# AMMONIA EMISSIONS IN EUROPE: EMISSION COEFFICIENTS AND ABATEMENT COSTS

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(Editor)

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## Foreword

Although public concern about the detrimental impacts of acidification in Europe initially centered on sulfur emissions, it is now widely accepted that nitrogen is also an important factor. Nitrogen deposition results both from emission of nitrogen oxides (mainly from energy combustion) and from emissions of ammonia. The most important sources of ammonia emissions are livestock farming and the use of artificial fertilizers.

An efficient strategy to reduce acidification should not only focus on a single pollutant (e.g. $SO_2$), but should balance reductions in emissions for all substances contributing to the problem. Cost-effective strategies, therefore, require knowledge on the most important emission sources as well as the costs for reducing emissions. Whereas such analysis for $SO_2$ and $NO_x$ emissions has been performed earlier, similar expertise on the potential and costs of reducing ammonia emissions has been lacking for a long time.

This paper, containing the proceedings of a workshop on ammonia emissions in Europe held at the International Institute for Applied Systems Analysis (IIASA) in 1991, makes a first attempt to create a comprehensive international overview on this subject. Thereby, it will provide an important basis for the design of cost-effective strategies for reducing acidification in Europe, balancing reductions in emissions of sulfur dioxide, nitrogen oxides and ammonia.

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# Part I. EMISSION INVENTORIES AND EMISSION COEFFICIENTS

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# AMMONIA EMISSION FOR USE IN ATMOSPHERIC TRANSPORT MODELS

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## Abstract

Ammonia ($NH_3$) concentrations are mainly influenced by local sources, whereas ammonium ($NH_4^+$) in air and in precipitation has more distant sources. Ammonia is due to its short lifetime not transported over long distances while ammonium is. A preliminary estimate of the ammonia emission in Europe is 8.4 x $10^6$ tonne $NH_3$ $a^{-1}$, which is 33% higher than estimated by Buijsman et al. (1987). In areas where the total deposition is dominated by dry deposition of $NH_3$, a detailed emission inventory (with grid elements of 5 x 5 $km^2$) is needed to obtain good results with atmospheric transport models and to estimate the effects of emission reductions and deposition to forests. In other areas a less detailed inventory is sufficient. A pronounced diurnal variation in the emission rate for $NH_3$ was found. Theoretical estimates of the seasonal variation in the emission rate are not in agreement with the measured variation.

# 1 Introduction

Ammonia ($NH_3$) and ammonium ($NH_4^+$) are important atmospheric components. $NH_3$ is the most abundant alkaline component in the atmosphere. A substantial part of the acid in the atmosphere, as generated by the oxidation of sulphur dioxide and nitrogen oxides, is neutralized by $NH_3$. As a result, $NH_4^+$ -the reaction product of $NH_3$- is a major component of a reaction product of $NH_4^+$ in aerosols and precipitation. $NH_3$ and $NH_4^+$ act as fertilizers, and deposition of these substances has unfortunate effects (Roelofs et al., 1985) and can lead to a change in the composition of the vegetation (Nilsson and Grennfelt, 1988). Oxidation of $NH_4^+$ in the soil leads to acidification of the soil. For these reasons the interest in the atmospheric behaviour of $NH_3$ and $NH_4^+$ is increasing.

$NH_3$ is mainly emitted from animal manure, but it also derives from the production and application of fertilizers (Buijsman et al., 1987). $NH_4^+$ is not emitted in significant quantities and virtually all $NH_4^+$ in the atmosphere originates from $NH_3$.

The requirements of an emission inventory for $NH_3$ depend very much on the specific atmospheric transport model in which the emission inventory is to be used. Further, the setup of a model depends in its turn on what kind of results are required and not least on the atmospheric behaviour of the components involved. Therefore the atmospheric behaviour of $NH_x$ ($NH_3$ + $NH_4^+$ aerosol) is discussed first, mainly by using information from Asman and van Jaarsveld (1990).

# 2 Atmospheric behavior of $NH_3$

$NH_3$ originates mainly from many, widely distributed, low-level sources. Within a few hundred metres of such sources, the $NH_3$ concentrations are relatively high near the earth's surface. Therefore there is a large dry deposition of $NH_3$ near the source (Figure 1). At a distance of a few hundred metres from the source, $NH_3$ will have become significantly diluted, and consequently the $NH_3$ concentration and dry deposition rate will have dropped. At such distances the contribution of a single source vanishes in comparison with the background concentration. Large dry deposition of $NH_3$ close to sources is a phenomenon which is not important for high sources such as industrial ($NH_3$ or $SO_2$) sources.

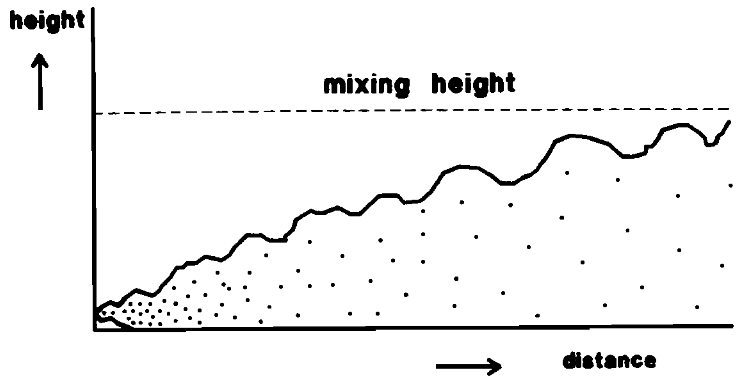


Figure 1. Plume as a function of distance from a low-level source.

The climatologically averaged dry deposition velocity of $NH_3$ has been found to be about $1.6 \times 10^{-2}$ m $s^{-1}$ over heather and purple moor grass (Duyzer et al., 1987). In a second-generation atmospheric transport model for $NH_x$ (Asman and van Jaarsveld, 1990; Asman and van Jaarsveld, 1992) this value of the dry deposition velocity was reduced somewhat, to take into account the low dry deposition velocities over agricultural surfaces, which themselves contain substantial $NH_3$ concentrations. A climatologically averaged value of about 1.2 x $10^{-2}$ m $s^{-1}$ was used. For forests the dry deposition velocity is considerably higher than this value.

In the model of Asman and van Jaarsveld (1990) the dry deposition velocity is variable; it is a function of meteorological circumstances, as in reality. After $NH_3$ is fully mixed throughout the surface mixing layer of the atmosphere the reduction of the $NH_3$ concentrations due to dry deposition is about 6% $h^{-1}$.

The dry deposition velocity of $NH_4^+$ aerosol is much lower than the dry deposition velocity of $NH_3$. It is on the average about 1.8 x $10^{-3}$ m $s^{-1}$ lower, but there is a large uncertainty involved. In the model an effective dry deposition velocity of $NH_4^+$ aerosol of 1.4 x $10^{-3}$ m $s^{-1}$ was adopted, resulting in a reduction of the $NH_4^+$ aerosol concentrations due to dry deposition of about 0.7% $h^{-1}$.

Both $NH_3$ and $NH_4^+$ aerosol are removed very efficiently by precipitation at a rate of the order of 70% $h^{-1}$. But as it rains only 5-10% of the time, removal by dry deposition can be as effective as removal by wet deposition.

$NH_3$ reacts with acid in the atmosphere (mainly sulphuric acid aerosol and nitric acid) whereby $NH_4^+$ aerosol is produced at a rate of about 30% $h^{-1}$. This is a rather high rate compared to the oxidation rates for sulphur dioxide or nitrogen oxides, which are only a few percent per hour. The high reaction rate for $NH_3$ with acids has some very important consequences, one being that $NH_3$ is not transported over long distances, as it is converted to $NH_4^+$ aerosol, which is not removed very effectively by dry deposition and therefore can be transported over long distances (if it does not rain). Another consequence is that the contribution from one country to another will be mainly in the form of $NH_4^+$ aerosol and $NH_4^+$ in precipitation and <u>not</u> in the form of $NH_3$.

The occurrence of many low-level sources and the relative short lifetime of $NH_3$ results in a very high spatial variability in $NH_3$ concentrations (Asman et al., 1989). This makes it almost impossible to measure the average $NH_3$ concentration for a country. It is for such purposes that atmospheric transport models can be used as an interpolation tool together with measurements at a few locations having widely differing concentrations, which are used to test the model results. As the $NH_4^+$ concentrations in air and precipitation vary relatively little, one can determine $NH_4^+$ concentration gradients over a country from measurements and then compute the average concentration for this country.

Figure 2 shows the setup of an atmospheric transport model for $NH_x$. Model results show that 44% of the emitted $NH_3$ is dry deposited as $NH_3$, 6% is wet deposited as the contribution of $NH_3$ to the wet deposition of $NH_x$ (measured as $NH_4^+$), 14% is dry deposited as $NH_4^+$ aerosol and 36% is wet deposited as the contribution of $NH_4^+$ aerosol to the wet deposition of $NH_x$. Figures 3 shows that there are two important removal mechanisms for $NH_x$: dry deposition of $NH_3$ close to the source and wet deposition of $NH_4^+$ aerosol at distances beyond 100 km from the source.

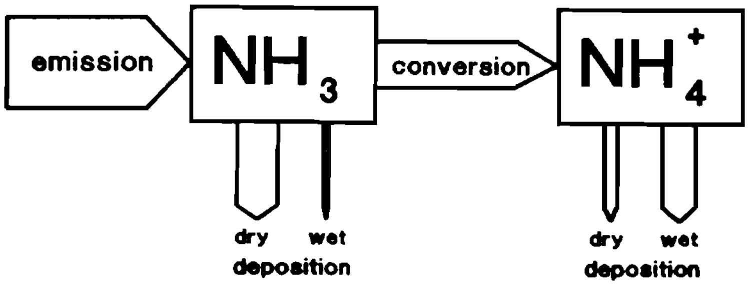


Figure 2. Schematic overview of the TREND model for $NH_x$ (apart from the diffusion part). The width of the arrow indicates the relative importance of the various processes as computed with the TREND model for Western Europe.

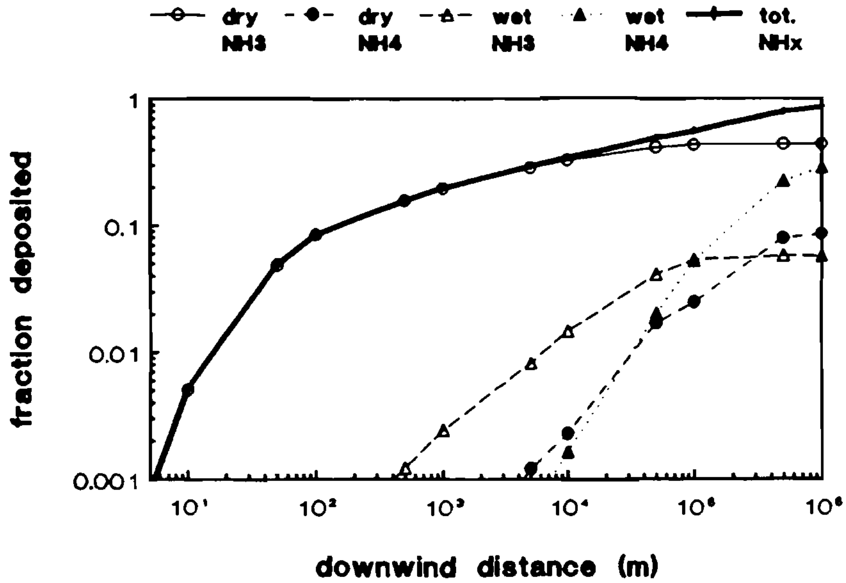


Figure 3. Cumulative depositions of different forms of $NH_x$ as a function of downwind distance, integrated over all wind directions.

