (Appendix 7.3). Machinery costs were on average 136 € h^{-1}, and varied from 43 to 285 € h^{-1}. Variations in machinery costs arise from differences in the components of the cost calculation and the time required for the manure application. The spreading capacity, expressed as the manure quantity that can be applied per hour, varied between farms, from 12 to 55 m^3 h^{-1}, with an average of 28 m^3 h^{-1}. Differences in spreading capacity may be due to the choice and capacity of the application technique, the target application rate and field size and location (especially the distance from the manure store).

Table 7.2. Investment costs (in €) used for a tanker and application techniques (derived from Gakeer, 1998; upper part of the table); also of the tractors used for each application technique including tractor power (kW) (lower part of the table) in standardised calculations

Application technique	Tanker size (m^3) 6		Tanker size (m^3) 10	
Slurry tanker	11,000		16,000	
Trailing hose	10,500		10,500	
Trailing shoe	6,800		9,000	
Shallow injector	7,700		11,500	
Arable land injector	5,500		9,500	
Tractor used *				
Broadcast spreading	43,500	(65)	56,500	(85)
Trailing hose	43,500	(65)	56,500	(85)
Trailing shoe	43,500	(65)	56,500	(85)
Shallow injector	56,500	(85)	66,250	(100)
Arable land injector	56,500	(85)	66,250	(100)

* The investment costs of the tractor depend on the power requirement needed to pull and operate the application technique.

The manure application costs were calculated at farm level, taking into account the machine costs and the annual operating time for manure application. These costs were on average 5.4 € m^{-3} manure applied, and varying from 1.6 to 13 € m^{-3} (Figure 7.2 and Appendix 7.3). The calculated costs provide a rough estimate of the range of European manure application costs. Farm situations (particularly manure management aspects), costs and choice of machinery have a large impact on the calculated costs. It is difficult to analyse the major cost variables, due to the large variation in the data and the small sample size within this study. The observed variation between the costs of manure application in the participating countries is difficult to explain. The overall costs of manure application clearly depend on a range of factors, including farm circumstances, which may differ between and within both regions and countries. Variation of the costs within a country may be at least as great as the variation between countries.

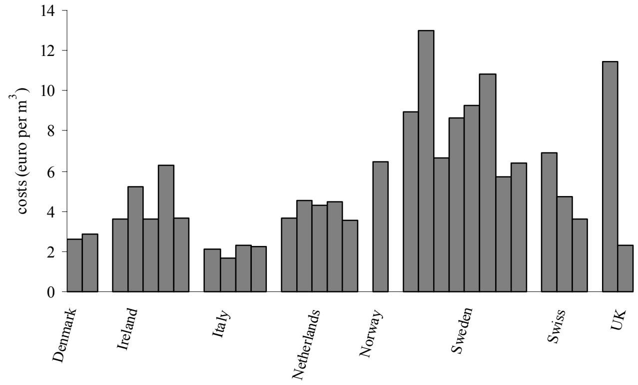


Figure 7. 2. Observed variation in manure application costs throughout Europe.

7.3.2 Standardised costs

The costs of manure application were calculated for the standardised farm situations. A summary is given of the average costs of manure application by various techniques at farms with a range of total annual manure production (Table 7.3 and Figure 7.3).

Table 7.3 Costs of manure application by various techniques for farms with a manure production of 500-3000 m^3 per year (€ m^{-3} applied)

Manure production	Application technique				
(m^3 year^{-1})	Broadcast spreading	Trailing hose	Trailing foot	Shallow injector	Arable land injector
500	8.46	14.04	13.06	14.53	13.41
1,000	5.07	7.86	7.58	8.60	8.03
2,000	3.38	4.78	4.84	5.63	5.35
3.000	2.82	3.75	3.92	4.64	4.45

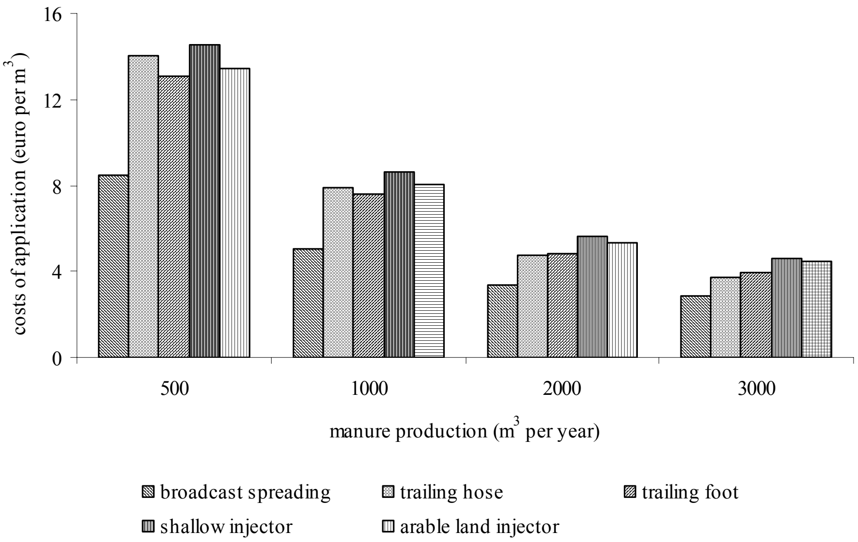


Figure 7. 3 Manure application costs for various application techniques for farms with manure production of between 500 and 3000 m^3 year^{-1} (€ m^{-3} applied).

On average, for the different farm sizes (farms with a range of manure production of 1000 to 3000 m^3 year^{-1}), manure application by trailing hose, trailing foot, shallow injector and arable land injector costs *ca* 2 € m^{-3} more than broadcast spreading. The cost difference between broadcast spreading and the other application techniques decreases with increasing farm size, *e.g.* with a manure production of 3000 m^3 year^{-1}, the cost of broadcast application is *ca.* 1.4 € m^{-3} less than other techniques. The differences in costs are highest on small, less intensive farms (producing up to 500 m^3 of manure year^{-1}).

The application rate does not affect the costs m^{-3} applied when applying manure by broadcast or trailing hose systems. The application rate is adjusted by the pump capacity of the manure pump in all situations. When the maximum capacity of the manure pump is reached, an increase in application rate will be attainable by decreasing the forward working speed in the field. When applying manure by use of the trailing foot or injection techniques, the costs m^{-3} will not change for application rates above 25 m^3 ha^{-1}. Differences in the costs between broadcast spreading and trailing hose and between shallow injector and arable land injector can be explained in full by their differences in investment costs (working widths, operating forward speeds, taken as the same).

The effect of field size on the average costs of manure application is highlighted in Table 7.4. The field size does not greatly effect costs. The lowest costs were achieved with the field located near the store (0.5 km), a high travelling speed (25 km h^{-1}), and a high total annual manure spreading requirement (3000 m^3). This was the case for all application techniques. Minimum costs were achieved at an application rate of 20 m^3 ha-1 for broadcast spreading and trailing hose and of 60 m^3 ha^{-1} for the other application techniques.

Table 7.4 The cost of manure application for the various application techniques for different field sizes (€ m^{-3})

Application technique	Field size					
	2.4 ha			5.4 ha		
	average	min	max	average	min	max
Broadcast spreading	4.84	2.05	10.13	5.03	2.20	10.43
Trailing hose	7.51	2.98	15.71	7.70	3.13	16.01
Trailing shoe	7.29	2.73	15.72	7.41	2.84	16.00
Shallow injector	8.27	3.12	17.93	8.43	3.24	18.14
Arable land injector	7.73	2.93	16.87	7.89	3.05	17.08

The effect of the distance to the store, slurry tanker size and road speed for a range of application rates is shown in Table 7.5

Table 7.5 Costs of manure application for varying distances to the store, tank sizes, road speeds and application rates (€ m^{-3}; averaged for the various application techniques)

Application rate (m^3 ha^{-1})	Distance (km)		Tanker size (m^3)		Road speed (km h^{-1})	
	0.5	2	6	10	15	25
10	7.36	8.21	7.18	8.39	7.96	7.61
15	6.97	7.88	6.83	8.02	7.61	7.24
20	6.72	7.62	6.58	7.76	7.36	6.98
30	6.58	7.53	6.41	7.70	7.25	6.86
40	6.48	7.42	6.31	7.58	7.14	6.75
60	6.41	7.36	6.23	7.54	7.08	6.68

The minimum cost was estimated at 4.43 € m^{-3} (0.5 km distance to store, broadcast spreading, application rate 20 m^3 ha^{-1}) and the maximum at 9.81 € m^{-3} (2 km distance to store, shallow injector and application rate 10 m^3 ha^{-1}). The minimum estimated costs were 4.43 € m^{-3} (6 m^3 tank size, broadcast spreading and application rate 20 m^3 ha^{-1}) and the maximum 10.05 € m^{-3} (10 m^3 tank size, shallow injector and application rate 10 m^3 ha^{-1}) for tanker size. For road speed the minimum estimated costs are 4.67 € m^{-3} (25 km h^{-1}, broadcast spreading and application rate 20 m^3 ha^{-1}) and the maximum 9.53 € m^{-1} (road speed 15 km h^{-1}, shallow injector and application rate 10 m^3 ha^{-1}).

