Comparison of models used for national agricultural ammonia emission inventories in Europe: Liquid manure systems

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Abstract

Ammonia (NH\textsubscript{3}) emissions from agriculture commonly account for >80% of the total NH\textsubscript{3} emissions. Accurate agricultural NH\textsubscript{3} emission inventories are therefore required for reporting within the framework of the Gothenburg Protocol of the UN Convention on Long-range Transboundary Air Pollution. To allow a co-ordinated implementation of the Protocol, different national inventories should be comparable. A core group of emission inventory experts therefore developed a network and joint programme to achieve a detailed overview of the best inventory techniques currently available and compiled and harmonized the available knowledge on emission factors (EFs) for nitrogen (N)-flow emission calculation models and initiated a new generation of emission inventories. As a first step in summarizing the available knowledge, six N-flow models, used to calculate national NH\textsubscript{3} emissions from agriculture in different European countries, were compared using standard datasets. Two scenarios for slurry-based systems were run separately for dairy cattle and for pigs, with three different levels of model standardisation: (a) standardised inputs to all models (FF scenario); (b) standard N excretion, but national values for EFs (FN scenario); (c) national values for N excretion and EFs (NN scenario). Results of the FF scenario showed very good agreement among models, indicating that the underlying N flows of the different models are highly similar. As a result of the different national EFs and N excretion rates, larger differences among the results were observed for the FN and the NN scenarios. Reasons for the differences were primarily attributed to differences in the agricultural practices and climatic factors reflected in the EFs and the N excretion rates. The scientific debate
necessary to understand the variation in the results generated awareness and consensus concerning available scientific data and the importance of specific processes not yet included in some models.

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1. Introduction

The Gothenburg Protocol of the UN Convention on Long-range Transboundary Air Pollution (UN-ECE, 1999) and the EU’s National Emission Ceiling Directive (EC, 2001) require the reporting of national annual emissions of ammonia (NH$_3$). Accurate inventories of agricultural NH$_3$ emissions are required to calculate the national emissions, since they commonly account for >80% of the total emissions (EMEP, 2005). Accurate inventories are also necessary to identify the major sources and hence to develop effective abatement strategies. Ideally, the emission inventory approach must give a true picture of emissions, reliably and reproducibly show changes over time, recognize relatively small changes and take into account all relevant and measurable variables that influence emissions. Furthermore, to allow a co-ordinated implementation of the Protocol, different national inventories should be comparable.

The first inventories of NH$_3$ emissions from livestock production were calculated by multiplying livestock numbers by emission factors (EFs) per animal (e.g. Buijsman et al., 1987). This approach did not allow for significant differences in the potential for NH$_3$ emissions due to differences in performance, diet and hence nitrogen (N) excretion, or differences in livestock and manure management practices among countries and regions.

More recent inventories have replaced EFs per animal with partial EFs for grazing, animal housing, manure storage and manure spreading. Such inventories can take into account differences in aspects of livestock management that have an influence on NH$_3$ emissions, such as the length of the grazing season for ruminants and whether livestock manures are managed in liquid or solid form (e.g. Chambers et al., 2003). However, increasing the number of EFs to discriminate among emissions at each stage of manure management and among systems is insufficient, since it cannot account for changes in farm management practices and for interactions among the stages of manure management that occur when abatement measures are applied. In particular, such an approach may fail to recognize that introducing abatement at an early stage of manure management (e.g. housing) will increase the potential of NH$_3$ emissions later (e.g. during storage or after spreading) by conserving NH$_4^+$-N. Thus a mass-flow approach is needed, in which the fate of N is followed throughout the manure management system. This is particularly important when attempting to rank the costs of introducing measures to reduce NH$_3$ emissions or the side effects on other gaseous N species (nitrous oxide, nitric oxide and dinitrogen). Such a mass-flow approach was used by Menzi and Katz (1997) and in the MARACCAS model (Cowell and ApSimon, 1998). Today such models have been developed to estimate emissions and the potential for abatement in a number of European countries: Switzerland (‘DYNAMO’, Menzi et al., 2003); UK (‘NARSES’, Webb and Mиселбрук, 2004); Germany (‘GAS-EM’, Dammgen et al., 2003); Netherlands (‘MAM’, Groenwold et al., 2002; ‘FarmMin’, Van Evert et al., 2003). Coordination of model development is advisable, to pool knowledge, create synergies and guarantee good congruency among emission models. A core group of emission inventory experts was therefore formed in 2003 to develop a network and joint programme. The aim was to achieve a detailed overview of the presently best available inventory techniques, compile and harmonize the available knowledge on EFs for mass-flow emission calculation models and initiate a new generation of emission inventories.