# 3 Emission inventory for Europe

Good emission inventories are difficult to construct and much boring work has to be done before they are ready for use. The following stages of the process can be distinguished:

1. To determination of the emission factors. This is done e.g. for $NH_3$ by using the results of agricultural research. But these results are usually valid for only the experiment under consideration. Different experiments often yield very different results. It is necessary to generalize to be able to make an inventory for a larger area than a specific experimental field. Normally some important information is lacking, which has to be estimated by making some assumptions.
2. Acquisition of statistical information, e.g. on livestock and consumption and production of fertilizers. If the inventory has to be used in atmospheric transport models, information on the geographical distribution of the sources is needed, which cannot be found in international statistics. Therefore, information should be obtained from national or sub-national authorities. For different countries there may exist different subdivisions of livestock or fertilizers, which may cause some trouble in further calculations.
3. Acquisition of information on the exact geographical location of administrative units (municipalities, provinces, agricultural areas). This is needed in order to translate the emission for administrative units to emission on a regular grid which is often needed for model calculations. This can be a very time-consuming part and can easily take one man-year for a detailed (5 x 5 $km^2$) emission inventory for a country or for a less detailed emission inventory for Europe.
4. Testing the inventory by using it in an atmospheric transport model. This is an often forgotten stage. Of course there exists an uncertainty in the model results because of the uncertainty in the values of the model parameters. But if the model results are too different from measurements there must be something wrong with the emission inventory: maybe wrong emission factors were used or important sources are lacking.

Buijsman et al. (1987) made the first gridded emission inventory for Europe. At that moment not many people were interested in atmospheric $NH_3$ emissions and there was much less work done in that area than nowadays. Buijsman et al. (1987) computed emissions from livestock using information on the number of animals for subcategories (e.g. "cattle 1-2 years") and emission factors for subcategories. In this way emissions were computed for different countries, which did not show exactly the same emission factor per animal category ("cattle" in this case). This method of calculation had some reality because the age or weight distributions for animal categories can be different in different countries. But part of the differences found, resulted simply from the fact that the subdivision of animal categories in different countries was different.

Asman and Janssen (1987) used the emissions of Buijsman et al. (1987) in their long-range transport model, increased by 20% to take some other emissions which were not part of the inventory into account. These increased emissions are used in the current EMEP emission inventory. Buijsman (1987) already stated that the Buijsman et al. (1987) emissions were conservative one and that the emissions could well be higher by 25-35%.

At the moment much more information is available on $NH_3$ emission. Asman (1990) made therefore a new emission inventory for Europe, which until now only exists in a draft version, which hopefully will be revised soon, leading to minor changes. The reason why this takes so much time is that almost all this work has to be done by the author in his sparetime. This report will be published by the National Institute of Public Health and Environmental Protection (RIVM), Bilthoven, the Netherlands. The emission factors for livestock were derived from recent Dutch emission factors for animal subcategories (De Winkel, 1988), apart from the emission factors for horses and camels (Table 1).

**Table 1.** Annually averaged overall emission factors for animal categories (kg $NH_3$ animal$^{-1}$ a$^{-1}$).
(For computational reasons the data is given more accurately than actually known).

| Category | This survey | Möller and Schieferdecker (1989) |
|---|---|---|
| Cattle | 25.1180 | 26.8 |
| Pigs | 4.8241 | 6.3 |
| Poultry | 0.3206 | 0.27 |
| Horses (incl. ponies) | 12.5000 | 18.2 |
| Sheep(incl. goats) | 1.9075 | 3.6 |
| Camels | 23.3000 | — |

These emission factors were then used for all European countries together with statistical information on the number of animals and the consumption of fertilizers for the year 1987. This is, of course, by no means correct, as emission factors will depend on local agricultural practise (including the duration of the period when the cattle are on the stable), local meteorology and local soil conditions. For most countries, however, only part of this information is available, which makes it difficult to take these local factors into account.

For fertilizers the emission factors of Buijsman et al. (1987) were used in the draft version (Table 2). In the definitive version other emission factors will be used, which will lead to lower emissions from fertilizers. Table 3 shows the $NH_3$ emission for different countries. The ratio of the here computed emissions and the emissions of Buijsman et al. (1987) is presented in Table 4. The estimated uncertainty in these emissions is at least 30-40%. For some countries the emission is much higher in the new inventory than in the old one. This is partly caused by the fact that not any longer emission factors for animal subcategories are used, as Buijsman et al. (1987) did. At the moment many countries (Asman, 1990) are adopting rules to reduce emissions of $NH_3$ to the atmosphere. This will greatly influence emissions in the future.

**Table 2.** Emission factors for N-fertilizers (% loss of N content).

| Fertilizer | Emission factor |
|---|---|
| Ammoniumnitrate | 10 |
| Ammoniumphosphate | 5 |
| Ammoniumsul.nitrate | 12.5 |
| Ammoniumsulphate | 15 |
| Urea | 10 |
| Complex N | 5 |
| Other N | 5 |
| Not specified N | 5 |

**Table 3.** $NH_3$ emission in European countries (tonne $NH_3$ $a^{-1}$).

| | Cattle | Pigs | Poultry | Horses | Sheep | Camels | Tot.a | Fertilizer | $NH_3$ ind | Total |
|---|---|---|---|---|---|---|---|---|---|---|
| Albania | 15598 | 1105 | 1603 | 525 | 3847 | 0 | 22678 | 9107 | 58 | 31844 |
| Austria | 66236 | 18336 | 4488 | 550 | 3706 | 0 | 93317 | 13111 | 191 | 106619 |
| Belgium | 76861 | 26942 | 6828 | 300 | 353 | 0 | 111285 | 11372 | 591 | 123247 |
| Bulgaria | 42801 | 19537 | 12503 | 1512 | 19783 | 0 | 96137 | 26714 | 602 | 123453 |
| CSFR | 127424 | 32963 | 15709 | 575 | 2205 | 0 | 178876 | 39234 | 538 | 218647 |
| Denmark | 65759 | 43846 | 4879 | 400 | 134 | 0 | 115018 | 28578 | 105 | 143701 |
| Finland | 37300 | 6315 | 2244 | 487 | 126 | 0 | 46472 | 14586 | 188 | 61247 |
| France | 572766 | 57899 | 69889 | 3875 | 22072 | 0 | 726500 | 245838 | 1272 | 973610 |
| Germany D.R. | 145785 | 61941 | 16029 | 1312 | 5089 | 0 | 230157 | 43040 | 773 | 273971 |
| Germany F.R. | 384431 | 118204 | 24365 | 4600 | 2709 | 0 | 534309 | 182898 | 888 | 718095 |
| Greece | 18688 | 5914 | 9938 | 813 | 31832 | 0 | 67185 | 44019 | 247 | 111451 |
| Hungary | 43329 | 41907 | 21159 | 1187 | 4498 | 0 | 112080 | 66625 | 552 | 179256 |
| Ireland | 141314 | 4728 | 2565 | 688 | 5623 | 0 | 154917 | 33300 | 164 | 188381 |
| Italy | 221516 | 44758 | 35906 | 3162 | 21253 | 0 | 326595 | 107176 | 956 | 434727 |
| Luxemburg | 5400 | 367 | 31 | 25 | 15 | 0 | 5838 | 1003 | 0 | 6841 |
| Netherlands | 118306 | 67267 | 30380 | 812 | 2297 | 0 | 219062 | 55465 | 1172 | 275698 |
| Norway | 24239 | 3580 | 1282 | 200 | 1761 | 0 | 31061 | 6284 | 340 | 37685 |
| Poland | 264317 | 89467 | 18594 | 14263 | 9059 | 0 | 395700 | 164485 | 1018 | 561203 |
| Portugal | 27303 | 14086 | 5771 | 362 | 11750 | 0 | 59273 | 16600 | 145 | 76018 |
| Romania | 181478 | 70967 | 43921 | 8500 | 37581 | 0 | 342447 | 43471 | 1457 | 387375 |
| Spain | 124435 | 67537 | 17312 | 3012 | 38619 | 0 | 250915 | 112906 | 712 | 364533 |
| Sweden | 41822 | 11433 | 3526 | 712 | 773 | 0 | 58266 | 15296 | 137 | 73699 |
| Switzerland | 47222 | 9518 | 1924 | 600 | 853 | 0 | 60116 | 8002 | 26 | 68144 |
| UK | 313372 | 38375 | 41677 | 2187 | 49660 | 0 | 445272 | 101454 | 1016 | 547741 |
| Yugoslavia | 127022 | 40807 | 25006 | 4800 | 15227 | 0 | 212862 | 30721 | 336 | 243920 |
| Turkey | 325027 | 58 | 18594 | 7750 | 104740 | 70 | 456239 | 116900 | 0 | 573139 |
| USSR(*) | 1100169 | 160642 | 133911 | 20487 | 39676 | 0 | 1454884 | 88167 | 0 | 1543052 |
| Whole area | 4659920 | 1058498 | 570034 | 83700 | 435239 | 70 | 6807460 | 1626352 | 13484 | 8447298 |

(*). **Only the following republics: Ukraine, White-Russia, Georgia, Azerbajdzjan, Moldavia, Lithunia, Latvia, Armenia and Estonia.**