Besides application of the manure to the field, the manure spreader is also used for transport to and from the store and for loading the tank. An overview is given of the proportion of total spreading time actually spent on applying manure in the field (Table 7.6). It is clear that manure spreaders spend less than 50% of their operating time on working in the field. The remaining operating time is spent on other aspects such as transport and loading the tanker, which are all independent of the application technique.

Table 7.6. Allocation of manure spreader time to actual manure application in the field (as % of the total time for manure application task)

Application rate (m^3 ha^{-1})	Application technique				
	Broadcast spreading	Trailing hose	Trailing shoe	Shallow injector	Arable land injector
10	25	25	41	46	46
15	18	18	33	37	37
20	15	15	28	33	33
30	14	14	21	24	25
40	14	14	17	20	20
60	14	14	14	15	15
Average	17	17	26	29	29

7.4 Summary and conclusions

The economics involved with manure handling and spreading are of fundamental importance in encouraging the improved recycling and efficient utilisation of manure. A model was used to calculate the costs of manure application by different techniques and for a range of European farms using country specific input data. Application costs varied from 1.65 to 13 € m^{-3} of manure applied at a mean cost of 5.4 € m^{-3}. Variations in costs may be attributable to country-specific differences, as well as to differences between farms within a country. Analysing the major factors affecting costs was difficult, due to the large variations in the data both within and between countries combined with the small sample size.

A further cost analysis was undertaken on the basis of a range of "typical" farm situations to allow systematic comparison of the cost components of manure application. The results indicated that the costs of manure application by trailing hose, trailing foot, shallow injector and arable land injector are *ca.* 2 € m^{-3} greater than the cost of broadcast spreading for farms with a manure production of 1000-3000 m^3 per year. The cost differential between broadcast spreading and the other application techniques decreases with farm size. For example, with a manure production of 3,000 m^3 per year, the costs of low emission application techniques are *ca.* 1.4 € m^{-3} higher than the broadcast costs. Conversely, the differences in costs are highest on small, less intensive farms (producing up to 500 m^3 of manure per year). Manure application by a contractor may be less expensive for the farmer, because the contractor may be able to offset the costs of the equipment against a much greater total number of working hours per year. This will almost certainly be the case on many small-sized farms.

APPENDIX (Chapter 7)

Appendix 7.1 Questionnaire completed by ALFAM participants to provide the parameters required for determining the operating costs of machinery per implement used

	Cost Item	Implement	Tractor, tanker, manure spreader, incorporator
A	Replacement cost	€	
B	Residual value (% of A)	%	Default 10% of A
D	Depreciation time	Years	
E	Machine use	Hours/year	
H	Interest (% of average A + B)	%	Default 6.5% of average A + B
I	Repairs by others (% of A)	%	Default 5% of A
J	Own repair (% of I)	%	Default 60% of I
K	Shelter (% of A)	%	Default 2% of A
L	Insurance (% of A)	%	Default 1% of A
M	General costs (% of A)	%	Default 3% of A
Q	Labour costs (hourly)	€	

Appendix 7.2 Questionnaire completed by ALFAM participants to provide the parameters required for determining the operating time for field application of manure

Parameter name	Parameter value
Working method	□ whole rounds □ whole passes □ interrupted passes
Area arable land	ha
Area grassland	ha
Manure production or applied per year	m^3
Length of field	m
Width of field (typical average)	m
Manure application rate per application	$m^3\ ha^{-1}$
Number of times manure is applied on a field	
Idle travelling speed of the spreader on the field	$km\ h^{-1}$
Working speed of the spreader (or injector)	$km\ h^{-1}$
Effective working width of the spreader (or injector)	m
Headland turning time for the spreader after each pass	s
Payload of the spreader	m^3
Travelling speed of the spreader on the road	$km\ h^{-1}$
Average distance to manure store from field access	km
Time for handling and turning before and after reloading	min
Loading (pump) capacity of the spreader	$m^3\ min^{-1}$

Where the manure is applied to arable land and is incorporated to reduce ammonia losses, the additional time for the incorporation is accounted for. The following parameters are required for this calculation.

Parameter name	Parameter value
Incorporator type (plough, cultivator, discs, rotavator)	
Working speed of the incorporator	km h^{-1}
Effective working width of the incorporator	m
Headland turning time for the incorporator	s

Appendix 7.3 The quantities of manure requiring annual spreading annually and the associated machinery and manure spreading costs for the individual farm case studies in each participating country

Country	Manure applied per year (m^3)	Spreading capacity (m^3 h^{-1})	Machine costs per hour (euro h^{-1})	Machine costs per year (euro $year^{-1}$)	Costs of manure applied (euro m^{-3})	Application technique *
Denmark	3,091	37.2	97.80	8,118	2.63	Broadcast +
	2,798	35.6	100.95	7,925	2.83	Broadcast +
Ireland	946	27.8	100.21	3,407	3.60	
	1,728	16.3	85.13	9,024	5.22	
	2,247	26.3	94.44	8,074	3.59	
	964	27.9	175.44	6,053	6.28	
	1,555	31.1	114.84	5,742	3.69	
Italy	12,600	31.3	65.65	26,390	2.09	
	24,500	34.3	56.73	40,502	1.65	
	6,260	38.1	88.28	14,522	2.32	
	8,075	38.3	85.86	18,116	2.24	
Netherlands	1,534	54.8	206.62	5,785	3.77	Shallow injection
	1,775	32.0	146.89	8,152	4.59	Shallow injection
	1,887	36.6	153.60	7,911	4.19	Shallow injection
	1,330	47.5	209.58	5,868	4.41	Shallow injection
	2,700	32.7	113.26	9,344	3.46	Trailing foot
Norway	600	16.4	106.30	3,880	6.47	
Sweden	800	21.9	196.82	7,184	8.98	
	800	21.9	285.43	10,418	13.02	Trailing hose
	1,600	17.9	118.51	10,606	6.63	
	1,600	17.9	154.64	13,841	8.65	Trailing hose
	1,600	24.1	222.35	14,786	9.24	Trailing hose
	1,600	23.7	255.96	17,277	10.80	Shallow injection
	4,800	19.6	112.52	27,510	5.73	Trailing hose
	4,800	18.9	120.41	30,645	6.38	Shallow injection
Switzerland	988	21.0	145.84	6,854	6.94	
	1,248	20.5	96.68	5,898	4.73	
	3,045	12.1	43.42	10,920	3.59	
UK	290	23.2	266.01	3,325	11.47	
	5,460	36.0	82.46	12,493	2.29	

* **Application technique:** In most countries manure is applied by a broadcast spreader (splash plate). For farms in the Netherlands and Sweden cost calculations included farms where manure is applied by trailing foot, trailing hose or shallow injector. The broadcast spreading on arable land in Denmark is followed by incorporation (broadcast +); no additional costs are taken into account for this tillage operation.

8. Ammonia losses from field applied animal manure – Research and Development Requirements

8.1 Introduction

New EU initiatives such as the Directive on national emission ceilings (NECD) and the Directive on integrated pollution prevention and control (IPPC Directive, 96/61/EC), together with the UNECE Gothenburg protocol, will require substantial reductions in ammonia emissions from some member states. Animal husbandry is the major source of ammonia emission within Europe, so significant changes in this area are inevitable.

In Chapter 6 it was noted that between 0 and 100% of the total ammoniacal nitrogen in the manure might be lost as ammonia after application to land. The factors contributing to this have been addressed, some in great detail, in a number of the preceding chapters. These include the manure type, the application technique and other factors like soil properties, and weather conditions (*cf.* Chapter 4). Changes in land spreading technology usually have the highest and the most cost-effective potential for emission abatement. Cost increases over broadcast spreading with splash plate were shown to range from 2 € m^{-3} on large farms (>2000 m^3 of manure to be managed annually) to more than of 14 € m^{-3} on smaller farms (<500 ^{3} of manure to be managed annually), as reported in Chapter 7. Changes in the field application and cultivation techniques to reduce emissions are far less expensive and much easier to implement than corresponding changes in the structure and management of animal houses and manure storage facilities. Field application is therefore the logical place to continue looking for cost-effective methods that will allow the member states to fulfil their legal obligations.