As a first step in summarizing the available knowledge, the objective of the work reported in this paper was to determine how far the results obtained with different models used for agricultural NH$_3$ emission inventory calculations agree for defined livestock scenarios. A detailed comparison of the models and the underlying EFs permitted common calculation principles to be described and the most important reasons for disagreements to be identified. To limit the resources required, the comparison was restricted to slurry-based manure management systems.
2. Material and methods

2.1. Livestock and manure management systems examined

The models were used to simulate the emissions from two types of livestock categories: dairy cows and fattening pigs. Both animal categories were kept in housing systems producing slurry (Table 1). Calculations were made on an annual basis.

2.2. Scenarios

Three different levels of model comparisons were conducted (Table 2). Calculating the emissions with standardized values allows differences among models with respect to the calculation of the N flow to be detected. For this purpose, the national specific N excretion rates and total ammoniacal N (TAN) contents as well as the EFs were replaced in each of the models by a set of standardised, fixed (F) values (scenario FF; Tables 3–6). At a second level of comparison, N excretions and TAN contents were kept standardized (scenario FN; Tables 3–6), whereas national EFs were used for the calculations (scenario FN; Tables 3–6). Thus, differences among the calculated emissions are related to different EFs that reflect differences in manure management systems and climate, and differences in the scientific basis for the EFs. At the last level of comparison, emissions were calculated using the national N excretion rates, TAN contents and EFs (scenario NN; Tables 3–6). As for the FN scenario, differences are expected to be primarily the result of differences among country-specific livestock and manure management systems. However, because N excretion rates and EFs are to some extent interdependent variables, differences of the observed emissions in the second scenario may be partly compensated (e.g. larger EFs may compensate for reduced N excretion). The calculated emissions can thus be assumed to truly represent the specific NH₃ emissions of a given livestock category of the respective country for the specific manure management scenario. It should be noted, however, that the N excretion rates, TAN contents and EFs used in this scenario are not comparable to the average situation of the different countries.

2.3. General model description

The models compared in this paper have been used in the framework of the national NH₃ emission inventory calculations and manure policy analyses in different countries of Europe. All the models use the mass-flow approach, starting with a specific amount of N excreted by a defined livestock category (Fig. 1). The advantage of this approach is that the effect of emissions that occur in an ‘upstream’ part of the manure management system (e.g. animal housing) on emissions in the subsequent ‘downstream’ parts (e.g. storage) can be taken into account. This enables the impact of abatement measures to be assessed at the system scale.

A livestock category describes a group of animals of the same species that are managed with the same production objective (e.g. dairy cattle). If census data permit a description of homogeneous animal

<table>
<thead>
<tr>
<th>Table 1</th>
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<tbody>
<tr>
<td>Activity data used in the different scenarios</td>
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<tr>
<td><strong>Dairy cows</strong></td>
</tr>
<tr>
<td>Animal numbers</td>
</tr>
<tr>
<td>Housing system</td>
</tr>
<tr>
<td>Type of collected manure</td>
</tr>
<tr>
<td>Housing period [days per year/hours per day]</td>
</tr>
<tr>
<td>Grazing period [days per year/hours per day]</td>
</tr>
<tr>
<td>Slurry storage</td>
</tr>
<tr>
<td>Slurry application</td>
</tr>
<tr>
<td>Application season: Summer/spring or autumn [%]</td>
</tr>
<tr>
<td>Application crop: Arable land/ grassland [%]</td>
</tr>
</tbody>
</table>

ᵃ5 h per day in the house for milking.