**Table 4.** Ratio this emission vs. emission computed by Buijsman et al. (1987).

| | Cattle | Pigs | Poultry | Horses | Sheep | Camels | Tot.a | Fertil | $NH_3$ ind | Total |
|---|---|---|---|---|---|---|---|---|---|---|
| Albania | 1.79 | 3.12 | 2.48 | 1.29 | 0.65 | 0.00 | 1.42 | 2.07 | 1.00 | 1.56 |
| Austria | 1.24 | 3.54 | 1.77 | 1.69 | 6.12 | 0.00 | 1.50 | 1.47 | 1.00 | 1.50 |
| Belgium | 1.34 | 2.53 | 1.35 | 1.05 | 1.02 | 0.00 | 1.51 | 2.63 | 1.00 | 1.57 |
| Bulgaria | 1.28 | 1.81 | 1.18 | 1.34 | 0.57 | 0.00 | 1.05 | 0.85 | 1.00 | 1.01 |
| CSFR | 1.41 | 1.63 | 1.23 | 1.39 | 0.72 | 0.00 | 1.41 | 1.01 | 1.00 | 1.31 |
| Denmark | 1.06 | 2.17 | 1.21 | 0.91 | 0.75 | 0.00 | 1.32 | 1.23 | 1.00 | 1.31 |
| Finland | 1.20 | 1.42 | 0.96 | 2.22 | 0.45 | 0.00 | 1.21 | 3.71 | 1.00 | 1.44 |
| France | 1.28 | 2.26 | 1.43 | 1.46 | 0.49 | 0.00 | 1.28 | 1.89 | 1.00 | 1.39 |
| Germany D.R. | 1.41 | 1.81 | 1.20 | 1.71 | 0.74 | 0.00 | 1.45 | 1.02 | 1.00 | 1.36 |
| Germany F.R. | 1.51 | 2.48 | 1.22 | 1.23 | 0.77 | 0.00 | 1.63 | 5.16 | 1.00 | 1.97 |
| Greece | 1.24 | 2.06 | 1.28 | 0.20 | 0.82 | 0.00 | 0.95 | 1.79 | 1.00 | 1.19 |
| Hungary | 1.24 | 1.64 | 1.79 | 1.05 | 0.46 | 0.00 | 1.35 | 1.57 | 1.00 | 1.42 |
| Ireland | 1.48 | 2.40 | 1.20 | 0.80 | 0.60 | 0.00 | 1.41 | 6.23 | 1.00 | 1.63 |
| Italy | 1.35 | 2.30 | 1.02 | 1.27 | 0.67 | 0.00 | 1.29 | 1.05 | 1.00 | 1.23 |
| Luxemburg | 1.30 | 2.68 | 0.99 | 1.88 | 1.47 | 0.00 | 1.34 | 4.13 | 1.00 | 1.49 |
| Netherlands | 1.36 | 3.13 | 1.91 | 4.31 | 0.61 | 0.00 | 1.71 | 4.55 | 1.00 | 1.94 |
| Norway | 1.29 | 1.88 | 1.11 | 0.89 | 0.33 | 0.00 | 1.14 | 0.95 | 1.00 | 1.10 |
| Poland | 1.22 | 1.62 | 1.08 | 0.87 | 0.75 | 0.00 | 1.25 | 2.05 | 1.00 | 1.41 |
| Portugal | 1.28 | 2.03 | 5.36 | 1.64 | 1.36 | 0.00 | 1.55 | 2.28 | 1.00 | 1.66 |
| Romania | 1.59 | 2.02 | 1.56 | 1.59 | 0.69 | 0.00 | 1.44 | 0.82 | 1.00 | 1.33 |
| Spain | 1.53 | 2.13 | 1.55 | 1.33 | 0.77 | 0.00 | 1.42 | 2.28 | 1.00 | 1.61 |
| Sweden | 1.26 | 1.51 | 1.19 | 1.33 | 0.57 | 0.00 | 1.28 | 2.74 | 1.00 | 1.43 |
| Switzerland | 1.14 | 2.25 | 1.24 | 1.24 | 0.98 | 0.00 | 1.24 | 2.22 | 1.00 | 1.31 |
| UK | 1.75 | 2.67 | 1.12 | 1.66 | 0.66 | 0.00 | 1.45 | 1.13 | 1.00 | 1.37 |
| Yugoslavia | 1.19 | 2.78 | 1.50 | 0.89 | 0.66 | 0.00 | 1.28 | 1.60 | 1.00 | 1.25 |
| Turkey | 0.83 | 0.67 | 1.24 | 1.07 | 0.49 | 0.00 | 0.72 | 2.48 | 1.00 | 0.84 |
| USSR(*) | 1.43 | 1.71 | 1.34 | 1.05 | 0.61 | 0.00 | 1.39 | 0.42 | 1.00 | 1.23 |
| Whole area | 1.32 | 2.05 | 1.34 | 1.07 | 0.62 | 0.25 | 1.30 | 1.49 | 1.00 | 1.33 |

(*). **Only the following republics: Ukraine, White-Russia, Georgia, Azerbajdzjan, Moldavia, Lithunia, Latvia, Armenia and Estonia.**

# 4 How detailed should an emission inventory be?

Figures 4 and 5 show the geographical distribution of the $NH_3$ emission for Europe and for Belgium, the Netherlands and the Western part of the F.R.G. (The emissions for the Netherlands in Figure 5 were given by Erisman, 1989). It is obvious from these figures that there exist large gradients in the emission density. For components which have mainly more distant sources like $NH_4^+$ aerosol and $NH_4^+$ in precipitation (at least the most important contribution of $NH_4^+$ aerosol to it), an emission inventory with a resolution of 75 x 75 $km^2$ (IE grid) or 150 x 150 $km^2$ will lead to reasonable model results (Asman and van Jaarsveld, 1990).

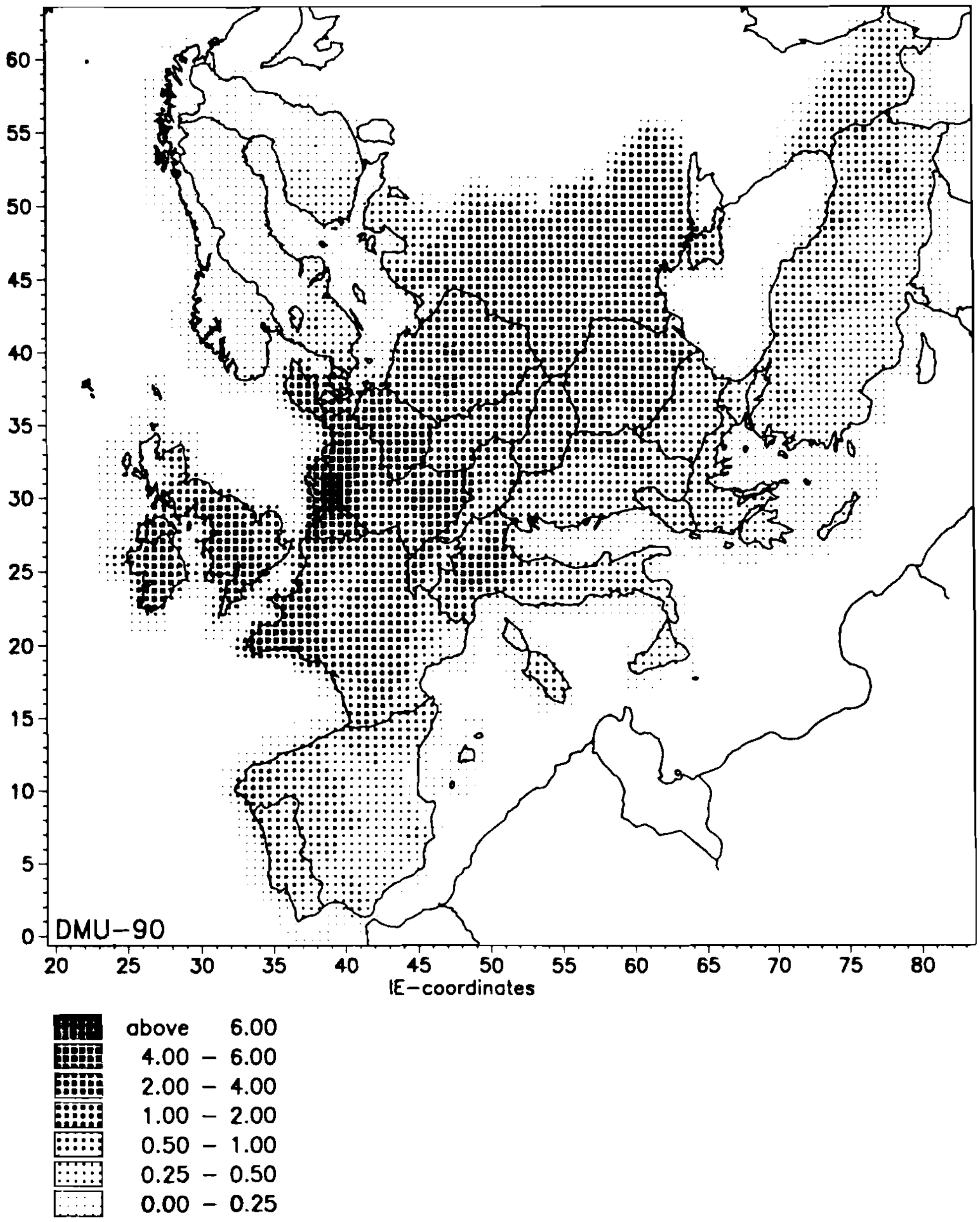


Figure 4. Emission density of $NH_3$ in Europe (tonne $NH_3$ $km^{-2}$ $a^{-1}$).

For $NH_3$ this is not the case (Figure 6). To be able to reproduce measured $NH_3$ concentrations an emission inventory is needed with a resolution of at least 5 x 5 $km^2$. One could though compute a correct average dry deposition of $NH_3$ for a larger grid element with a model. The problem in this case is, however, that it is nearly impossible to check this computed value with measurements only. In that case a station would be needed for every 5 x 5 $km^2$. Moreover, the $NH_3$ concentration would be underestimated for the land area for those grid elements, which also partly cover sea areas.

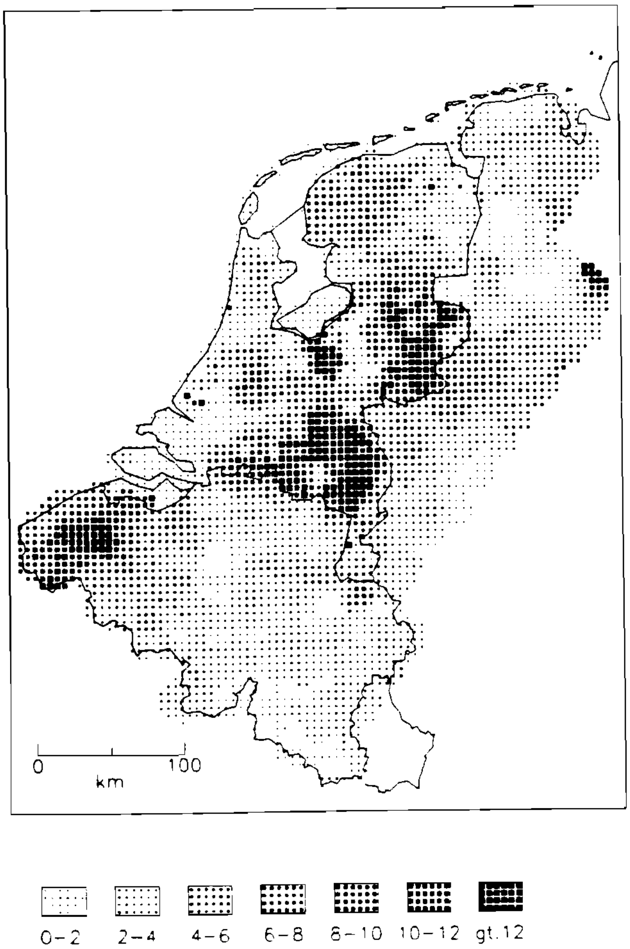


Figure 5. Emission density of $NH_3$ in the Netherlands (tonne $NH_3$ $km^{-2}$ $a^{-1}$).