It is also expected that under the NECD, member states will be obliged to report emission inventories on a regular basis. Emissions from field-applied manure are a significant source of ammonia, accounting for about one third of European ammonia emissions (ECETOC, 1994), so it is important that emissions from this source are accurately represented.

The international agreements mentioned above will mean an increasing focus on ammonia emissions in many European countries in the coming years. The reduction of ammonia emissions that will be required and the measures that are appropriate to achieve these reductions will vary between countries. This chapter addresses a number of topics that policymakers, advisors and researchers will have to consider when meeting the challenge posed by these reductions, and recommendations should be made as to what further action should be taken, by using the results and conclusions of the previous chapters, whenever appropriate.

8.2 Research and development requirements

8.2.1 Manure application techniques

A number of factors must be considered to identify the requirements for research and development regarding the field application of manure. These factors include the structure of animal husbandry in different regions in Europe, the types and amounts of manure produced, the main crops cultivated in these regions, climatic conditions, landscape types and farm management systems. It is possible to identify six main regional types (Table 8.1). Depending on the regional type, the demand for new techniques and procedures may be quite different.

Table 8.1 The typical characteristics of six broad animal production regions or agro-climatic regions within Europe based on climate and animal enterprise

No.	Typical characteristics	Examples
1	Largely grassland (more than half of the agricultural area), cool and wet climate with regular precipitation throughout the year and a high livestock density, mainly with liquid manure systems.	Parts of Netherlands, Brittany, South West and West England, parts of Ireland, the northern foothills of the Alps, North West Germany,
2	Mainly dairy farms, small enterprises, mainly grassland, mainly producing solid manure	Low mountain range regions, *e.g.* South Germany
3	Temperate climate, arable land often cultivated with winter crops	Denmark, Germany, East England, North France, Parts of Netherlands, South Sweden
4	Continental climate, mainly low livestock density, large enterprises, mainly arable land	East Germany, Hungary, Slovenia, Czech Republic, Slovakia
5	Mediterranean climate, dry; small and medium sized enterprises	South France, most of Spain, Po Valley (Italy)
6	Cool climate, mainly grassland, low livestock density.	Parts of Scandinavia, Wales, Scotland, higher farmed areas in Alps, Northern Spain.

Cattle manure remains the most significant manure in Europe in terms of the amount of nitrogen produced (Table 8.2). Pig manure can be of greater importance in some regions, because of a localised high density. Slurry remains the dominant manure type in terms of the volume produced. However, small farms often produce farmyard manure, so a more even distribution would be apparent if expressed in terms of the number of farms producing a particular type of manure.

Table 8.2 Amount and type of animal manure in Europe

Manure type	Production*	Distribution**		
	Mt N/year	Slurry	Solid manure	Liquid manure (urine)
Cattle	6.7	62	32	6
Pig	1.2	86	12	2
Sheep	2.4	NA	NA	NA
Goats	0.4	NA	NA	NA
Chickens	0.5	NA	NA	NA
Total	11.2			

* estimated from agricultural statistics and Danish animal excretion standards
** estimated from replies from ALFAM participants
NA = not available.

8.2.2 Emission reduction techniques

The choice of low emission manure application techniques that is open to farmers can be limited by a number of factors:

- Regulations relating to nitrate leaching or surface runoff may restrict the timing of applications, the amount of manure applied and the land to which manure may be applied (*e.g.* avoiding land that is steeply sloping or close to sensitive ecosystems).
- The slope of the land, its stoniness or the presence of a crop may prevent the use of some techniques.
- The use of large or heavy vehicles may not be possible, due to restricted access, poor traction on wet soils or the risk of damaging the soil structure.

Since broadcast spreading with splash plate is still the most common manure application technique in most countries, the first step for the reduction of the emissions would be to investigate the applicability of existing low emission techniques within the different agro-climatic regions of Europe. In this connection, costs and cost-effectiveness are major issues, although it is also important to assess the extent to which some of the practical limitations mentioned above limit their use. This might point to further needs for research and development. For example, there is a need for development and testing of application techniques that can be used under difficult conditions such as on stony, very dry or compacted soils, or the application of dilute slurries at a high application rates.

An initial assessment of the specific problems related to ammonia emission abatement in the agro-climatic regions of Europe is given in Table 8.3.

Table 8.3 An evaluation of the need for new ammonia emission abatement technologies for land spreading of manure by agro-climatic region (Definition of these regions are given in Table 8.1)

Region	Most common techniques	Problems arising	Need for new abatement techniques
1	Mainly broadcast spread slurry, some band spreaders	Application rate limited in autumn, high amounts of cattle slurry to be applied in spring and summer	High for manuring of growing crops in spring
2	Broadcast solid manure spreaders	Considerable emission of ammonia from solid manures	High for solid manures
3	Mainly broadcast spread slurry, some band spreaders	Application rate limited in autumn, high amounts of cattle and pig slurry to be applied in spring	High for the manuring of growing crops in spring
4	Mainly broadcast spread slurry, some simultaneous incorporation.	–	Small
5	Mainly broadcast spread slurry, some band spreaders	High ammonia losses due to high temperatures	High for manuring of growing crops in spring
6	Mainly broadcast spread slurry, some band spreaders	Short growing season, short time period for spreading in spring	Medium

In areas with a low livestock density (<0.5 dairy cow equivalents per ha) and high availability of arable land for application, the demand for new techniques is currently low. Even taking into account the need to avoid applications to sandy soils in the autumn (due to the risk of nitrate leaching) or clay soils in the spring (due to the risk of soil damage from heavy machinery), sufficient areas should be available for application of manure by using existing low-emission techniques.

In regions with higher livestock densities (>0.5 dairy cow equivalents per ha), the amount of manure produced is large in comparison to the demand for crop fertilisation. Furthermore, the application of manure is often undesirable or prohibited in winter and early spring for environmental reasons. In regions with predominantly arable cropping, the presence of growing crops limits the application in late spring/early summer. This means that time available for application is restricted. The late spring/early summer would be very suitable for manuring because of the high nutrient demand of the crops in the growing season, if application techniques were available to avoid damage to the crops. This suggests an urgent need to develop such application techniques, that manure can be applied further into the growing season. These techniques have to confer both a sufficient and a predictable crop response to the manure nutrients and facilitate a reliable emission reduction of ammonia and, if possible, other nitrogen gases of environmental importance.

Currently, low emission application techniques are not available for solid manure. On arable land, this is not a serious problem, as emissions can be reduced by rapid incorporation of the manure into the soil. But it is a serious problem where grassland is the predominant crop. Grassland is predominant on many cattle farms, and around one third of all cattle manure is still produced in the solid form. Although animal housing systems producing solid manure have been in decline over the last 30 years, this trend may be stopped or reversed if consumer concern over animal welfare favours the reintroduction of straw-based systems. There is no indication of the development of low emission techniques for solid manure. Without wishing to completely dismiss the chance that low emission application techniques can be developed for solid manures, it seems likely that the most promising approach would be to consider some of the up-stream technologies described in (*cf.* chapter 6).

Finally, it should be emphasised that losses of ammonia from field-applied manure cannot be viewed separately. For example, transformations of nitrogen in animal housing and manure storage will affect losses in the field. Likewise, ammonia emission in the field will affect the amount of nitrogen that could be subsequently lost as nitrate or nitrous oxide. Consequently, there is a need for more integrated studies, including other components of the manure handling chain and the losses of other environmentally polluting compounds.

8.2.3 Ammonia measurement in experiments

The report of the task group on measurement techniques described the theory and practicalities of use of the three most commonly employed methods, viz. micrometeorological mass balance, wind tunnels and equilibrium concentration technique, together with a discussion of the advantages and disadvantages of each. The group concluded that the usefulness of emission data, particularly for model development, could be increased by making additional measurements to characterise the manure, crop, soil and environmental conditions at the time of manure application, and a list of suggested additional parameters to be measured is given.

The analysis of data collated in the ALFAM database found that there were significant differences in the calculated emission between the different methods, which complicates cross-comparison of the results. There is a need for experiments of cross-calibration between methods to enable cross-comparison of research results from different experiments in which different measurement methods are adopted.

8.2.4 Economic analysis
An economic analysis of the low emission application techniques showed that the likely cost of implementation of the different emission reduction techniques varied considerably, with as much variation within a country as between countries. This was because the costs mainly varied according to farm size and spatial distribution of fields rather than country per *se* with costs highest for the small farms. A Europe-wide analysis was not possible with the resources and input data available. Such an analysis requires quality data from all European countries, and while some of the data are available from EUROSTAT, additional data would need to be collected by surveying. This analysis could usefully be extended to examine some of the options available to reduce the cost of low emission technology to farmers, in general, and small farmers, in particular. Co-operative investment or the use of agricultural contractors are two options worthy of investigation, as these have already proven successful in some countries.