<table>
<thead>
<tr>
<th>Table 2</th>
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<tbody>
<tr>
<td>Description of the different scenarios</td>
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<tr>
<td>Scenario</td>
</tr>
<tr>
<td>FF</td>
</tr>
<tr>
<td>FN</td>
</tr>
<tr>
<td>NN</td>
</tr>
</tbody>
</table>

ᵃSame value used in all models.ᵇModel-specific values used.
populations that differ in age, sex, feeding or manure management, a livestock category can be further disaggregated into subcategories.

The N excreted is dependent on the livestock category and is influenced by the amount and composition of the feed ration and the performance of the animal. Since NH₃ emissions originate from the TAN pool of the excreta and manure (e.g. James et al., 1999; Külling et al., 2001), rather than from the organic N, the models simulate separately the TAN and organic N flows over the different stages of emission (grazing, housing, manure storage and application). Ammonia emissions are generally calculated with EFs, i.e. the proportion of the annual flow of TAN through the source that is emitted. The standard EFs are fine-tuned by a set of input variables reflecting specific manure management parameters influencing the NH₃ emissions (e.g. feeding strategy, type of housing systems, type of slurry store cover, manure application rate).

Calculations can be made for individual farms and distinct geographical regions.

The total amount of TAN and organic N excreted by the livestock category must first be partitioned among that which is deposited in animal housing and that which is deposited on pasture during grazing. In all models, this is related to the proportion of the year that the livestock are on the grazed pasture, taking into account the number of grazing days and the grazing hours per day. The NH₃ emission from the excreta deposited on pasture is calculated as the product of the annual amount of TAN deposited and an EF.

In general, the NH₃ emission from the excreta deposited in animal housing is calculated from the annual amount of TAN deposited and an EF depending on the housing type. The remaining TAN and the entire organic N deposited then passes to slurry storage, with the addition of any N added in bedding. Some models take into account the mineralisation of organic N to TAN or losses of TAN via nitrification and denitrification that occur within the slurry storage. Ammonia emissions are calculated as the product of the annual TAN flow through the storage and an EF, where the EF reflects different storage types and covers. The loss of NH₃ from field-applied slurry is calculated as the product of the annual amount of TAN applied and an EF reflecting different application techniques, seasonal distribution of application and time until incorporation. The models only consider the NH₃ emission from the field-applied slurry and not any subsequent emission of N₂ or N₂O. All the field-applied slurry N that is not emitted as NH₃ enters the soil. Each model checks to ensure that mass conservation is obeyed, i.e. that the total N excreted

Table 3
Nitrogen excretion, TAN content of excretions and milk yield used for the different dairy cow scenarios

<table>
<thead>
<tr>
<th>Model</th>
<th>N excretion kg place⁻¹ a⁻¹ N</th>
<th>TAN content %</th>
<th>Milk yield kg place⁻¹ ECMa</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>FF and FN</td>
<td>NN</td>
<td>FF and FN</td>
</tr>
<tr>
<td>DYNAMO</td>
<td>110</td>
<td>110</td>
<td>60</td>
</tr>
<tr>
<td>DanAm</td>
<td>133</td>
<td>55</td>
<td>60</td>
</tr>
<tr>
<td>GAS-EM</td>
<td>126</td>
<td>60</td>
<td>60</td>
</tr>
<tr>
<td>NARSES</td>
<td>106</td>
<td>60</td>
<td>60</td>
</tr>
<tr>
<td>MAM</td>
<td>134</td>
<td>59b</td>
<td>7494</td>
</tr>
<tr>
<td>FarmMin</td>
<td>130</td>
<td>59b</td>
<td>7494</td>
</tr>
</tbody>
</table>

aECM = energy corrected milk.
bTAN content is calculated by the model.