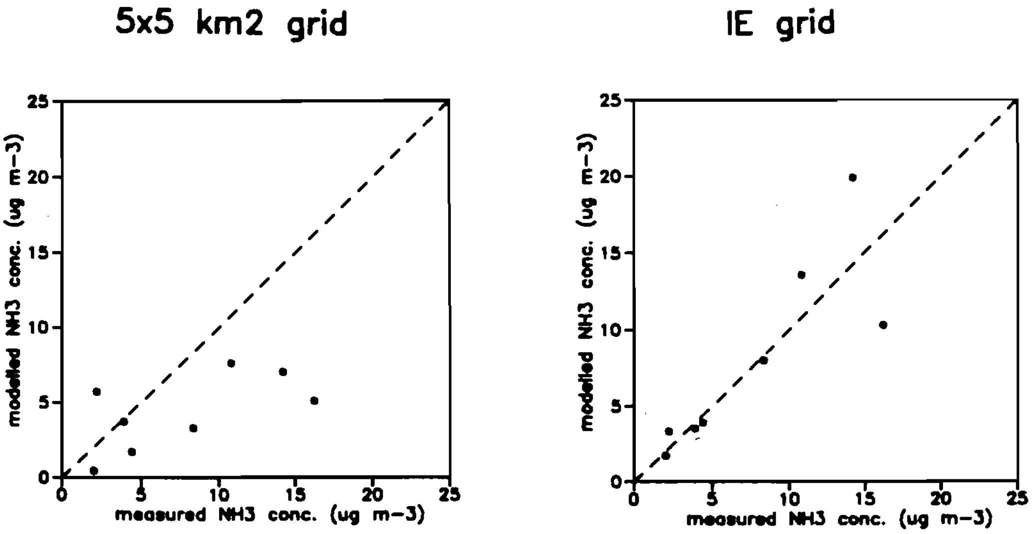


Figure 6. Modelled vs. measured concentrations of $NH_3$ in the Netherlands, computed with emissions on 5x5 $km^2$ grid and with emissions on IE grid (75x75$km^2$)

In areas where wet deposition is more important than dry deposition, e.g. Sweden, the total deposition to a small area could be modelled satisfactory with a less detailed emission inventory. In and near areas with a high emission density like the Netherlands or Denmark dry deposition of $NH_3$ is dominant, and the total deposition to a small area can only be computed with a detailed inventory. If one would like to calculate the effects of emission reductions on the deposition to a forest this could be done with a less detailed emission inventory for Sweden, but for areas like the Netherlands, Germany, Belgium and Denmark, an emission inventory on a 5 x 5 $km^2$ grid is needed to do this.

# 5 Diurnal and seasonal variations in the emission rate

Asman (1990) found a substantial diurnal variation in the emission rate, which could be explained from diurnal variations in the aerodynamic resistance and temperature (Figure 7). Adoption of such a diurnal variation results in a 20% reduction of the annually averaged concentrations of $NH_3$ and $NH_4^+$ aerosol.

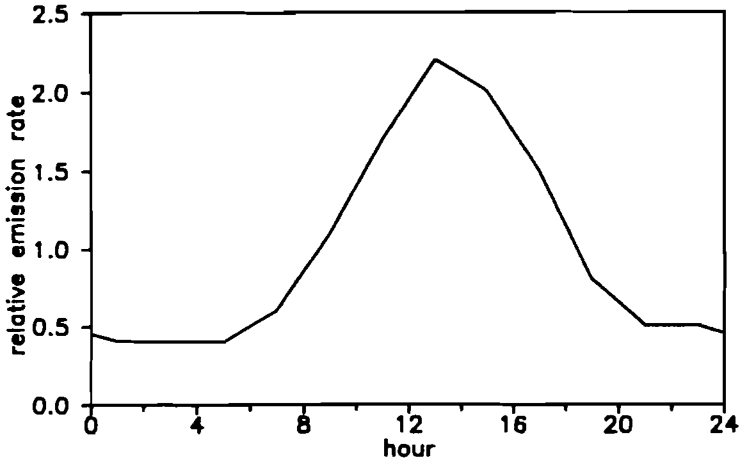


Figure 7. Average diurnal relative variation in the $NH_3$ emission rate (average value = 1.).

It is important to know the seasonal variation in the emission rate, especially for the calculation of effects on ecosystems. This was done for the Netherlands from information on the period the cattle are in the stable, manure is spread, using appropriate emission factors for each activity. The seasonal variation found in this way was very pronounced showing up to maximal a factor 5 difference in emission rate for different months. This variation was, however, much larger than the variation derived from measurements (maximal a factor 2). This means that still some more research has to be done before the seasonal variation can really be understood.

# Acknowledgement

This investigation was supported by the Netherlands Directorate-General of the Environment within the framework of the project 228471 "Acid Deposition" and by the Danish National Agency of Environmental Protection within the framework of the projects "Konsekvensberegninger" and "Marine Research Programme '90".

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# EXPERIENCE FROM MODELLING OF LONG-RANGE TRANSPORT OF REDUCED NITROGEN AT MSC-W OF EMEP

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## Abstract

Results from model-calculated concentrations of reduced nitrogen (ammonia and ammonium) are shown. The calculations form parts of the acid deposition model at the Meteorological Synthesizing Centre - West operating in routine. By comparing measurements, calculations and the emissions used in the calculations, it is demonstrated that there are large potentials for improvements in the calculated budgets by increasing the quality of the emission data for ammonia in Europe. A major weakness of the measured data pointed out is the closeness of the measurement sites to rural emissions of ammonia.

# 1 The model

The MSC-W acid deposition model calculates transboundary budgets of oxidised sulphur and nitrogen as well as reduced nitrogen routinely for the European Monitoring and Evaluation Programme (EMEP). It includes 10 different components in air, three of which are anthropogenically emitted: $SO_2$, $NO_x$ and $NH_3$. The most important role of reduced nitrogen (ammonia and ammonium) in the model chemistry is to determine the fraction between gaseous nitric acid and particulate ammonium nitrate. This indirectly influences the transport distance of oxidised nitrogen, since nitric acid gas is very efficiently dry deposited while particulate nitrate has a very small dry deposition speed.

There are two ways to form ammonium from ammonia in the model. If there is any free gaseous ammonia ($NH_3$) and liquid sulphuric acid in the air, ammonium sulphate is immediately formed until there either is no ammonia or sulphuric acid left. If the latter is used up first, some ammonia is left and this then enters into equilibrium with gaseous nitric acid ($HNO_3$) to possibly form ammonium nitrate. However, depending on the air's relative humidity and temperature, some ammonium nitrate may evaporate to form gaseous ammonia and nitric acid instead.

The transport part of the model is solved by using 4 days long back trajectories, all ending up in grid-points in a regular grid or in measurement points. The model consists of one layer describing the well mixed, boundary layer. Details about the model can be found in Iversen *et al.* (1991).

# 2 Input data

The model needs input of meteorological data as well as emissions. The meteorological data are taken from short-term prognoses of the numerical weather prediction model with resolution 150 km, except for precipitation over land and mixing heights which are analysed directly from observations. Inputs are needed with six-hourly frequency.

Emission data are as far as possible those calculated by the different countries in Europe and submitted to the ECE secretariate annually. The quality of these data for $SO_2$ and $NO_x$ is believed to be reasonably good with a few exceptions. For $NH_3$, however, the emissions are probably much worse. Only five countries have supplied official data for a recent year, and only one of these data sets is in gridded form. Other data are national totals, and these data are distributed in space in the same way as given in Buijsman *et al.* (1985) (see also Buijsman *et al.*, 1987). In countries that have not given any data (23 countries), the data from Buijsman *et al.* (1985) are used directly after being multiplied with 1.2 (informal communication with the authors). Even these do not cover the whole domain, and in the Russian part of USSR a subjective judgement has been used to produce data that should be consistent with those estimated by Buijsman *et al.* (1985) in Belorussia and Ukraine.

# 3 Calculation results

Figure 1 shows a map for calculated deposition of reduced nitrogen ($NH_3$ + $NH_4^+$). It clearly shows very large gradients as one moves away from the major emission sources. The reason for this is that ammonia, being the primary component, is efficiently deposited dry as well as wet. This causes shorter typical transport distances than for sulphur and in particular for oxidised nitrogen.

Figure 2 shows a scatter plot of model calculated towards observed concentrations of ammonium in precipitation for a selection of stations which

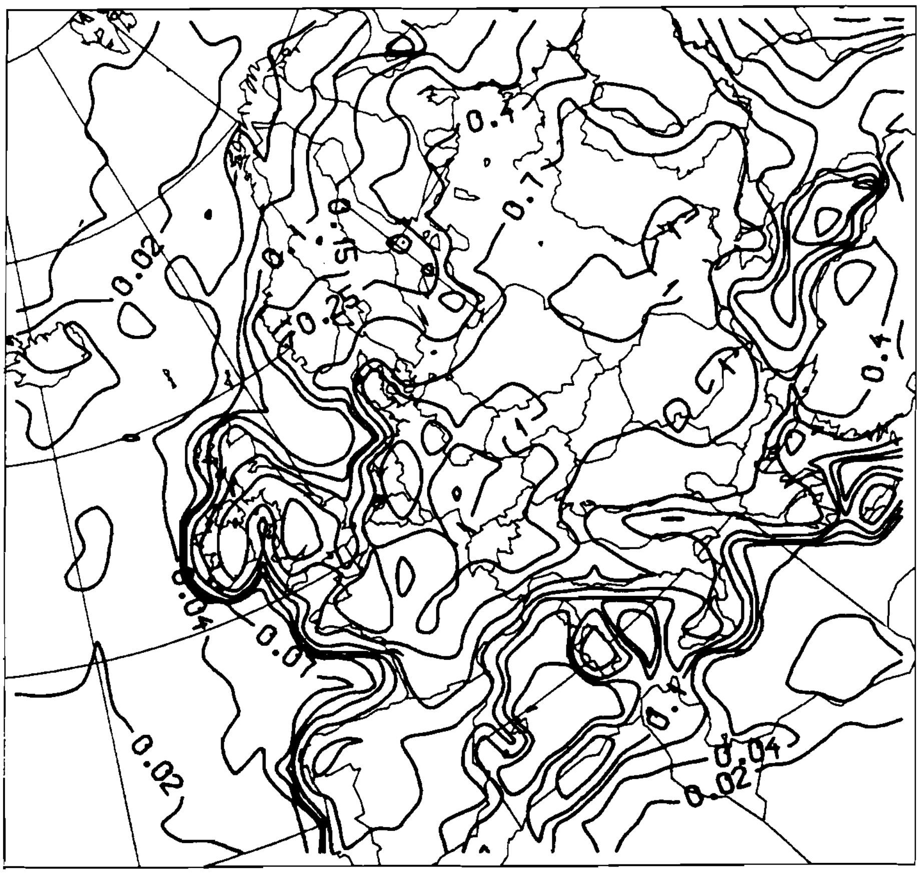



Figure 1. Model estimated annual deposition of reduced nitrogen for 1988.
Isolines for 0.01, 0.02, 0.04, 0.07, 0.1, 0.15, 0.25, 0.4, 0.7, 1.0, 1.5, 2.5, 4.0, 7.0, 10.0 g(N)/m$^2$.

have an observation coverage of more than 25%. The plot reveals a clear tendency of the model to underestimate the measured concentrations. In particular, there are four measurement sites with very large measured concentrations which govern the regression line to a large degree. These stations are SU 6 and SU 7 in the Soviet Union, SE 8 in Sweden, and AT 2 in Austria. Since ammonia is efficiently dissolved in precipitation to form ammonium with large efficiency, ammonium in precipitation is expected to show a certain dependency with the size of close emission sources of ammonia, even if the long-range transported part of the reduced nitrogen of course precludes a linear relationship. Unfortunately, very few EMEP stations report measurements of ammonia, and hence ammonium in precipitation has to be used as a control of the model's quality and of the emission data. Figure 3 shows