8.2.5 Technology transfer
Farmers still do not adequately integrate animal manure nutrients in their fertiliser planning, with the result that nutrients are often supplied in excess, leading to pollution of surface and ground waters. One reason why farmers do not fully allow for the nutrients supplied in animal manure is uncertainty over its fertiliser value. Variations in ammonia volatilisation account for much of this variation as the majority of the plant-available nitrogen in manure is in the ammoniacal form. To provide farmers with an estimate of the amount of ammonium lost *via* volatilisation would improve farmers' willingness to take account of manure nitrogen in fertiliser planning. This information could be provided by using a paper-based decision support tool, but it could be made more robust if provided *via* the Internet, as this would allow the underlying model to account for a larger number of management, climatic and soil variables. The experience from the data collation and analysis exercise within ALFAM (*Cf.* Chapter 4) has shown the benefits to be achieved from gathering the data collected in different countries. Given our knowledge of the processes of controlling ammonia volatilisation from field applied animal manures (refer to conceptual model here), we consider that an emission model could be produced that could be tailored to the widely varying climates and farming systems within Europe.

8.2.6 National emission inventories

The abatement of ammonia is of increasing importance in political strategies to combat acidification and eutrophication. In order to reduce the environmental impact of ammonia, legislation exists or is in preparation at an EU level (Draft of the EU Directive on National Emission Ceilings) and also at a UN level (Convention on Long-range Transboundary Air Pollution) concerning ammonia emissions. Within the scope of these treaties, the assessment of emissions is normally based on emission factors. The so-called ”simpler methodology” applies an average emission factor per animal, while the “detailed methodology” breaks the emissions down according to source (animal housing, storage, application). To be truly representative of the emissions from a particular country, these emission factors need to take into account the differing farming systems, application methods, variations in soil characteristics and climates.

For a more detailed calculation of emissions and abatement potentials an even more detailed methodology is required, including detailed emission factors for different types of application techniques, as well as manure types, weather and soil conditions. If differences in emission factors are to reflect real differences in emission rather than in the method of calculation, a standardised method for estimating the detailed emission factors should be developed. Alternatively, the emission factors could be replaced by a standardised model for ammonia emission from field applied manure. For field applied manure, these developments should utilise the results from field measurements, such as those gathered in the ALFAM database, supplemented with additional measurements from Eastern European continental climates and the Mediterranean regions of Europe.

Even if the emission factors are scientifically justified, the accuracy of the calculated emission will also depend on the quality of the activity data used as input. Reliable statistics are required, for example, to describe the distribution of manure production between solid and liquid manure systems, manure application technique and timing, cultivation techniques and soil/climatic factors. Currently, this type of detailed information is lacking for much of Europe.

8.3. Recommendations

The following research initiatives are recommended:

1. There is evidence that some technologies that have been shown to work well in a limited number of research projects are less effective when used under more extensive and less controlled conditions on commercial farms. These abatement technologies need to be tested under commercial farming conditions. An assessment of the practical effectiveness of abatement technologies could be used to improve both emission inventories and the recommendations given to farmers.
2. The low emission manure application techniques that are currently available should be evaluated to assess what is technically feasible under differing country (or region) specific conditions. This investigation needs also to assess the likely collateral effects of the different application methods. For example, on soil structure *via* compaction and on the

losses of other environmentally important compounds such as nitrate, nitrous oxide, methane and phosphate, and on pathogen survival.

3. There has been relatively little research into the emissions from field-applied solid manures in the past. Although more research is in progress, this is a more complex issue than emission from applied slurry. In addition, the decrease in housing systems producing solid manure seems likely to stop or be reversed as farmers respond to consumers' concerns about animal welfare. A continued research effort will therefore be necessary, and more attention should be paid to abatement technologies for solid manures.
4. There is an urgent need to develop low emission application techniques, so that manure can be utilised most beneficially on more mature crops without causing damage. This would extend the period during which manure could be utilised. These techniques need to offer a sufficient and predictable crop response from the manure with a reliable reduction of the emission of ammonia and other nitrogen gases.
5. A Europe-wide analysis of the likely cost of implementation of the different emission reduction techniques should be undertaken, with additional data collected by surveying. This analysis should consider the options available to reduce the cost of low emission technology to farmers, in general, and small farmers, in particular. Such an analysis would provide a useful supplement to the more wide-ranging analyses conducted by IIASA (http://www.iiasa.ac.at/~rains/ciam.html).
6. Although a number of ammonia measurement techniques are available, additional development is needed to enable non-obtrusive measurement of emission from small plots. There is also a need for cross-calibration experiments between methods to enable cross-comparison of results from different experiments in which different measurement methods are adopted. Researchers working on ammonia emission from field-applied animal manures should be encouraged to use the standard methods identified within the ALFAM project and make the full range of additional measurements noted by the task group on measurement techniques.
7. The knowledge already gained by scientists concerning ammonia emission from manures needs to be transferred more effectively to farmers. An outline of how this could be achieved is given in *Chapter 2*. Experience, *e.g.* in The Netherlands shows that this should not only be done *via* booklets or brochures, but much more *via* practical demonstrations of techniques and especially *via* fertiliser management demonstrations. Demonstrations should preferably offer practical solutions how to best use manure nitrogen within the whole nutrient cycle. Decision support systems including predictions on ammonia losses, availability of mineralised organic soil and manure nitrogen should be developed.
8. There is a need to construct a decision support system to allow policy makers to assess the cost and impact on ammonia emission of different abatement scenarios. An outline of such a tool is described in *Chapter 2*.
9. There is a lack of statistical information on the spatial variation in manure handling systems and management practices throughout the EU and neighbouring countries. This con-

strains the construction of reliable emission inventories and the ability of policymakers to assess the cost effectiveness of abatement measures.

10. Comparisons of the trends in estimated emission and measured ammonia concentrations in the air in NL and UK suggest that current emission estimates are too low (the so-called 'ammonia gap'). The origin of this gap needs to be investigated. The reliability of emission inventories would be increased if a standardised method for estimating the detailed emission factors were developed or if emission factors were replaced by a standardised model for ammonia emission from field applied manure. Given the uncertainty over the reliability of emission inventories, we would recommend that atmospheric ammonia monitoring systems be established in all European countries, as a check on the accuracy of emission estimates.
11. Losses of ammonia from field-applied manure cannot be viewed in isolation as transformations of nitrogen in animal housing and storage affect losses after field application. There is a need for more integrated studies. At the farm scale, this needs to focus on the consequences of changes in management practice early in the manure handling system (animal housing and storage) and in animal feeding on subsequent losses after field application. At the field scale, there is a need to study the consequences of changes in manure application practice on other gases (*e.g.* methane, nitrous oxide) and on leaching and crop uptake.

References

Aarnink, A.J.A., Canh, T.T. & Mroz, Z., 1997: Reduction of ammonia volatilization by housing and feeding in fattening piggeries. In: Voermans J.A.M.& Monteny G.J. (Eds.): "Ammonia and odour emissions from animal production facilities", *Proc. International Symposium, Vinkeloord*, NL, 6-10 October, 1997, 283-291.

Amon, B., Boxberger, J., Amon, Th., Gronauer, A., Depta, G., Neser, S. & Schafer, K., 1997. Emission from agrarian sources: FTIR measurement techniques with white cell, large chamber or open-path. In: Ammonia and odour control from animal production facilities (eds Voermans, J. A. M. & Monteney, G. –J.), Vinkleloord, NL, 161-167.

Amon, M., Dobeic, M., Misselbrook, T.H., Pain, B.F., Phillips, V.R. & Sneath, R.W., 1995: A farm scale study on the use of De-Odorase(R) for reducing odour and ammonia emissions from intensive fattening piggeries. *Bioresource Technology*, 51, 163-169.

Beauchamp, *E.G.*, Kidd, G.E. & Thurtell, B., 1978. Ammonia volatilization from sewage sludge applied in the field. *Journal of Environmental Quality,* 7, 141-146.

Bless, H.G., Beinhauer, R. & Stattelmacher, B., 1991. Ammonia emission from cattle slurry applied to wheat stubble in North Germany. *Journal of Agricultural Science*, Cambridge 117, 225-231.

Braschkat, J., Mannheim, T. & Marschner, H., 1997. Estimation of ammonia losses after application of liquid cattle manure on grassland. *Zeitschrift für Pflanzenernahrung und Bodenkunde* 160, 117-123.

Brundin, S.G. & Rodhe, L.K.K., 1994. Comparisons of Manure Handling Systems under Swedish Conditions. *J. Agric. Engng. Res*. 58, 181-190.

Brunke, R., Alvo, P., Schuepp, P. & Gordon R., 1988. Effect of meteorological parameters on ammonia loss from manure in the field. *Journal of Environmental Quality,* 17, 431-436.