Table 4
Nitrogen excretion, TAN content of excretions used for the different fattening pig scenarios

<table>
<thead>
<tr>
<th>Model</th>
<th>N excretion kg place⁻¹ a⁻¹ N</th>
<th>TAN contenta %</th>
<th>Milk yield kg place⁻¹ ECMa</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>FF and FN</td>
<td>NN</td>
<td>FF and FN</td>
</tr>
<tr>
<td>DYNAMO</td>
<td>15.6</td>
<td>13.0</td>
<td>70</td>
</tr>
<tr>
<td>DanAm</td>
<td>13.7</td>
<td>68</td>
<td>67</td>
</tr>
<tr>
<td>GAS-EM</td>
<td>14.8</td>
<td>67</td>
<td>70</td>
</tr>
<tr>
<td>NARSES</td>
<td>15.6</td>
<td>63</td>
<td>63</td>
</tr>
<tr>
<td>MAM</td>
<td>11.9</td>
<td>63</td>
<td>63</td>
</tr>
</tbody>
</table>

aTAN content is calculated by the model.
equals the total emission of gaseous N via NH$_3$ volatilization and denitrification plus the TAN and organic N entering the soil. Despite the common underlying approach, the models used in the study differ with respect to animal categories, livestock and manure management systems and farm management parameters considered, and the degree to which additional processes are taken into account.

2.3.1. **DYNamic Ammonia MOdel (DYNAMO)**

The Swiss DYNamic Ammonia MOdel (DYNAMO) (Menzi et al., 2003) is used for the calculation of the Swiss NH$_3$ emission inventory (Reidy et al., 2003) and the abatement potential (Reidy and Menzi, 2007).

DYNAMO differs from the generalized model in the following aspects: (1) Emissions are calculated in per cent of total N for grazing and housing. (2) The model does not linearly reduce the housing emissions due to decreased N excretion in the house for grazing animals. This is because the floor surface

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**Table 5**

<table>
<thead>
<tr>
<th>Scenario</th>
<th>FF</th>
<th>FN and NN</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>DYNAMO</td>
<td>DanAm</td>
</tr>
<tr>
<td>Grazing</td>
<td>8.0</td>
<td>6.7$^a$</td>
</tr>
<tr>
<td>Housing</td>
<td>24.1</td>
<td>16.7$^a$</td>
</tr>
<tr>
<td>Storage</td>
<td>15.6</td>
<td>27.7$^c$</td>
</tr>
<tr>
<td>Application</td>
<td>46.9</td>
<td>48.0</td>
</tr>
</tbody>
</table>

$^a$Calculated from total N to TAN.

$^b$Average value, weighted with respect to manure production, using 6.6% for November–April and 16.9% from May to October.

$^c$The basic emission factor is 15% for fully housed animals and a surface to volume ratio of 0.77. With grazing, it is corrected in such a way as to keep emissions per unit surface constant, since it is assumed that partial grazing does not change the emitting surface and TAN concentration during storage.

**Table 6**

<table>
<thead>
<tr>
<th>Scenario</th>
<th>FF</th>
<th>FN and NN</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>DYNAMO</td>
<td>DanAm</td>
</tr>
<tr>
<td>Housing</td>
<td>31.2</td>
<td>20.0</td>
</tr>
<tr>
<td>Storage</td>
<td>6.5</td>
<td>12.0</td>
</tr>
<tr>
<td>Application</td>
<td>46.9</td>
<td>48.0</td>
</tr>
</tbody>
</table>

$^a$Calculated from total N to TAN.

Fig. 1. Schematic N flow of the different models used in the scenario calculations (TAN = total ammoniacal Nitrogen).
will remain contaminated with excreta while the animals are in the house for milking and therefore remain a source of emissions throughout the day (Menzi and Katz, 1997). DYNAMO therefore corrects the housing EF with a stepwise reduction reflecting the time the animals spent grazing. (3) The EF for storage is adjusted in such a way that emissions per unit emitting surface area per year remain constant also when the N flow is reduced during the grazing season.