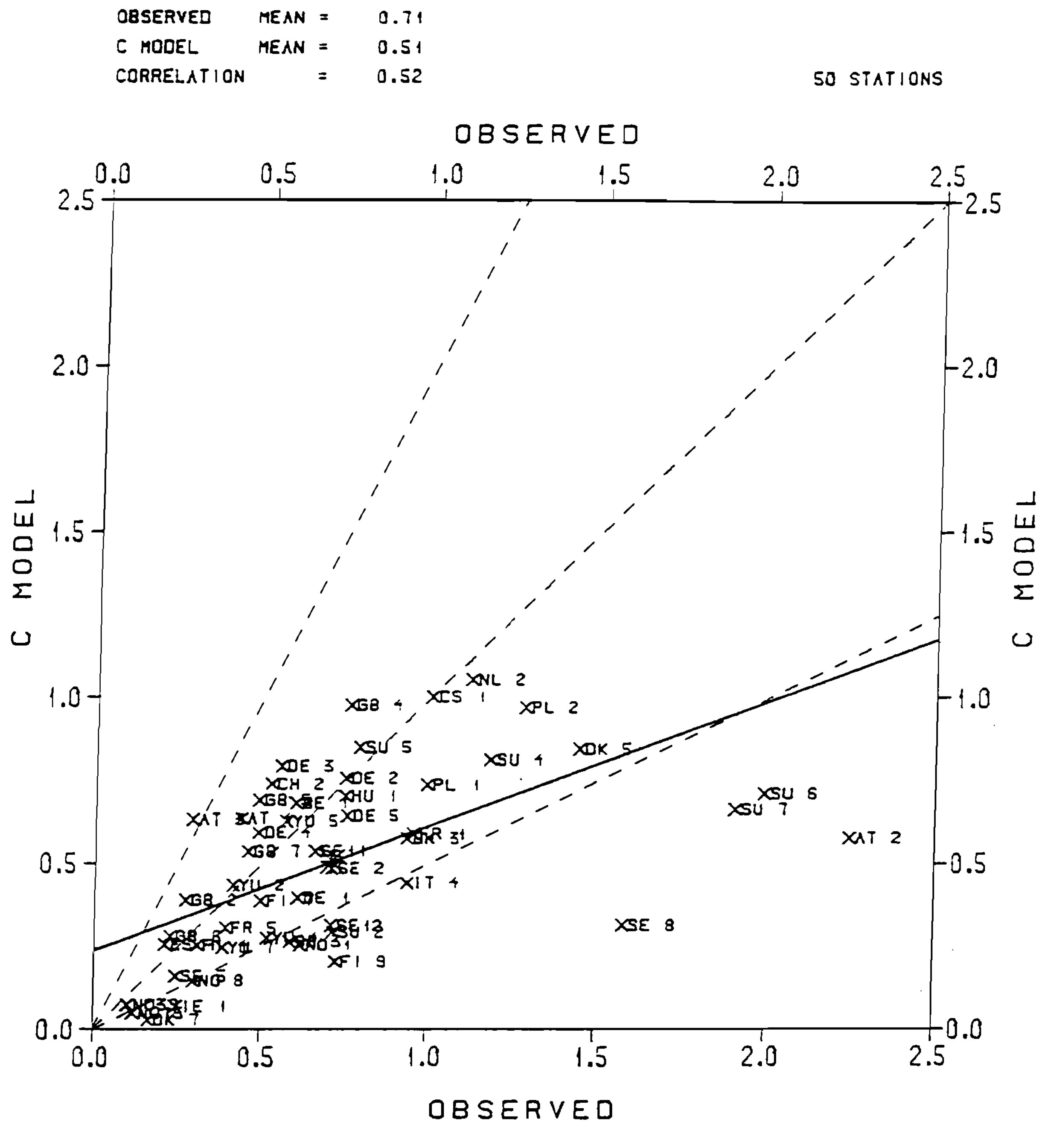
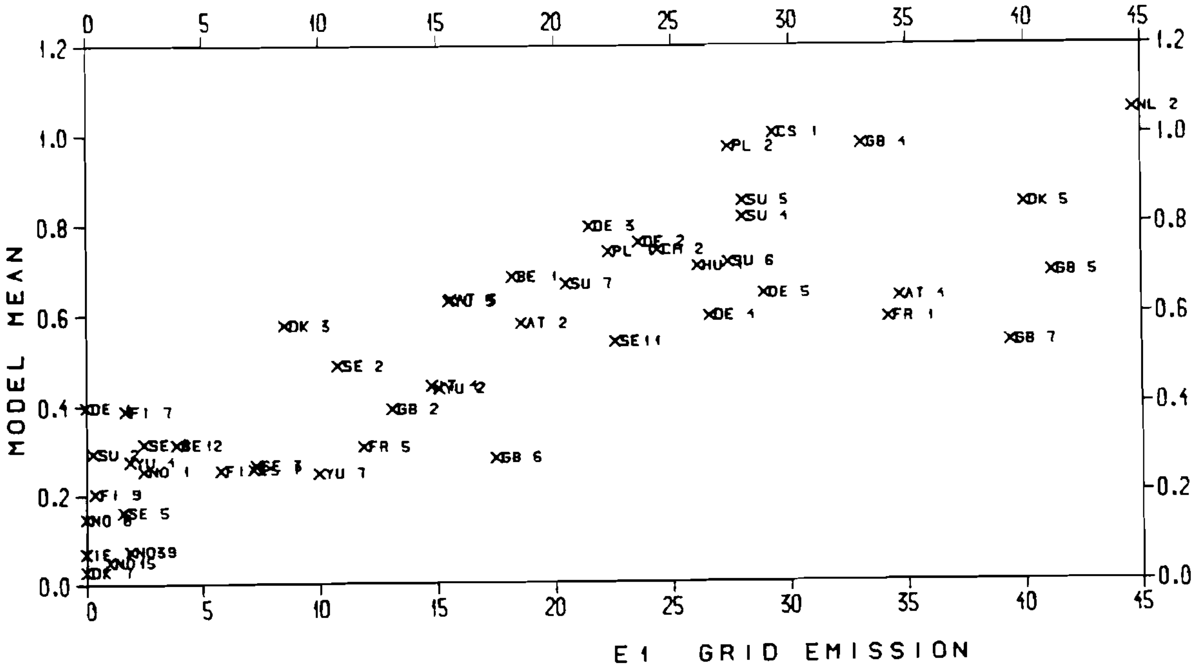
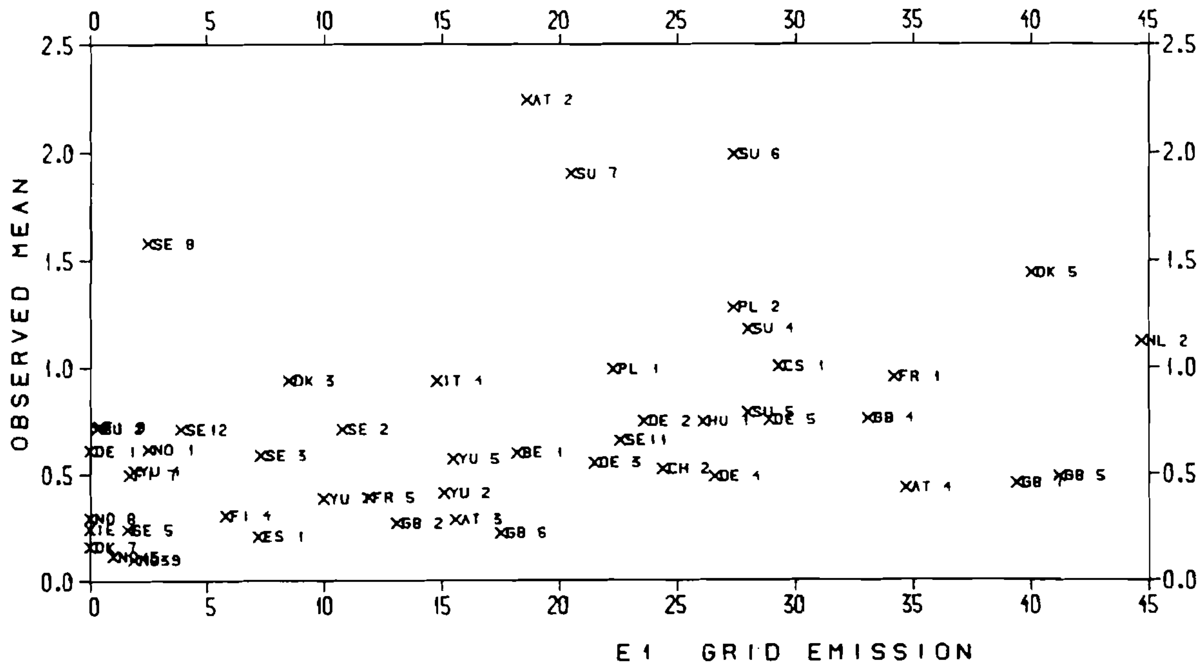



Figure 2. Modelled versus observed concentrations of ammonium in precipitation for stations with data coverage more than 25%, unit mg(N)/l. The dashed lines show perfect agreement and disagreement with a factor of 2. The full line represents optimal linear regression.
See Iversen *et al.* (1991) for station codes.




a) Model calculated concentrations




b) Observed concentrations

Figure 3. Annually averaged concentrations of ammonium in precipitation (unit: mg(N)/l) at EMEP sites with more than 25 % data coverage, as a function of the emitted amount of ammonia (unit: kt($NH_3$)a) in the grid-square in which the sites are situated.
See Iversen *et al.* (1991) for station codes.