Burton, C.H. & Martinez, J. (Eds.), 1996. Animal manures and environments in Europe. *Ingénieries: Eau-agriculture-territoires*. Special issue, 100 pp.

Businger, J.A. & Oncley, S.P., 1990. Flux measurement with conditional sampling. *Journal of Atmospheric Ocean Technology,* 7, 349-352.

Bussink D.W., 1994: Relationship between ammonia volatilization and nitrogen fertilizer application rate, intake and excretion of herbage nitrogen by cattle on grazed swards. *Fertilizer Research*, 38, 111-121.

Bussink, D.W., Huijsmans, J.F.M. & Ketelaars, J.J.M.H., 1994. Ammonia volatilization from nitric-acid-treated cattle slurry surface applied to grassland. *Netherland Journal of Agricultural Science* 42, 293-309.

Canh, T.T., Aarnink, A.J.A., Schutte, J.B., Sutton, A., Langhout, D.J. & Verstegen, M.W.A., 1998a: Dietary protein affects nitrogen excretion and ammonia emission from slurry of growing-finishing pigs. *Livestock Production Science*, 56, 181-191.

Canh, T.T., Aarnink, A.J.A., Mroz, Z., Jongbloed, A.W., Schrama, J.W. & Verstegen, M.W.A., 1998b: Influence of electrolyte balance and acidifying calcium salts in the diet of growing-finishing pigs on urinary pH, slurry pH and ammonia volatilisation from slurry. *Livestock Production Science*, 56, 1-13.

Canh, T.T., Sutton, A.L., Aarnink, A.J.A., Verstegen, M.W.A., Schrama, J.W. & Bakker, G.C.M., 1998c: Dietary carbohydrates alter the faecal composition and pH and the ammonia emission from slurry of growing pigs. *Journal of Animal Science*, 76, 1887-1895.

Canh, T.T., Aarnink, A.J.A., Schrama, J.W. & Haaksma, J., 1997: Ammonia emission from pig houses affected by pressed sugar beet pulp silage in the diet of growing-finishing pigs. In: Voermans, J.A.M. and Monteny, G.J. (Eds): "Ammonia and odour emissions from animal production facilities", *Proc. International Symposium, Vinkeloord*, NL, 6-10 October 1997, 273-281.

Castillo, A.R., Kebreab, E., Beaver, D.E. & France, J., 2000: A review of efficiency of nitrogen utilisation in lactating dairy cows and its relationship with environmental pollution. *Journal of Animal and Feed Sciences*, 9, 1-32

Chadwick, D.R., 1997. Nitrous oxide and ammonia emissions from grassland following applications of slurry: potential abatement practices. In *Gaseous Nitrogen Emissions from Grasslands*, (eds. S.C. Jarvis & B.F. Pain), CAB International, 257-264.

Denmead, O.T., Harper, L.A., Freney, J.R., Griffith, D.W.T., Leuning, R. & Sharpe, R.R., 1998. A mass balance method for non-intrusive measurements of surface-air trace gas exchange. *Atmospheric Environment* 32, 3679-3688.

Denmead, O. T., 1983. Micrometeorological methods for measuring gaseous losses of nitrogen in the field. In *Gaseous Loss of Nitrogen from Plant-Soil Systems* (eds. J. R. Freney and J. R. Simpson), Martinus Nijhoff/Dr W Junk, The Hague, 133-158.

Denmead, O.T., Simpson, J.R. & Freney, J.R., 1977. A direct field measurement of ammonia emission after injection of anhydrous ammonia. Soil Science Society of America Journal 41, 1001-1004.

Donovan, W.C. & Logan, T.J., 1983. Factors affecting ammonia volatilization from sewage sludge applied to soil in a laboratory study. *Journal of Environmental Quality* 12, 584-590.

Döhler, H., 1991. Laboratory and field experiments for estimation ammonia losses from pig and cattle slurry following application, 132-140. In *VC Nielsen et al. (eds). Ammonia and odour emissions from livestock production*. Elsevier Applied Science Publ., New York.

Dosch, P. & Gutser, R., 1996: Reducing N losses (NH_3, N_2O, N_2) and immobilization from slurry through optimized application techniques. *Fertilizer Research* 43(1-3), 165-171.

Dourmad, J.Y., Guingand, N., Latimier, P. & Sève, B., 1999a: Nitrogen and phosphorus consumption, utilisation and losses in pig production: France. *Livestock Production Science*, 58, 199-211.

Dourmad, J.Y., Sève, B., Latimier, P., Boisen, S., Fernandez, J., van der Peet-Schwering, C. & Jongbloed, A.W., 1999b: Nitrogen consumption, utilisation and losses in pig production in France, The Netherlands and Denmark. *Livestock Production Science*, 58, 261-264.

ECETOC, 1994. Ammonia Emissions to Air in Western Europe. European Center for Ecotoxicology and Toxicology of Chemicals. Brussels, 196 pp.

Elwinger, K. & Svensson, L., 1996: Effect of dietary protein content, litter and drinker type on ammonia emission from broiler houses. *Journal of Agricultural Engineering Research*, 64, 197-208.

Ferguson, N.S., Gates, R.S., Taraba, J.L., Cantor, A.H., Pescatore, A.J., Straw, H.L., Ford, M.J. & Burnham, D.J., 1998: The effect of dietary protein and phosphorus on ammonia concentration and litter composition in broilers. *Poultry Science*, 77, 1085-1093.

Ferm, M., 1979. Method for determination of atmospheric ammonia. *Atmospheric Environment* 13, 1385-1393.

Ferm, M, 1991. A sensitive diffusional sampler. Report B-1020, Swedish Environmental Research Institute, Gothenburg, 12 pp

Ferm, M., Kasimir Klemedtsson, A., Weslien, P. & Klemedtsson, L., 1999. Emission of NH_3 and N_2O after spreading of pig slurry by broadcasting or band spreading. *Soil Use and Management* 15, 27-33.

Flesch, T, Wilson, J.D. & Yee, E., 1995. Backward time Lagrangian stochastic dispersion models, and their application to estimate gaseous emissions. *Journal of Applied Meteorology* 34, 1320-1332.

Flessa, H. & Beese, F, 2000. Laboratory estimates of trace gas emissions following surface application and injection of cattle slurry. *Journal-of-Environmental-Quality*. Jan.-Feb., 2000, 29 (1), 262-268.

Fowler, D., Hargreaves, K.J, Skiba, U., Milne, R., Zahniser, M.S., Moncrieff, J.B., Beverland, I.J. & Gallagher, M.W., 1995. Measurements of CH_4 and N_2O fluxes at the landscape scale using micrometeorological methods. *Philosophical Transactions of the Royal Society of London Series a – Mathematical Physical and Engineering Sciences,* 351, 339-355.

Frost, J.P., 1994. Effect of spreading method, application rate and dilution on ammonia volatilisation from cattle slurry. *Grass and Forage Science* 49, 391-400.

Frick, R. & Menzi, H., 1997. Hofdüngeranwedung: Wie Ammoniakverluste vermindern? (Use of animal manure: How to reduce ammonia losses.) *FAT-Berichte*: 496, 29-40.

Gaakeer, W.A., 1998. Mestaanwendings gids '98, technische gegevens en prijzen, *Wageningen Pers, Wageningen*, 47 pp.

Galle, B., Klemedtsson, L., Bergqvist, B., Ferm, M., Tornqvist, K., Griffith, D. W.T., Jensen, N.O. & Hansen, F., 2000. Measurements of ammonia emissions from spreading of manure using gradient FTIR techniques. *Atmospheric Environment* 34, 4907-4915.

Genermont, S. & Cellier, P., 1997. A mechanistic model for estimating ammonia volatilization from slurry applied to bare soil. *Agricultural and Forest Meteorology* 88, 145-167.

Génermont, S., Cellier, P., Morvan, T., Flura, D. & Laville, P., 1996. Mesure de la volatilisation d'ammoniac à l'aide de la méthode aérodynamique dans les conditions de la pratique agricole. C. R. Acad. Sci., Sciences de la vie/Life sciences, 319: 1145-1151.

Genermont, S., Cellier, P., Flura, D., Morvan, T. & Laville, P., 1998. Measuring ammonia fluxes after slurry spreading under actual field conditions. *Atmospheric Environment* 32, 279-284.

Griffith, D.W.T. & Galle, B. (2000). Flux measurements of NH_3, N_2O and CO_2 using dual beam FTIR spectroscopy and the flux-gradient technique. *Atmospheric Environment* 34, 1087-1098.

Hansen, M.N., Sommer, S.G. & Madsen, N.P., 2001. Reduction of ammonia emission by shallow slurry injection – effects of injection efficiency. *J. Environ. Qual*. Submitted.