2.3.2. DanAm
The Danish NH₃ emission inventory system follows the generalised model and was originally developed in the late 1990s (Hutchings et al., 2001). The model is currently being updated and the methods adopted in the new version are used here. The new version has a structure similar to its predecessor but uses EFs expressed on a TAN basis. The EFs for animal housing and storage have been subject to minor updating but the method for calculating emission from field-applied slurry has been revised. The method used has been derived from the statistical model Alfam (Sogaard et al., 2002) and uses two separate parameterizations for slurry from ruminant and non-ruminant animals and is based on data for typical TAN and dry matter contents of Danish dairy cow and pig slurry. To account for the effect of weather on field emissions, seasonal EFs were developed for applications made in spring, summer, autumn and winter, using average Danish meteorological conditions. The annual EFs were calculated as weighted averages of these values, based on the seasonal distribution of slurry applications. Finally, the annual EFs are adjusted to account for the distribution of slurry among application techniques (e.g. trailing hose, injection).

2.3.3. GASeous EMissions (GAS-EM)
GASeous EMissions (GAS-EM) has been used for the calculation of the German agricultural emission inventory (Dämmgen et al., 2003). It was developed to serve as a policy advice tool. In contrast to the generalized model, it calculates emissions not only for ammonia but also for greenhouse gases (e.g. CH₄, N₂O) and for particulate matter (PM₁₀ and PM₂.₅) and considers losses of NO and N₂. Immobilization and mineralization are considered in storage. For the latest version, see Dämmgen et al. (2006).

2.3.4. National Ammonia Reduction Strategy Evaluation System (NARSES)
The National Ammonia Reduction Strategy Evaluation System (NARSES) model (Webb and Misselbrook, 2004) is used to estimate NH₃ emission from agriculture. The objective of the model is to develop a national-scale model to estimate the magnitude, spatial distribution and time course of agricultural NH₃ emissions and the potential applicability of abatement measures with associated costs. NARSES differs from the generalized model in three respects: (1) Partitioning of N excretion by cattle among housing and grazing is not linearly related to the length of time but reflects the greater N concentration of grazed herbage. (2) A proportion of slurry is spread directly from buildings and therefore not subject to storage losses. (3) Ten percent of organic N is calculated to be mineralized to TAN during slurry storage.

2.3.5. Manure and Ammonia emission Model (MAM)
The Manure and Ammonia emission Model (MAM) (Groenwold et al., 2002) is primarily constructed as a tool for Dutch manure policy analysis (Luesink et al., 2004), to calculate manure surpluses and deficits at the farm level and to optimize the distribution of manure at a national scale. MAM differs from the generalized model in the following three aspects: NH₃ emissions are calculated by EFs expressed in per cent of the amount of total N except for field application (per cent of TAN). For ruminants the N excretion and the EFs for housing are seasonally differentiated, and partitioning of N excretion by cattle among housing and grazing is not linearly related to the length of time but to excretion activity at the milking period and the rest of the day.

2.3.6. FarmMin
FarmMin simulates the nutrient flows on a dairy farm, including nutrient losses such as leaching of nitrate and volatilization of NH₃ (Van Evert et al., 2003). The model’s intended purpose is to study the effects of strategic and tactical measures to reduce losses. In contrast to the generalized model the factors for NH₃ emission during grazing and from the house and storage are based on total N. Nitrogen excretion is related to the amounts of N applied to forage crops.
3. Results and discussion

3.1. FF scenario

3.1.1. Dairy cows

Running the models with the defined livestock and manure management parameters and the standardised N excretion and EFs (scenario FF) gave similar estimates of total NH$_3$ emission for dairy cows (Fig. 2). However, greater housing emissions were calculated by DYNAMO. This is because the model does not linearly reduce the housing emissions due to decreased N excretion in the house during the grazing season (see Section 2.3.1). The greater NH$_3$ emissions during housing reduce the TAN pool and therefore also influence the subsequent emissions during storage and application, resulting in lower emissions at these stages as compared to the other models.