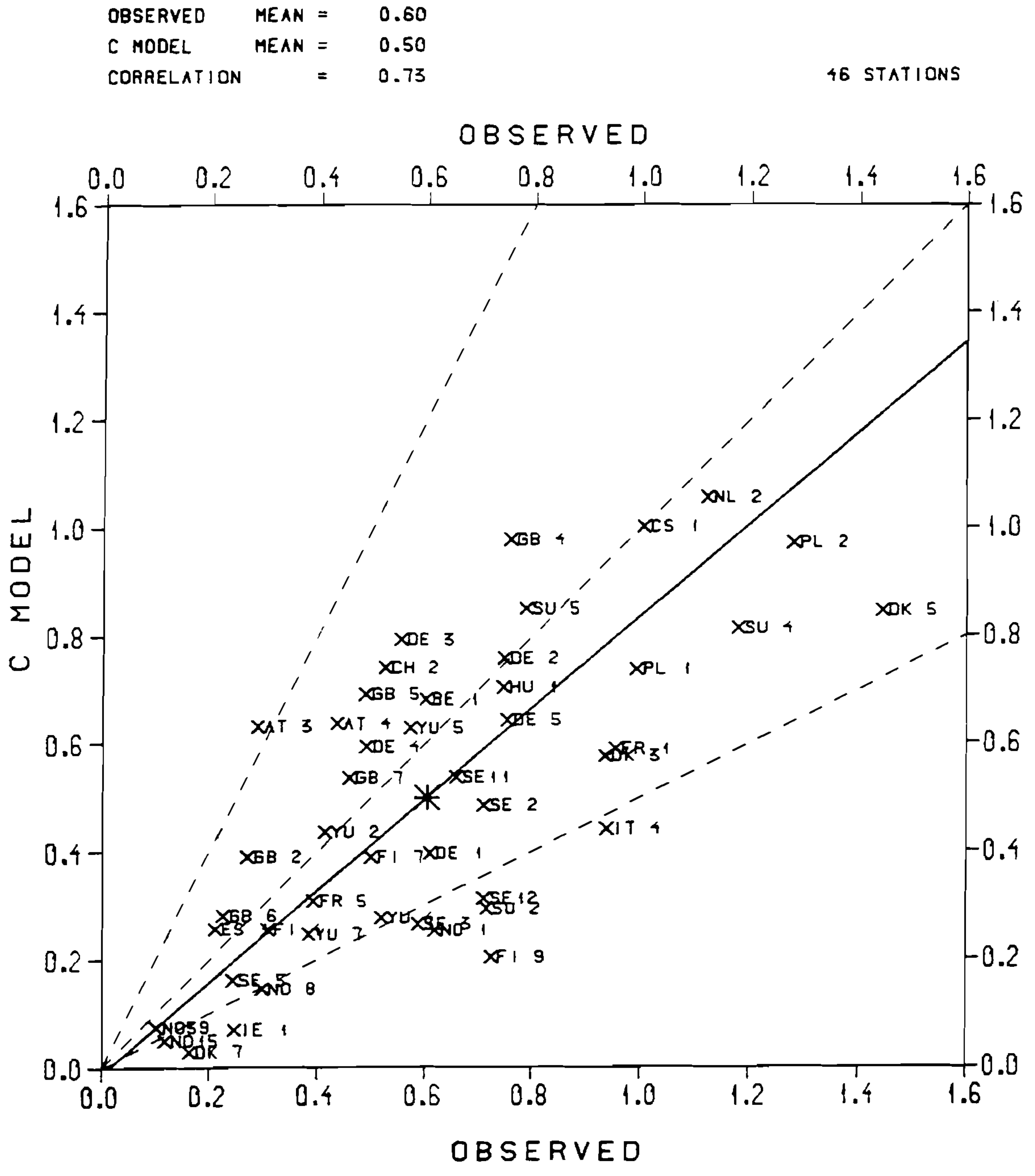



Figure 4. Modelled versus observed concentrations of ammonium in precipitation for stations with data coverage more than 25 %, unit mg(N)l. Data for the sites SU 6, SU 7, SE 8, and AT 2 are excluded.
The dashed lines show perfect agreement and disagreement with a factor of 2. The full line represents optimal linear regression.
See Iversen *et al.* (1991) for station codes.

two diagrams: the modelled and the observed concentrations of ammonium in precipitation as a function of the emission of ammonia within the grid-square in which the measurement site is situated. The modelled concentrations show a clear linear trend with increasing local emissions and so do the observed, except that the same 4 measurement sites as mentioned above report concentrations that are much larger than expected from the distributions of points in the diagram made up by all the remaining sites. Thus, either the measurements reported are of very bad quality, or (more probable) the emissions close to the sites are much too small. There are certain problems with the representativity of the ammonium measurements, since the sites often are situated close to emissions of ammonia. Nevertheless, we believe in this case that since the deviations are so enormous, the emissions used must also be wrong.

If the four stations in question are taken out from the statistics, the scatter plot looks like in figure 4. According to this, the quality of the ammomium calculations is similar to sulphate and nitrate (Iversen *et al.*, 1991).

## Conclusion

This paper is intended to illustrate that there are large potentials for improvements of calculated budgets of reduced nitrogen if the quality of ammonia emissions can be made better.

## Acknowledgement

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# EMISSIONS OF AMMONIA IN EUROPE AS INCORPORATED IN RAINS

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## Abstract

The ammonia emissions of the RAINS model are presented. Sources of ammonia considered are: livestock farming, fertilizers, industry, human population and other anthropogenic sources. Data on emission factors are based on recent insights in the Netherlands but are adapted to account for country specific elements such as: stable period, N-excretion, and the age and weight distribution. Ammonia emissions in 1980 in 26 European countries and Turkey are estimated at 7960 kilotons; 10 per cent higher than Buijsman et al. (1987) estimated. Ammonia emissions in 1987 are 8400 kilotons; 10 per cent lower than Asman (1990) suggested and in line with EMEP. Country and source specific estimates, however, are more uncertain: differences of 5 to 40 per cent are possible between the various international estimates. Based on national agricultural forecasts and trend analysis, future emissions of NH3 are expected to rise to 8620 kilotons in 2000.

# 1 Introduction

Nitrogen deposition from ammonia emissions is an important factor in regional acidification and eutrophication in Europe. Strategies to reduce nitrogen emission in Europe must include efforts to reduce ammonia emissions. During the past several years IIASA (The International Institute for Applied Systems Analysis) has been expanding the Regional Acidification INformation and Simulation (RAINS) model to include nitrogen compounds. During the past year attention was turned to ammonia including a detailed assessment of its sources, and the cost of controlling its emissions. This paper represents the effort of IIASA in quantifying past and future European emissions of ammonia.

This paper describes the design for the $NH_3$ emission module as incorporated in RAINS. In addition, the data on emission coefficients are presented and elucidated and some preliminary results are shown.

The remainder of the paper is organized as follows. Section 2 describes the overall set up and algorithm. Section 3 presents the emission coefficients for livestock farming and Section 4 the coefficients of nitrogen fertilizer. Industrial emission coefficients and the emissions of human population and other sources are explained in Section 5. Section 6 compares RAINS estimates for 1980 and 1987 with other national and international estimates. Section 7 projects the development of ammonia emissions to the year 2000.

# 2 The emission module

The emission module distinguishes the following sources of ammonia emissions:

1. Livestock farming:
   - dairy cows,
   - other cattle (including buffaloes)
   - pigs,
   - laying hens,
   - broilers (all other poultry, including turkeys and ducks),
   - sheep (including goats)
   - horses.
2. Nitrogen fertilizer consumption,
3. Industry (fertilizer and ammonia production plants),
4. Other anthropogenic sources (i.e. human respiration).

Other anthropogenic sources, of minor importance, include: human respiration, cats and dogs, sewage sludge, wild animals, traffic and coal combustion. Natural soils are an additional source.

Generally, $NH_3$ emissions are calculated as a product of the emission coefficients and the level of activity (livestock population, fertilizer consumption and production, human population). The following description uses the indices i and l, to describe the nature of the parameters:

i the type of animal
l the country

Ammonia from livestock farming is released during three basic processes:
- in the stable and during storage of manure,
- during the application of manure,
- in the meadow or grazing period.

These processes are explicitly distinguished in the model since this enables the possibility to calculate the potential of emissions that can be reduced through abatement measures such as: direct application of manure into the soil, cleaning of stable air and covering of manure storage facilities. The (unabated) ammonia emissions from livestock farming ($NH_3L_{i,l}$) are therefore calculated using the following equation:

$$NH_3L_{i,l} = (nh3s_{i,l} + nh3a_{i,l} + nh3m_{i,l}) * QL_{i,l} \tag{1}$$

In which:

| | |
|---|---|
| $nh3s_{i,l}$ | emission coefficient of stable |
| $nh3a_{i,l}$ | emission coefficient of application |
| $nh3m_{i,l}$ | emission coefficient meadow |
| $QL_{i,l}$ | animal population |

This equation is used for each of the seven animal types.

Ammonia emissions resulting from the consumption of nitrogen fertilizer ($NH_3F_l$) depend on the amount of fertilizer used and the N-loss per fertilizer:

$$NH_3F_l = nf_l * 17/14 * QF_l \tag{2}$$

In which:

| | |
|---|---|
| $nf_l$ | the n-loss per fertilizer |
| $QF_l$ | the fertilizer consumption |

Since the n-loss is expressed as per cent of the total nitrogen in the fertilizer the factor 17/14 is used to convert the losses expressed in nitrogen into ammonia. Note that fertilizer use and losses are country specific.

Industrial ammonia emissions are mainly related to the production of fertilizer and ammonia. The total industrial ammonia emissions ($NH_3P_l$) are therefore the product of the production of nitrogen fertilizer in each country and the emission coefficient:

$$NH_3P_l = nh3p * QP_l \tag{3}$$

With:

| | |
|---|---|
| $nh3p$ | the emission coefficient for industry |
| $QP_l$ | N-fertilizer production |

Other sources of ammonia are: human respiration, cats and dogs, sewage sludge, wild animals, traffic, natural soils and coal combustion. Of these sources human respiration is explicitly incorporated. Remaining anthropogenic sources are included insofar as national data are available. However, emissions of natural soils are ignored in view of the large uncertainties in their order of magnitude (Buijsman et al, 1987). Buijsman et al. (1987) estimate total Europe wide ammonia emissions from natural soils at 750 kilotons of ammonia per year. This would be 10 per cent of the total ammonia emission in Europe. Other sources ($NH_3O_l$) are incorporated in the following manner:

$$NH_3O_l = nh3h * QH_l + Cnh3_l \qquad (4)$$

With:

| | |
|---|---|
| nh3h | emission coefficient human population |
| $QH_l$ | size human population |
| $Cnh3_l$ | constant for other anthropogenic emissions |

# 3 Emission coefficients for livestock animals

## 3.1 Introduction

In the past, several overviews have been made that describe ammonia emissions in Europe (Bonis, 1980; Buijsman et al., 1987; Asman, 1990; Iversen et al., 1990). A problem of the estimate made by Buijsman et al. (1987) is that they probably underestimate the emissions since for most countries their results go back to research in the Netherlands on the nitrogen content of the excretion as carried out in 1978 (Sluijsmans et al., 1979). Only for Denmark and the United Kingdom country specific data were used. In view of more recent information (De Winkel, 1988; Möller and Schieferdecker, 1989) on the nitrogen content of the excretion, the estimate made by Buijsman et al (1987) needs revision. Estimates by EMEP (Iversen et al, 1990) are the ones by Buijsman et al times a factor 1.2. A weak spot of the emission calculation by Asman (1990) is that emission factors typically for one country, the Netherlands, although based on recent insights, are used to calculate emissions for every country. In view of large differences in agricultural practices, this appears to be inappropriate.

In contrast to the detailed information available about emission factors for $NH_3$ in the Netherlands, data on ammonia emission factors based on country specific data on nitrogen excretion and volatilization of ammonia, is available only for a few other European countries:

- Finland (Niskanen et al., 1990),
- German Democratic Republic (Möller and Schieferdecker, 1989),

- the Netherlands (Erisman, 1989),
- the United Kingdom (ApSimon et al., 1989).

For other countries, estimates are based on general rather than country specific emission factors:

- Czech and Slovak Federal Republic (Zavodsky and Mitosinkova, 1984),
- Denmark (Schröder, 1985; Laursen, 1989),
- Federal Republic of Germany (Isermann, 1990),
- Hungary (Bonis, 1981),
- Norway (Bockmann et al., 1990).

Or they are (partly) based on the same, rather outdated, estimates of the nitrogen content that were used by Buijsman et al. (1987). Examples are: the Federal Republic of Germany (Fabry et al., 1990) and Switzerland (Stadelmann, 1988). Table 1 presents an overview of national estimates.