Harper, L.A., Catchpoole, V.R., Davis, R & Weir, K.L., 1983. Ammonia volatilization: soil, plant and microclimate effects on diurnal and seasonal fluctuations. *Agronomy Journal* 75, 212-218.

Healy, T.V., McKay, H.A.C., Pilbeam, A. & Scargill, D., 1970. Ammonia and ammonium sulfate in the trophosphere over the United Kingdom. *Journal of Geophysical Research* 75, 2317-2327.

Heise, H.M., Muller, U., Gartner, A.G. & Holscher, N., 2001. Improved chemometric strategies for quantitative FTIR spectral analysis and applications in atmospheric open-path monitoring. *Field Analytical Chemistry and Technology,* 5, 13-28.

Heber, A.J., Duggirala, R.K. & coll., 1997. Manure treatment to reduce gas emissions from large swine houses. Proceedings of the International Symposium "Ammonia and odour control from animal production facilities", Vinkeloord, The Netherlands, October 6-10, 1997. Jan A.M. Voermans & GertJan Monteny (eds.), 449-457.

Hendriks, J.G.L., Vrielink, M.G.M. & van der Peet-Schwering, C.M.C., 1997: Reducing ammonia emission from pig housing by adding acid salts to the feed. *Livestock environment*, 5, 65-70.

Hol, J.M.G. & Huijsmans, J.F.M., 1998. Mogelijkheden voor hogedruk mestinjector (possibiltities of a high pressure injector). *Landbouwmechanisatie* 49: 6/7: 22-23.

Huijsmans, J., 1998. Application techniques to evenly distribute slurry and to reduce ammonia losses on grasland. In: *Proceedings of International Workshop on Environmenally Friendly Management of Farm Animal Waste,* 207-211.

Huijsmans, J.F.M. & De Mol, R.M., 1999. A model for Ammonia Volatilization after Surface Application and Subsequent Incorporation of Manure on Arable Land. *J. Agric. Engng Res,* 74, 73-82.

Huijsmans, J.F.M., Hol, J.M.H. & Bussink, D.W., 1997. Reduction of ammonia emission by new slurry application techniques on grassland. In *Gaseous Nitrogen Emissions from Grasslands*, (eds. S.C Jarvis & B.F. Pain), CAB International, 281- 285.

Huijsmans, J. & Bussink, D.W., 1990. Mestoediening op grasland (Slurry application to grassland). *Landbouwmechanisatie* Vol. 41:5, 38-39.

Itier, B., 1981. Une méthode simplifiée pour la mesure de l'évapotranspiration réelle à l'échelle de la parcelle. *Agronomie,* 1, 869-876.

James, T., Meyer, D., Esparza, E., Depeters, E.J. & Perez Monti, H., 1999. Effects of dietary nitrogen manipulation on ammonia volatilization from manure from Holstein heifers. *Journal of Dairy Science*, 82, 2430-2439.

Kay, R.M., Lee, P.A., 1997: Ammonia emission from pig buildings and characteristics of slurry produced by pigs offered low crude protein diets. In: Voermans J.A.M. and Monteny G.J. (Eds): "Ammonia and odour emissions from animal production facilities", *Proc. International Symposium, Vinkeloord*, NL, 6-10 October, 1997, 253-260

Kirchgessner, M., Windisch, W. & Roth, F.X., 1994: The efficiency of nitrogen conversion in animal nutrition. *Nova Acta Leopoldiana*, 70/288, 393-412.

Kirchgessner, M., Kreuzer, M., 1986: Beeinflussung der Gülle durch Variation der Ernährungsintensität bei Milchkühen. *Landwirtschaftliche Forschung*, 39, 351-368.

Kreuzer, M., Machmüler, A., Gerdemann, M.M., Hanneken, H. & Wittmann, M., 1998: Reduction of gaseous nitrogen loss from pig manure using feeds rich in easily-fermentable non-starch polysaccharides. *Animal Feed Science and Technology*, 73, 1-19.

Kröber, T.F., Külling, D.R., Menzi, H., Sutter, F. & Kreuzer, M., 2000b. Quantitative Effects of Feed Protein Reduction and Methionine on Nitrogen Use of Cows and Nitrogen Emission from Slurry. *Journal of Dairy Science*, 83, 2941-2951.

Külling, D.R., Menzi, H., Kröber, T.F., Neftel, A., Sutter, F., Lischer, P. & Kreuzer, M., 2001a: Emissions of ammonia, nitrous oxide and methane from different types of dairy manure during storage as affected by feed protein content. *Journal of Agricultural Science*, in press.

Külling, D.R., Menzi, H., Sutter, F., Lischer, P. & Kreuzer, M., 2001b: Ammonia, nitrous oxide and methane emissions from differently stored dairy manure derived from grass- and hay-based rations. *Nutrient Cycling in Agroecosystems*, in press.

Lenis, N.P. & Jongbloed, A.W., 1999. New technologies in low pollution swine diets: diet manipulation and use of synthetic amino acids, phytase and phase feeding for reduction of nitrogen and phosphorus excretion and ammonia emission – review. *Asian-Australasian Journal of Animal Sciences*, 12, 305-327.

Lenschow, D.H., 1995. Micrometeorological techniques for measuring biosphere-atmosphere trace gas exchange. In: *Biogenic Trace Gases: Measuring Emissions from Soil and Water*. Matson, P.A. & Harriss, R.C. (eds.), Blackwell Science, Oxford.

Leuning, R., Freney, J.R., Denmead, O.T. & Simpson, J.R., 1985. A sampler for measuring atmospheric ammonia flux. *Atmospheric Environment,* 19, 1117-1124.

Livingston, G.P. & Hutchinson, G.L., 1995. Enclosure-based measurements of trace gas exchange: application and sources of error. In: *Biogenic Trace Gases: Measuring Emissions from Soil and Water*. Matson, P.A. & Harriss, R.C. (Eds.) Blackwell Science, Oxford.

Lockyer, D.R., 1984. A system for the measurement in the field of losses of ammonia through volatilization. *Journal of the Science of Food and Agriculture* 35, 837-848.

Loubet, B., Cellier, P., Flura, D. & Genermont, S., 1999. An evaluation of the wind-tunnel technique for estimating ammonia volatilization from land: Part 1. Analysis and improvement of accuracy. *Journal of Agricultural Engineering Research,* 72, 71-81.

Malgeryd, J. & Wetterberg, C., 1996. Physical properties of solid and liquid manures and their effects on the performance of spreading machines. *Journal of Agricultural Engineering Research,* 64, 289-298.

Magliulo, V., Peressotti, A. & Alterio, G., 2000. Improved microclimate techniques for monitoring trace gas emissions in animal waste. Poster presentation at 9th International Workshop of the FAO Network on Recycling of Agricultural, Municipal and Industrial Residues in Agriculture, 6-9 September, 2000, Gargnano, Italy.

Martinez, J., Jolivet, J., Guiziou, F. & Langeoire, G., 1997. Ammonia emissions from pig slurries: evaluation of acidification and the use of additives in reducing losses. *Proceedings of the International Symposium "Ammonia and odour control from animal production facilities" , Vinkeloord, The Netherlands*, October 6-10, 1997. Jan A.M. Voermans & GertJan Monteny (eds.), 485- 492.

McGinn, S.M. & Pradhan, R., 1997. Reducing ammonia loss from manure amended soils. *Proceeding of 34th Annual Alberta Soil Science Workshop, Calgary*, AB.

McInnes, K.J., Kissel, D.E. & Kanemasu, E.T., 1985. Estimating ammonia flux – a comparison between the integrated horizontal flux method and theoretical solutions of the diffusion profile. *Agronomy Journal,* 77, 884-889.

Mennen, M.G., VanElzakker, B.G., VanPutten, E.M., Uiterwijk, J.W., Regts, T.A., VanHellemond, J., Wyers, G.P., Otjes, R.P., Verhage, A.J.L., Wouters, L.W., Heffels, C.J.G., Romer, F.G., VandenBeld, L. & Tetteroo, J.E.H., 1996. Evaluation of automatic ammonia monitors for application in an air quality monitoring network. *Atmospheric Environment* 30, 3239-3256.

Mennicken L., 2000. Möglichkeiten der Minimierung gasförmiger Emissionen in Legehenneställen. In: *Jahrbuch für die Geflügelwirtschaft 2001*, Peterson J. (Ed.), Verlag Eugen Ulmer, 34-43.

Menzi, H., Frick, R. & Kaufmann R., 1997. Ammoniakemissionen in der Schweiz: Ausmass und technische Beurteilung des Reduktionspotentials. *Schriftenreihe der Forschungsanstalt für Agrarökologie und Landbau (FAL)*, 26, 107 pp.