Slightly smaller emissions during grazing were calculated with MAM and NARSES. However, this effect was overcompensated by greater emissions during housing, storage and application. The differences can be explained by differences in the way excreted N is partitioned among grazing and housing. Most of the models assume that excreta are partitioned in direct proportion to the time spent in the housing (here 20.8% of the daily N excretion). However, MAM uses 28.5% and NARSES 25.0% to take into account that dairy cows generally show increased excretion activity during the milking period, as compared with the rest of the day (Anon., 1999). As a consequence, more N will be deposited in the housing, thus increasing the TAN flow and the subsequent NH$_3$ losses during housing, storage and application. Because total emissions from excreta collected in the animal houses and handled as manure are greater than emissions from the grazing, the increase in the proportion of N deposited in the house also increases total emissions.

3.1.2. Fattening pigs

Differences between the modelled emissions were also minimal for the FF scenario for fattening pigs (Fig. 3). The greatest total emissions were observed with GAS-EM. Whereas the calculated housing emissions were identical for all models, slightly greater emissions for storage and application were obtained with GAS-EM because this model takes into account a conversion of organic N to TAN during storage (Beline et al., 1998). As a result of the mineralisation, the TAN pool increases and influences the emissions during storage and application. Somewhat smaller emissions during application were obtained with NARSES because the model allows for denitrification losses during storage. The TAN pool is therefore reduced not only by NH$_3$ emissions but also by losses of N$_2$O and N$_2$.

3.2. FN scenario

3.2.1. Dairy cows

In contrast to the FF scenarios, substantial differences in total emissions and in the individual
emission stages were observed for the dairy cow FN scenario. Higher total emissions were observed for the models MAM and FarmMin, whereas lowest emissions were obtained with DYNAMO and NARSES (Fig. 4).

Although all animals were assumed to spend equal times on the pasture, a great variation in the grazing emissions can be seen (Fig. 4). The grazing emissions calculated with DYNAMO and NARSES were lower than the other models. All the models used a relationship between NH₃ emission and the fertilization rate of the grass. This is because greater fertilization leads to an increase in herbage N concentration, N intake and excretion of TAN. However, the data used to construct the relationship vary; DYNAMO and DanAm use data from Bussink (1992), MAM and FarmMin also use additional data from Bussink (1994) and NARSES includes data from (Jarvis et al., 1989) and New Zealand (Ledgard, 1996). GAS-EM uses the default

Fig. 3. Annual ammonia emissions (kg NH₃-N) for the fattening pig FF scenario (fixed N excretion values and EFs, 1000 pig places) as calculated by the different models. The stacked columns represent the amount of ammonia emitted in the respective part of the manure handling chain.

Fig. 4. Annual ammonia emissions (kg NH₃-N) for the dairy cow FN scenario (fixed N excretion values and national EFs, 100 heads) as calculated by the different models. The stacked columns represent the amount of ammonia emitted in the respective part of the manure handling chain.
value provided by the Atmospheric Emission Inventory Guidebook (IPCC, 1996), as modified by Misselbrook et al. (2000). Some of the differences in emission are therefore due to differences in the model used. However, most of the differences can probably be explained by the different assumptions made concerning the N fertilization rate; DYNAMO uses 140 kg N ha\(^{-1}\) a\(^{-1}\), NARSES 134 kg N ha\(^{-1}\) a\(^{-1}\), DanAm 200 kg N ha\(^{-1}\) a\(^{-1}\) and MAM and FarmMin 400 kg N ha\(^{-1}\) a\(^{-1}\).

Similar NH\(_3\) emissions for housing were calculated with DYNAMO, GAS-EM and FarmMin (Fig. 4). This observation can be explained by the similar EFs, all of which are based on Dutch measurements and data published by Groot Koerkamp et al. (1998).

The greater housing EF used in NARSES is derived from data collected in the UK (Misselbrook et al., 2000). Slurry-based cattle houses in the UK differ from those in other NW European countries in that the floors of UK buildings are usually solid and hence excreta remains on the floor until removed for outside storage by scraping, thus increasing the emission potential of the floor surface.