**Table 1.** National $NH_3$ estimates

| Estimate Country | $NH_3$ emission (Kton $NH_3$) | | Year | Reference |
|---|---|---|---|---|
| CSFR | 128-222 | 3) | 1981 | Zavodsky et al. (1984) |
| Denmark | 106-138 | 2) | 78/82 | Sommer et al. (1984) |
| | 196 | 1) | 85/86 | Schröder (1985) |
| | 155 | 1) | 1980 | Laursen (1989) |
| Finland | 52 | | 84/86 | Niskanen et al. (1990) |
| FRG | 348-360 | | 1988 | Fabry et al. (1990) |
| | 641 | 1) | 1986 | Isermann (1990) |
| GDR | 345-355 | 3) | 80/85 | Möller et al.(1989) |
| Hungary | 90-157 | | 1976 | Bonis (1981) |
| | 150 | | 80/87 | Fekete (1990) |
| Netherlands | 258 | | 1987 | Erisman (1989) |
| | 154 | | 1092 | Buijsman et al. (1984) |
| Norway | 57 | 1) | 80/89 | Bockmann et al. (1990) |
| Switzerland | 64 | 3) | 1987 | Stadelman (1988) |
| UK | 451 | 1) | 83/84 | ApSimon et al. (1989) |

1) Only agricultural sources.
2) Livestock manure only.
3) Includes emissions from natural sources

Therefore this study's starting point is more recent information on emission coefficients in the Netherlands, summarized in Table 2. These emission coefficients are based on the work of a working group of scientists, established in the Netherlands to evaluate the present knowledge and to obtain more consistent and improved estimates on emission factors for $NH_3$ from livestock farming (De Winkel, 1988; Van der Hoek, 1989, Hannessen, 1991). For the most relevant animal categories the working group has derived average annual emission factors per animal. Emission factors for stall and storage, manure application and the meadow period were based on the application of nitrogen mass balances.

Their principle approach can be summarized in four equations:

N excretion = N feed - N retention (5)
N stable = N excretion * volatilization_s (6)
N application = (N excretion-N stable)*volatilization_a (7)
N meadow = N excretion * volatilization_m (8)

**Table 2.** Emission coefficients for livestock animals in the Netherlands (per animal in kg $NH_3$/annum)

| Subcategory | Emission coefficient | | | |
|---|---|---|---|---|
| | Stable/ Storage | Application | Meadow period | Total |
| Dairy and calf cows | 8.79 | 14.40 | 12.34 | 35.53 |
| Other cattle | 3.61 | 6.14 | 2.74 | 12.49 |
| Pigs | 2.27 | 2.85 | 0.00 | 5.12 |
| Laying hens | 0.14 | 0.18 | 0.00 | 0.32 |
| Broilers | 0.07 | 0.11 | 0.00 | 0.18 |
| Sheep | 0.39 | 0.71 | 0.96 | 2.06 |
| Horses | 5.00 | 4.00 | 3.50 | 12.50 |

Data based on de Winkel (1988), Van der Hoek (1989) and Hannessen (1991). Horses based on Asman (1990). Detailed data have been aggregated using national livestock data for the Netherlands in 1988 (see Klaassen, 1991). Other cattle are total cattle minus dairy cows. Sheep include goats. Broilers include other poultry such as turkeys and ducks.

The results of nutritional research were used to compute the nitrogen content of the feed per animal (N feed) as well as the retention of nitrogen (N retention) in various animal products such as meat and milk. As a result the nitrogen remaining in the excretion (N excretion) could be calculated. The volatilization of ammonia in the stall and during storage of manure (volatilization_s), or in other words the loss of nitrogen, was determined by looking at the difference between the N/P ratio in excrements and in stored manure. P is regarded as a conservative component, whereas N may evaporate as $NH_3$. The volatilization of ammonia can then be computed from changes in the N/P ratio during storage. Where possible the average emissions factors per animal were differentiated for different housing systems using recent emission measurements (Hannessen, 1991; Van der Hoek, 1989). The volatilization coefficient of ammonia (volatilization_a) during application (N application) and during the grazing or meadow period (volatilization_m) was based on experiments described in the literature and additional experiments carried out by various research groups. The more detailed results for the Netherlands are included in Klaassen (1991). These results have been summarized per animal category (dairy cows, other cattle, pigs, laying hens, broilers, sheep and horses) using data for 1988 on the composition of the animal population in the Netherlands (Central Bureau of Statistics, 1989) and are presented in Table 2.

The remaining sections will explain how the emission coefficients of Table 2 were modified for several livestock categories to arrive at country specific emission coefficients.

### 3.2 Emission coefficients for dairy cows

Regarding dairy cows, the major elements influencing emission factors are:

- feed composition, amount and its nitrogen content,
- retention of nitrogen in milk and meat,
- volatilization of ammonia in the stable,
- volatilization of ammonia during application,
- volatilization of ammonia in the meadow period.

Van Dijk and Hoogervorst (1984) indicate that the share of grass in the total feed consumption differs among countries. The nitrogen content of the grass will differ since the amounts of nitrogen fertilizer applied on a pasture varies between countries (CEC, 1989). In addition, international statistics show that large differences in the annual milk production per cow exist. This suggests that the retention of nitrogen in milk might differ considerably amongst countries. As a result, the nitrogen content of the excretion is likely to vary between countries. The volume of ammonia emitted in the stall and during storage depends on the volatilization coefficient and the stall period. The number of days spent in the stall varies (Asman, 1990). Method of storing manure, stable type and type of manure (liquid/solid) are other factors affecting ammonia volatilization in the stall. Emission during application depends on factors such as the type of manure (liquid/solid), soil type, temperature, wind speed and method of applying manure. Although differences amongst countries do exist, data and lack of theoretical insight do not allow to quantify the impact of these other factors on the volatilization. In summary, on the one hand it does not seem appropriate to use the emission coefficients from the Netherlands for other countries. On the other hand, for only a few of the potentially large number of factors affecting emissions, data and sound theory is available.

To compute country specific emission coefficients for dairy cows in RAINS we decided to take into account differences in the level of nitrogen fertilizer application as well as differences in meadow and stall period. For both these elements data was available. Moreover, the differences in meadow periods were thought to be relevant because they influenced the volume of $NH_3$ emission released during stall, application and meadow period. Consequently, this affects the potential of emissions to be abated and the associated abatement costs.

The method used is the following. A recent study (Baltussen et al, 1990) indicates that there is a relationship between the nitrogen excretion of dairy cows and the nitrogen level of grassland. The nitrogen level of grassland is, to a large extent, determined by the amount of fertilizer applied. Based on data for the Netherlands the following relation has been estimated:

$$\text{N-excretion} = 126.22252 + 0.1932 * \text{N-fertilizer} \qquad (9)$$

In which N-excretion is the nitrogen excretion per animal and N-fertilizer is the fertilizer use per hectare. This relation has been used to estimate the N-excretion for other countries in Europe. The relation between the N-excretion per dairy cow in the Netherlands and the other countries is then used to correct the Netherlands emission factors. Details on the method and the data used are provided in Klaassen (1991). In addition, the amount of N-excretion produced in the meadow period and the stall period has been corrected using information on the meadow periods in several countries in Europe (Asman, 1990). This is based on the following equations:

$$\text{N-excretion stall} = \text{N\_excretion* st period/st period NL} \qquad (10)$$
$$\text{N-excretion meadow} = \text{N\_excretion *meadowperiod/meadow NL} \qquad (11)$$

where st period is the stall period in the specific country (in days) and st period NL is the stall period in the Netherlands (in days/year). Using equations (9) to (11) and data on the volatilization factors based on De Winkel (1988), country specific emission coefficients for stall, application and meadow have been calculated for dairy cows. Details are provided in Klaassen (1991) and summarized in Table 3.

Total emission coefficients vary between 24.0 kg $NH_3$/animal per year and 35.5 kg $NH_3$/animal per year, mainly due to the differences in fertilizer level. The coefficients for stall, application and meadow differ roughly by a factor two (Klaassen, 1991). These differences considerably influence the

**Table 3.** Emission coefficients RAINS (kg $NH_3$ per animal per year)

| COUNTRY | DAIRY COWS | OTHER CATTLE | PIGS | SHEEP AND GOATS |
|---|---|---|---|---|
| Albania | 27.3 | 12.5 | 5.1 | 2.5 |
| Austria | 27.9 | 12.5 | 5.2 | 1.9 |
| Belgium | 26.4 | 14.1 | 5.3 | 1.7 |
| Bulgaria | 27.3 | 12.5 | 5.1 | 2.0 |
| CSFR | 32.0 | 12.5 | 5.1 | 2.0 |
| Denmark | 31.3 | 12.6 | 4.6 | 1.9 |
| Finland | 33.2 | 11.4 | 5.1 | 3.0 |
| France | 24.6 | 14.2 | 5.0 | 2.0 |
| FRG | 32.6 | 12.4 | 5.0 | 2.4 |
| GDR | 30.3 | 12.5 | 5.1 | 1.9 |
| Greece | 25.6 | 11.9 | 4.8 | 2.2 |
| Hungary | 24.6 | 12.5 | 5.1 | 1.9 |
| Ireland | 24.9 | 13.9 | 5.1 | 1.9 |
| Italy | 26.0 | 13.8 | 4.9 | 2.0 |
| Luxembourg | 29.9 | 14.5 | 5.0 | 2.5 |
| Netherlands | 35.5 | 12.5 | 5.1 | 2.0 |
| Norway | 33.7 | 12.5 | 5.1 | 2.9 |
| Poland | 27.8 | 12.5 | 5.1 | 1.9 |
| Portugal | 26.3 | 12.5 | 5.1 | 2.1 |
| Romania | 27.4 | 12.5 | 5.1 | 2.0 |
| Spain | 25.4 | 12.3 | 5.0 | 2.1 |
| Sweden | 30.2 | 12.5 | 5.1 | 2.9 |
| Switzerland | 32.9 | 13.3 | 4.2 | 2.1 |
| Turkey | 24.0 | 12.5 | 5.1 | 2.2 |
| UK | 26.5 | 14.8 | 5.1 | 2.7 |
| USSR | 24.3 | 12.5 | 5.1 | 2.0 |
| Yugoslavia | 25.2 | 12.5 | 5.1 | 1.9 |

potential for abatement in the various countries. It is recalled, however, that we were not able to take into account all the relevant factors. For example, ammonia emission in Southern European countries might be underestimated since manure is usually stored outside as solid manure. Although the nitrogen content of the excretion might be less, this method of storing manure is likely to increase the ammonia emission again.

### 3.3 Other cattle

For other cattle there also may be differences among countries regarding the nitrogen content of the feed, nitrogen retention in meat, and the volatilization during stall and storage and the application of manure. Due to a lack of data, we were only able to take into the weight and age distribution within the category other cattle to calculate country specific coefficients on the basis of the detailed emission coefficients for the Netherlands (see Klaassen, 1991). The results are presented in Table 3.

### 3.4 Pigs, laying hens, broilers and horses

For these animals nitrogen content of the excretion may differ among countries due to differences in nitrogen content of the feed and nitrogen retention. In addition, the ammonia emitted from stall and manure might vary due to differences in stall type (mechanical/natural ventilation for example; Asman, 1990) and manure storage system. Differences in stable and manure handling systems are likely to cause differences in ammonia emissions from the stall, especially for laying hens. However, the stall period will not differ too much since pigs and poultry are usually inside the whole year (Asman, 1990). The losses of ammonia during application may also differ in view of differences in the usual factors affecting ammonia volatilization during application. Due to lack of data we used the data from the Netherlands (Table 2) for each country for laying hens, broilers and horses. For pigs we took into account the weight- and age distribution to arrive at country specific emission coefficients (see Table 3), starting from the detailed emission coefficients reported by the Netherlands.