Mestoediening op grasland (Slurry application to grassland). *Landbouwmechanisatie* Vol. 41:5, 38-39.

Miner, J.R., 1984. Use of natural zeolites in the treatment of animal wastes. In *Zeo-Agriculture*. Use of Natural Zeolites in Agriculture and Aquaculture. Eds. WG Pond and FA Mumpton, 257-262. Westview Press, Boulder, Colorado.

Misselbrook, T.H., Chadwick, D.R., Pain B.F. & Headon D.M., 1998. Dietary manipulation as a means of decreasing N losses and methane emissions and improving herbage N uptake following application of pig slurry to grassland. *Journal of Agricultural Science*, 130, 183-191.

Misselbrook, T. & Hansen, M., 2001. Field evaluation of the equilibrium concentration technique (JTI method) for measuring ammonia emission from land spread manure or fertiliser. *Atmospheric Environment* 35, 3761-3768.

Moal, J-F., Martinez, J., Marol, C. & Guiraud, G., 1994. Soil-air nitrogen balance using 15N labelled pig slurry during ammonia volatilisation studies. Report No. 94-A-003 *Proceedings of AgEng 94 Conference*, Milan.

Moal, J.-F, Martinez, J., Guiziou, F. & Coste, C.-M., 1995. Ammonia volatilization following surface-applied pig and cattle slurry in France. *Journal of Agricultural Science,* Cambridge, 125, 245-252.

Molen, van der J., Beljaars, A.C.M., Chardon, W.J., Jury, W.A. & Faasen, van der H.G., 1990. Ammonia volatilisation from arable land after application of cattle slurry. 2. Derivation of a transfer model. *Netherlands Journal of Agricultural Science* 38, 239-254.

Monin, A.S. & Obukhov A.M., 1954. Basic laws of turbulent mixing in the groud layer of the atmosphere. *Trudy Geoph. Inst. Akad. Nauk SSSR* 24, 163-187.

Monteith, J.L., 1973. Principles of Environmental Physics. Edward Arnold, London.

Morken-John, Sakshaug-Stein, 1998. Direct ground injection of livestock waste slurry to avoid ammonia emission. *Nutrient-Cycling-in-Agroecosystems*. May, 1998, 51 (1), 59-63.

Morvan, T., Leterme, Ph., Arsene, G.G. & Mary, B., 1997. Nitrogen transformations after the spreading of a pig slurry on bare soil and ryegrass using 15N-labelled ammonium. *European Journal of Agronomy*, 7, 181-188.

Oenema, O.; Velthof, G.L., 1993. Denitrification in nitric-acid-treated cattle slurry during storage. *Netherlands Journal of Agricultural Science,* 41, 63-80.

Oenema, O., Velhof Yamulki, S. & Jarvis S.C., 1997. Nitrous oxide emissions from grazed grassland. *Soil Use and Management*, 13, 288-295.

Paul J.W., Dinn N.E., Kannangara T. & Fisher L.J., 1998. Protein content in dairy cattle diets affects ammonia losses and fertilizer nitrogen value. *Journal of Environmental Quality*, 27, 528-534.

Pain, B.F., Phillips, V.R., Clarkson, C.R. and Klarenbeek, J.V., 1989. Loss of nitrogen through ammonia volatilisation during and following the application of pig or cattle slurry to grassland. *Journal of the Science of Food and Agriculture*, 47, 1-12.

Pain, B.F., Thompson, R.B. Rees, Y.J. & Skinner, J.H., 1990. Reducing gaseous losses of nitrogen from cattle slurry applied to grassland by the use of additives. *J. Sci. Food Agric*. 50, 141-153.

Peet Schwering, C.M.C., Van der, Aarnink, A.J.A., Rom, H.B. & Dourmad, J.Y., 1999a. Ammonia emissions from pig houses in The Netherlands, Denmark and France. *Livestock Production Science*, 58, 265-269.

Peet Schwering, C.M.C. Van der, Jongbloed, A.W. & Aarnink, A.J.A., 1999b. Nitrogen and phosphorus consumption, utilisation and losses in pig production: The Netherlands. *Livestock Production Science*, 58, 213-224.

Philip, J.R., 1959. The Theory of Local Advection. *International Journal of Meteorology* 16, 535-547.

Phillips, V.R., Cowell, D.A., Sneath, R.W., Cumby, T.R., Williams, A.G., Demmers, T.G.M. & Sandars, D.L., 1999. An assessment of ways to abate ammonia emissions from UK livestock buildings and waste stores. Part 1: ranking exercise. *Bioresource Technology*, 70, 143-155.

Priesmann, T. & Petersen, J., 1995. Einfluss einer kontrollierten Fütterung von Legehennen auf die Leistung und den Anfall von Stickstoff und Phosphor im Kot. *Archiv Geflügelkunde*, 59, 198-207.

Reitz, P. & Kutzbach, H., 1997. Accuracy of a wind tunnel system for measuring ammonia emissions after slurry application. In: *Ammonia and odour control from animal production facilities* (eds. Voermans, J.A M. & Monteney, G.–J.), Vinkleloord, NL, 595-603.

Raupach, M.R. & Legg, B.J., 1984. The uses and limitations of flux-gradient relationships in micrometeorology. *Agriculture and Water Management,* 8, 119-131.

Rodhe, L & Karlson, S., 2000. Ammonia emissions from broiler manure – influence of storage and spreading method, *AgEng paper 00 AP-020*, Warwick.

Rodhe, L. & Rammer, C., 2001. A Comparison of Methods for Applying Cattle Slurry to Ley. Submitted to *J. Agric. Engng Res.*

Rodhe, L, Salamon, E., & Johansson, S., 1997. Spreading of Cattle Urine to leys: techniques, ammonia emissions and crop yields. In *Gaseous Nitrogen Emissions from grassland* (eds.: S.C. Jarvis & B.F. Pain).

Rubæk, G.H., Henriksen, K., Petersen, J., Rasmussen, B. & Sommer, S., 1996. Effects of application technique and anaerobic digestion on gaseous loss from animal slurry applied to ryegrass (lolium perenne). *Journal of Agricultural Science*, Cambridge, 126: 481-492.

Ryden, J.C. & McNeil, J.E., 1984. Application of the micrometeorological mass balance method to the determination of ammonia loss from a grazed sward. *Journal of the Science of Food and Agriculture,* 35, 1297-1310.

Ryden, J.C. & Lockyer, D.R., 1985. Evaluation of a system of wind tunnels for field studies of ammonia loss from grassland through volatilization. *Journal of the Science of Food and Agriculture,* 36, 781-788.

Safley, L.M., Barker, J.C. & Westerman, P.W. 1992. Loss of nitrogen during sprinkler irrigation of swine lagoon liquid. *Bioresource Technology,* 40, 7-15.

Schafer, K., Emeis, S., Stockhause, M., Sedlmaier, A., Hoffman, H., Depta, G. & Gronauer, A., 1998 Emission rates of benzene and ammonia area sources determined by spectroscopic remote measurements and inverse dispersion modelling. In: *Environmental Monitoring and Remediation Technologies*. Vo-Dinh, T. & Spellicy, R.L. (Eds) Proceedings of the Society for Optical Engineering, Vol. 3534, Washington, 212-219.

Schjoerring, J.K., Sommer, S.G. & Ferm, M., 1992. A simple passive sampler for measuring ammonia emission in the field. Water Air and Soil Pollution, 62, 13-24.

Sharpe, R.R. & Harper, L.A., 1997. Ammonia and nitrous oxide emissions from sprinkler irrigation applications of swine effluent. *Journal of Environmental Quality,* 26, 1703-1706.

Smith, K.A. & Chambers, B.J., 1995. Muck from waste to resource – utilization: the impacts and implications. Agricultural Engineering, 33-88.

Smith, K.A., Brewer, A.J., Dauven, A. & Wilson, D., 2000. A Survey of the Production and Use of Animal Manures in England and Wales. I. Pig manure. *Soil Use & Management* 16, 124-132.

Smits, M.C.J., Valk, H., Elzing, A. & Keen, A., 1995. Effect of protein nutrition on ammonia emission from a cubicle house for dairy cattle. *Livestock Production Science*, 44, 147-156

Sommer, S.G. & Hutchings, N., 1995. Techniques and strategies for the reduction of ammonia emission from agriculture. *Water, Air and Soil Pollution,* 85, 237-248.

Sommer, S.G. & Jacobsen, O.H., 1999. Infiltration of slurry liquid and volatilization of ammonia from surface applied pig slurry as affected by soil water content. *Journal of Agricultural Science*, Cambridge, 132, 297-303.

Sommer, S.G., Mikkelsen, H. & Mellqvist, J., 1995. Evaluation of meteorological techniques for measurements of ammonia loss from pig slurry. *Agricultural and Forest Meteorology,* 74, 169-179.