Apart from DYNAMO, all models use similar EFs for the calculation of the storage emissions (Table 5). The lower emissions calculated by NARSES are therefore largely the effects of the reduced TAN pool resulting from the greater housing losses (Fig. 4). The larger value given for DYNAMO results from the fact that storage emissions are little reduced by partial grazing for the reasons described earlier. The EF for storage is adjusted in such a way that emissions per unit emitting surface area per year remain constant.

NH\(_3\) emissions for slurry application were greater in DanAm, GAS-EM, MAM and FarmMin than in DYNAMO or NARSES, primarily because of larger EFs. The greater EF used in the Dutch models is based on experiments with broadcasted pig slurry (Huijsmans et al., 2001). Because broadcast application has been banned in the Netherlands, no effort has been undertaken to develop a more specific EF for broadcast spreading of cattle slurry. With 10% dry matter, the pig slurry used in the experiments had a much greater dry matter content compared to other European countries. Since dairy cow slurry in the Netherlands usually has lower dry matter concentrations than pig slurry, FarmMin and MAM tend to overestimate the emissions for broadcasting. The EF in DanAm has been derived from the statistical model Alfam (Sogaard et al., 2002) using data for a typical TAN and dry matter content of Danish dairy cow slurry and average Danish meteorological conditions. The smallest emissions were calculated by NARSES and DYNAMO. Both EFs are based on national experiments (Misselbrook et al., 2000; Menzi et al., 1998), with the Swiss EF additionally backed up by the Alfam model using average Swiss TAN and dry matter contents and meteorological conditions. Although the Swiss EF is slightly greater than the EF used in NARSES, the calculated application emissions were similar. This is a consequence of the greater losses in preceding stages in DYNAMO.

### 3.2.2. Fattening pigs

As for dairy cows, substantial differences of total emissions were observed between the pig FF and FN scenarios (Fig. 5). Emissions were greatest with MAM and DYNAMO. In both models, more than half of the emissions originate from slurry application to fields. Both models use the same EFs for pig and cattle. While the EF in MAM is based on experiments conducted by Huijsmans et al. (2001), the EF in DYNAMO has been deduced from national experiments which included only a very small number of pig slurry treatments (Menzi et al., 1998). Both EFs stand in clear contrast to the EFs used in the other models, in which significantly smaller EFs for pig slurry than for cattle slurry are used (Tables 5 and 6). The EFs used in DanAm, GAS-EM and NARSES are primarily based on the observation that pig slurry in these countries has a lower dry matter content than slurry from dairy cows. This is well documented for the EFs used in NARSES, which is based on pig slurry with an average dry matter content of 3.7%, which is well below the value for typical dairy cow slurry (7.8%, Anon., 2003). The EFs used for the calculation of the pig housing emissions ranged from 20% to 31% (Table 6). These differences are primarily a result of the national datasets used and reflect differences in production systems prevalent in the various countries (e.g. partly vs. fully slatted floors, removal frequency of slurry stored in pits below the floor).

### 3.3. NN scenario

#### 3.3.1. Dairy cows

As differences observed among the FN and NN scenarios would be due only to N excretion and TAN content of excreta, this scenario can show how
feeding practices or sources for data on N excretion affect the models.

As compared with the dairy FN scenario, the calculated NH$_3$ emissions for DanAm, GAS-EM and MAM were primarily influenced by the greater N excretion rates resulting from the increased N uptake associated with the greater milk yield of the cows (Table 3, Fig. 6). The relatively small N excretion rate used in DanAm may reflect some bias in the data used to estimate the diet composition. The data were obtained from about 400 farms that were smaller in size than the national average. Due to the lesser milk yield, and the corresponding smaller N excretion, NARSES calculated slightly smaller emissions than in the FN scenario. FarmMin calculated smaller emissions in this scenario than in the FN scenario. When compared with the other models, FarmMin routed a larger portion of

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N excretion through the low-emission path of grazing and hence a smaller portion of N through the high-emission path of housing, storage and slurry application, where the EF during slurry application (67%) is particularly important. The larger portion of total excretion is routed to the pasture because summer rations contain more N than winter rations.