Sommer, S.G., Olesen, J.E. & Christensen, B.T., 1991. Effects of temperature, wind speed and air humidity on ammonia volatilization from surface applied cattle slurry. *J. Agric. Sci*. 117, 91-100.

Sommer, S.G. & Ersbøll, S.K., 1994. Soil tillage effects on ammonia volatilization from surface-applied or injected animal slurry. *Journal of Environmental Quality,* 23, 493-498.

Sommer, S.G. & Olesen, J.E., 2000. Modelling ammonia volatilization from animal slurry trailing hose applied to cereals. *Atmos. Environ*. 34: 2361-2372.

Sommer, S.G. & Olesen, J.E., 1991. Effect of dry matter content and temperature on NH_3 loss from surface-applied cattle slurry. *Journal of Environmental Quality,* 20, 679-683.

Sommer, S.G. & Sherlock, R.R., 1996. pH and buffer component dynamics in the surface layers of animal slurries. *Journal of Agricultural Science*, Cambridge 127, 109-116.

Sommer, S.G., Friis, E., Bak, A.B. & Schjørring, J.K., 1997. Ammonia volatilization from pig slurry applied with trail hoses or broadspread to winter wheat: Effects of crop developmental stage, microclimate, and leaf ammonia absorption. *Journal of Environmental Quality*, 26, 1153-1160.

Spiekers, H & Pfeffer, E., 1990: Emissionsminderung durch angepasste Fütterung. In: *Proc. "Ammoniak in der Umwelt: Kreislaufe, Wirkungen, Minderung"*, conference held in Braunschweig, Germany, 10-12 October, 1990, Hartung J., Paduch M., Schirz S., Döhler H. and Van den Weghe H. (Eds.), KTBL, Darmstadt, Germany, 24.1-24.15.

Stadelmann, F., Menzi, H., Pfefferli, S. & Zimmermann, A., 1998. Ammonia Emissions in Switzerland: Present situation, development, technical and economic assessment of abatement measures, recommendations. Institute of Environmental Protection and Agriculture Liebefeld (IUL) and Swiss Federal Research Station for Agricultural Economics and Engineering Tänikon (FAT), 56 pp.

Steffens, G & Lorenz, F., 2000. Aspects of good agricultural practice in the application of organic wastes. In: *Scientific basis for codes of good agricultural practice.* EUR 14957 en, ISSN 1018-5593.

Stevens, R.J., Laughlin, R.J. & Frost, J.P., 1989. Effect of acidification with sulphuric acid on the volatilization of ammonia from cow and pig slurries. *Journal of Agricultural Science*, Cambridge, 113, 389-395.

Stevens, R.J.; Laughlin, R.J., O'Bric, C.J., Carton, O.T. & Lenehan J.J., 1997. The efficiency of the nitrogen in cattle slurry acidified with nitric acid for grass production. *Journal of Agricultural Science*, Cambridge, 129, 335-342.

Subair, S., Fyles, J.W. & O'Halloran, I.P., 1999. Ammonia volatilization from liquid hog manure amended with paper products in the laboratory. *Journal of Environmental Quality* 28, 202-207.

Sutter, F. & Kreuzer, M., 1995. Environmental Consequences of an Early-Lactational Protein Deficit in Cows. *Proc. symposium "Applied Research for sustainable dairy farming"*, May 31-June 2, 1995, Research station for cattle sheep and horse husbandry, Lelystad, The Netherlands, 125.

Sutton, M.A., Moncrieff, J.B. & Fowler, D., 1992. Deposition of atmospheric ammonia to moorlands. Environmental Pollution, 75, 15-24.

Sutton, A.L., Kephart, K.B., Patterson, J.A., Mumma, R., Kelly, D.T., Bogus, E., Jones, D.D. & Heber, A., 1997: Dietary manipulation to reduce ammonia and odorous compounds in excreta and anaerobic manure storages. In: Voermans J.A.M. and Monteny G.J. (Eds): "Ammonia and odour emissions from animal production facilities", *Proc. International Symposium, Vinkeloord, NL*, 6-10 October, 1997, 245-252.

Svensson, L., 1994. A new dynamic chamber technique for measuring ammonia emissions from land-spread manure and fertilizers. Acta Agriculturae Scandinavica Section B-Soil and Plant Science, 44, 35-46.

Svensson, L. & Ferm, M., 1993. Mass transfer coefficient and equilibrium concentration as key factors in a new approach to estimate ammonia emission from livestock manure. *J. Agric. Engng. Res.,* 56, 1-11.

Tamminga, S., 1992. Nutrition management of dairy cows as a contribution to pollution control. *Journal of Dairy Science*, 75, 345-357.

Tamminga, S., 1990. Feeding management of dairy cows as a contribution to pollution control. *Journal of Dairy Science*, 73 (Supplement 1), 123.

Thompson, R.B., Pain, B.F. & Lockyer, D.R., 1990a. Ammonia emission from catte slurry following surface application to grassland. Influence of mechanical separation, changes in chemical composition during emission and the presence of the grass sward. Plant Soil, 125, 109-117.

Thompson, R.B., Pain, B.F. & Rees, Y.J., 1990b. Ammonia emission from cattle slurry following surface application to grassland. II. Influence of application rate, wind speed and applying slurry in narrow bands. Plant Soil. 125, 119-128.

UN/ECE, 1999. Control technique for preventing and abating emissions of ammonia. 37 pp + annexes.

Vandré, R. & Kaupenjohann, M., 1998. In-situ measurement of ammonia emissions from organic fertilizers in plot experiments. *Soil Science Society of America Journal,* 62, 467-473.

Vandré, R. & Clemens, J., 1997. Studies on the relationship between slurry pH, volatilization processes and the influence of acidifying additives. *Nutrient Cycling in Agroecosystems* 47, 157-165.

Vandré, R., Clemens, J., Goldach, H. & Kaupenjohann, M., 1997. NH_3 and N_2O emissions after landspreading of slurry as influenced by application technique and dry matter-reduction. I. NH_3 emissions. Z. Pflanzenernähr. bodenk., 160, 303-307.

Vogt, F., Klocke, U., Rebstock, K., Schmidtke, G., Wander, V. & Tacke, M., 1999. Optical UV derivative spectroscopy for monitoring gaseous emissions. Applied Spectroscopy, 53, 1352-1360.

Vuuren, A.M. Van, van der Koelen, C.J., Valk H. & de Visser, H., 1993a. Effects of partial replacement of ryegrass by low protein feeds on rumen fermentation and nitrogen loss by dairy cows. *Journal of Dairy Science*, 76, 2982-2993.

Vuuren, A.M. Van, van der Koelen, C.J. & Vroons de Bruin, J., 1993b. Ryegrass versus corn starch or beet pulp fiber diet effects on digestion and intestinal amino acids in dairy cows. *Journal of Dairy Science*, 76, 2692-2700.

Weslien, P., Klemedtsson, L., Svensson, L., Galle, B., Kasimir-Klemedtsson, Å. & Gustafsson, A., 1998. Nitrogen losses following application of pig slurry to arable land. *Soil Use and Management,* 14, 200-208.

Witter, E., 1991. Use of $CaCl_2$ to decrease ammonia volatilization after application of fresh and anaerobic chicken slurry to soil. *Journal of Soil Science*, 42, 369-380.

Warland, J.S., Dias, G.M. & Thurtell, G.W., 2001. A tunable diode laser system for ammonia flux measruements over multiple plots. *Environmental Pollution,* 114, 215-221.

Weerden, T.J., van der Moal, J.F., Martinez, J., Pain, B.F. & Guiziou, F., 1996. Evaluation of the wind-tunnel method for measurement of ammonia volatilization from land. *Journal of Agricultural Engineering Research,* 64, 11-13.

Wilson, J.D. & Shum, W.K.N., 1992. A re-examination of the integrated horizontal flux method for estimating volatilization from circular plots. *Agricultural and Forest Meteorology,* 57, 281-295.

Wilson, J.D., Thurtell, G.W., Kidd, G.E. & Beauchamp, *E.G.*, 1982. Estimation of the rate of gaseous mass-transfer from a surface source plot to the atmosphere. *Atmospheric Environment* 16, 1861-1867.

Wilson, J.D., Catchpole, V.R., Denmead, O.T. & Thurtell, G.W., 1983 Verification of a simple micrometeorological method for estimating the rate of gaseous mass transfer from the ground to the atmosphere. *Agricultural Meteorology*, 29, 183-189.

Zhu, J., Bundy, D.S., Li, X.W. & Rashid, N., 1996. Reduction of odor and volatile substances in pig slurries by using pit additives. *J. Environ. Sci. Health*, A31(10), 2487-2501.