### 3.3.2. Fattening pigs

The substantial differences among the models observed in the pig FN scenario were largely compensated in the NN scenario by the use of model-specific N excretion rates and TAN contents. Total emissions were similar in DYNAMO, GAS-EM, MAM and NARSES (Fig. 7). The overall smallest N excretion and TAN contents in MAM reduced the total emissions as compared with the FN scenario by about 25%. As in the FN scenario, the smallest total emissions observed with DanAm can primarily be explained with the small EF used for manure application. Substantial differences in N excretion rates were observed between the different national values. They primarily reflect different production intensities (slaughter weight, number of production cycles per year) and specific feeding practices prevalent in different countries. The overall lowest N excretion rates used in MAM (Table 4) can primarily be attributed to highly optimized diets with respect to the crude protein content used for pig rearing.

### 4. Conclusions

The results of the FF scenario showed that the different models give similar results when both the N excretion and EF were set to the same values in all models. This indicates that the underlying N flows of the different models are similar. The small differences can be explained by slight differences in the assumptions concerning emissions during the grazing period (partitioning of excretal N between grazing and animal housing; emissions in houses and manure stores when cattle are largely outside) and by the inclusion of mineralization and immobilization in the models. These variations primarily reflect the different perceptions of the builders of models concerning the importance of these additional features. Larger differences among results were observed when emissions were calculated with model-specific EFs and N excretion rates.

Comparing emission inventories on the basis of EFs and N excretion rates can identify differences among models but not the respective reasons. The reasons can be divided into four main types: (1) errors, (2) differences in agricultural practice, (3) differences in the model structure and (4) differences in model parameterisation. The differences in agricultural practice may include different excretion rates resulting from different feeding practice and production intensities (e.g. the protein concentration in the diet, milk yield per cow, growth rate per pig), variations in the types of animal housing, storage and application technology and from...
variations in climate. These factors are fully valid and explain why it is necessary to construct emission inventories at a national or even regional scale. Differences in the model structure may be related to the inclusion of additional sources (e.g. hard standings) or processes (e.g. mineralisation, denitrification in GAS-EM). Differences in parameterization of EFs for what are essentially similar husbandry systems arise due to the access to different sources of information, different interpretations of the same information or different assumption for special situation (e.g. emissions in houses and manure storage when cattle are mainly outside during the grazing season). Such differences are inevitable; given the varied backgrounds of the scientists involved and the ample variation in the experimental data. Nevertheless, differences resulting from differing scientific assumptions are relatively small and mainly arise from differences in national emission measurements and can largely be explained by documented differences in farm management and climatic conditions.

In the course of the congruency-testing exercise some minor weaknesses were identified in all the models tested; some were rectified before production of the results shown here. The scientific debate necessary to understand the variation in results from the different models generated awareness and consensus concerning available scientific data and the importance of some processes (e.g. mineralization). The congruency exercise has therefore already led to a better harmonization of the structure and function of the models tested. In some cases, the consensus relied on work that was only available in reports to funding bodies or in languages other than English, highlighting the need for the collation and publication of such information in a form readily available to an international readership. Complete harmonization of models was not considered desirable, given that the relative importance of the processes involved will vary among countries, due to different agricultural practices and different natural conditions. It should also be remembered that the modeller is always at the mercy of the statistical data available as model input. There is therefore little point in creating a model that uses activity data that are likely to remain unavailable for the foreseeable future. Nevertheless, the considerable effort required to construct, quality assure and document inventory models of the type tested here should encourage anyone contemplating improving their inventory by moving from tier 1 or 2 to tier 3 (IPCC, 1996) to seriously consider using or adapting an existing model.

This comparison of inventory models proved to be a valuable exercise that will contribute to the quality assurance of the inventories. A similar exercise for solid manures would demand more resources, due to the greater complexity of the processes involved. Nevertheless, the experience gained here suggests that it would be worthwhile.

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