### 3.5 Sheep

For sheep differences in meadow period and the composition of the sheep flock over sheep and goats have been taken into account to arrive at country specific factors. The Netherlands emission factors (Table 2) were modified as follows. Ammonia losses (as kg $NH_3$ per animal per year) in the stall (N_stall), during application (N_application), and in the meadow (N_meadow) are calculated as follows.

$$N\_stall = N\_excretion * \text{stall period} * 0.12 * 17/14 \quad (12)$$
$$N\_application = N\_excretion * (1-0.12) * \text{stall period} * 0.25 * 17/14 \quad (13)$$
$$N\_meadow = N\_excretion * \text{meadow period} * 0.12 * 17/14 \quad (14)$$

where N_excretion is the nitrogen content in the excretion per animal per year in the Netherlands. The stall period is expressed as part of the year. 0.12 is the part of the nitrogen in the excretion that is released as ammonia in the stall (Equation 13) as well as in the meadow (Equation 14). During application, a share of 0.25 is released of the nitrogen in excretion, taking into account the loss that already occurred in the stable (1 - 0.12). The N_excretion used is 9.8535 kg N/animal per year for sheep (including lambs) and 15.567 kg for goats. All data is based on Van der Hoek (1989). Using the above equations, data on the meadow period (derived from Asman, 1990: see Klaassen, 1991) and the number of sheep and goats in each country, the average emission coefficients for the category sheep in each country have been calculated (Table 3).

Table 3 shows that emission coefficients vary between 1.3 kg $NH_3$ per animal per year (United Kingdom) and 3.0 kg $NH_3$ per animal per year (Finland) as a result of differences in meadow period and the ratio between sheep and goats. One should realize, however, that the data on nitrogen excretion and volatilization factors were still based on Dutch data.

### 3.6 A comparison with other emission coefficients

Table 4 compares the results of the emission coefficients used in RAINS with other estimates. The RAINS emission coefficients for dairy cows are generally below the ones of Möller and Schieferdecker (1989) and more recent Dutch ones (Table 2). Neither Buijsman et al. (1987) nor Asman (1990) explicitly distinguish between dairy cows and other cattle. The emission coefficients for other cattle are somewhat lower than Möller and Schieferdecker (1989) but in line with the Netherlands. The average, country specific emission coefficients in RAINS for cattle (dairy cows and other cattle) are difficult to compare with the other estimates since they depend on the share of dairy cows in the total cattle stock. For pigs RAINS estimates are comparable with the ones reported in the literature. The emission coefficients for poultry (laying hens and broilers) are difficult to comparable since RAINS distinguishes laying hens and other poultry. Estimates in RAINS for sheep are below the ones provided by Buijsman et al. (1987) and Möller and Schieferdecker (1989) but comparable with Asman (1990) and the Netherlands. Emissions coefficients for horses are in between both other estimates. Major differences and uncertainties appear to

Table 4. Comparison of emission coefficients (in kg $NH_3$ per animal per year)

| Livestock Category | RAINS (1991) | Buijsman et al. (1987) | Möller et al. (1989) | Netherlands (1988/1991) | Asman (1990) |
|---|---|---|---|---|---|
| Dairy cows | 24.0-35.5 | 18.4 | 42.5 | 35.5 | 25.1 |
| Other cattle | 11.4-14.8 | 18.4 | 18.7 | 12.5 | 25.1 |
| Pigs | 4.2-5.3 | 2.8 | 6.3 | 5.1 | 4.8 |
| Laying hens | 0.32 | 0.26 | 0.27 | 0.32 | 0.32 |
| Broilers | 0.18 | 0.26 | 0.27 | 0.18 | 0.32 |
| Sheep | 1.7-3.0 | 3.1 | 3.6 | 2.1 | 1.9 |
| Horses | 12.5 | 9.4 | 18.2 | 12.5 | 12.5 |

exist especially for cattle. For the other animals RAINS estimates are comparable with the (wide) ranges observed in the literature.

# 4 Fertilizer use

Ammonia emissions released when nitrogen fertilizer is applied depend on elements such as: the type of fertilizer, soil PH and cation exchange capacity, drying conditions and irrigation. In this study we use average emission coefficients for each type of fertilizer used by Buijsman et al. (1987) and Asman (1990). Using information on the type of fertilizer for each country (Klaassen, 1991) average N-losses as ammonia from fertilizer have been determined (Table 5).

**Table 5.** Average N_losses of fertilizers

| Country | (% Loss of N_Content) |
|---|---|
| Albania | 6.0 |
| Austria | 1.7 |
| Belgium | 2.0 |
| Bulgaria | 5.0 |
| CSFR | 5.0 |
| Denmark | 1.7 |
| Finland | 1.3 |
| France | 2.7 |
| FRG | 3.4 |
| GDR | 3.3 |
| Greece | 5.8 |
| Hungary | 7.0 |
| Ireland | 3.8 |
| Italy | 5.8 |
| Luxembourg | 2.0 |
| Netherlands | 1.9 |
| Norway | 1.1 |
| Poland | 9.8 |
| Portugal | 4.2 |
| Romania | 5.0 |
| Spain | 4.6 |
| Sweden | 2.2 |
| Switzerland | 4.1 |
| Turkey | 6.5 |
| UK | 5.4 |
| USSR | 5.0 |
| Yugoslavia | 5.0 |

# 5 Industry and other anthropogenic sources

Ammonia production and fertilizer plants are the main sources of industrial ammonia emissions. Following Buijsman et al. (1987) we assumed the total production of ammonia plants in each country to be proportional to the fertilizer production. Emission coefficients for ammonia plants are taken as 0.8 kg $NH_3$ /ton fertilizer produced (Ministry of Housing, Physical Planning and Environment, 1983). According to the same source emission factors for fertilizer plants may vary between 0.01 kg $NH_3$ per ton and 12.5 kg $NH_3$ per ton produced. As Buijsman et al. (1987) we assumed an average coefficient of 5 kg $NH_3$/ ton fertilizer produced. As a result, the total emission coefficient used for industrial ammonia sources is 5.8 kg $NH_3$/ton fertilizer produced.

For human population we use an emission coefficient of 0.3 kg $NH_3$/head (Buijsman, 1984; Erisman, 1989). For the other anthropogenic sources, different national sources have been used to estimate these (Stadelman, 1988; Möller and Schieferdecker, 1989; Erisman, 1989; Niskanen et al. 1990). These other sources are generally negligible (Klaassen, 1991). In all cases natural sources were excluded.

# 6 A comparison of past estimates

The ammonia emissions for 1980 and 1987 were calculated using the emission coefficients of Table 2 and 3 and data on livestock population, fertilizer consumption and production, as well as human population. Data on livestock population and fertilizer use is from FAO (1990a, 1990b) and national livestock statistics for Belgium, Luxembourg and the USSR (Institut Economique Agricole, 1989; Statistical Board of the USSR, 1989). Human population data are based on United Nations (1989a, 1989b) and estimates of IIASA's Population Program for the EMEP (European Monitoring and Evaluation Program) part of the USSR. The data on the USSR refer only to that part of the USSR that is within the grid used by EMEP. That includes the USSR republics Ukraine, White Russia, Georgia, Azerbajdzjan, Lithunia, Moldavia, Latvia, Armenia, Estonia and that part of the RSFSR (Russia) which is within the EMEP grid. For fertilizer use data of the British Sulphur Corporation (1987) were used.

Table 6 compares the emissions of various authors on a country-by-country basis. Since the various authors make different assumptions on that part of the USSR which is included in their calculations estimates for the USSR show wide differences. We will therefore compare the total estimates for Europe excluding the USSR. Table 6 shows that our estimate for 1980 is 10 per cent higher than the ones from Buijsman et al.(1987). The estimate for 1987 is

**Table 6.** $NH_3$ emission estimates per country (kton $NH_3$)

| Estimate | Buijsman et al. | IIASA | IIASA | Asman | EMEP |
|---|---|---|---|---|---|
| Country | (1987) 1980/83 | 1980 | 1987 | (1990) 1987 | (1990) 1988 |
| Albania | 21 | 25 | 27 | 32 | 24 |
| Austria | 72 | 79 | 79 | 107 | 85 |
| Belgium | 82 | 102 | 105 | 123 | 94 |
| Bulgaria | 126 | 122 | 120 | 123 | 147 |
| CSFR | 170 | 200 | 197 | 219 | 200 |
| Denmark | 111 | 116 | 103 | 144 | 129 |
| Finland | 44 | 56 | 49 | 61 | 43 |
| France | 709 | 679 | 650 | 974 | 841 |
| FRG | 371 | 529 | 533 | 718 | 380 |
| GDR | 207 | 228 | 239 | 274 | 242 |
| Greece | 95 | 88 | 100 | 111 | 112 |
| Hungary | 130 | 156 | 155 | 179 | 151 |
| Ireland | 117 | 128 | 128 | 188 | 139 |
| Italy | 361 | 359 | 366 | 435 | 426 |
| Luxembourg | 5 | 5 | 5 | 7 | 6 |
| Netherlands | 150 | 224 | 239 | 276 | 218 |
| Norway | 36 | 37 | 47 | 38 | 41 |
| Poland | 405 | 570 | 528 | 561 | 478 |
| Portugal | 47 | 66 | 65 | 76 | 55 |
| Romania | 301 | 297 | 340 | 387 | 350 |
| Spain | 232 | 251 | 317 | 365 | 273 |
| Sweden | 52 | 66 | 59 | 74 | 62 |
| Switzerland | 53 | 64 | 60 | 68 | 61 |
| Turkey | 683 | 532 | 476 | 573 | 699 |
| UK | 405 | 482 | 492 | 548 | 478 |
| USSR | 1256 | 2288 | 2446 | 1543 | 3182 |
| Yugoslavia | 198 | 214 | 217 | 235 | 235 |
| EUROPE | 6434 | 7961 | 8143 | 8439 | 9129 |
| Europe minus USSR | 5178 | 5676 | 5696 | 6903 | 5969 |

Note: Due to rounding total might differ from the sum.

comparable with the EMEP estimate (Iversen et al., 1990) but 20 per cent lower than the one by Asman (1990). The Table indicates that IIASA estimates for 1980 are considerably higher for some countries (the Netherlands, FRG, Poland) than the Buijsman et al. (1987) computation. For several countries (Turkey e.g.) the IIASA estimate is lower. Generally IIASA estimates for 1987 are (slightly) lower than the ones by Asman (1990), (for example, Spain, France and the FRG) but are sometimes higher (for example Norway). IIASA estimates are generally in line with EMEP but for some countries differ (Finland, Turkey to name a few). Estimates for countries show considerably larger differences (up to 40 per cent, for example FRG) than the ones for Europe as a whole.

That there exists wide uncertainty in country estimates can also be concluded when comparing Table 1 with Table 6. IIASA estimates differ with 10 to 20 per cent with national estimates. In view of the lack of fundamental data (nitrogen content feed, volatilization factors) for most countries the calculation of country specific ammonia emission coefficients using nitrogen mass balance remains difficult.

Table 7 compares the estimates by source, excluding the USSR. The Table shows that our estimates are higher than Buijsman et al. (1987) for 1980 since our estimates for pigs are higher and we include other sources as well. By contrast, our estimates for sheep are lower. Estimates for cattle, poultry and industry have the same order of magnitude. Our overall estimate for 1987 is roughly 10 per cent lower than the one by Asman (1990). The main reason is that our estimates for cattle and for fertilizer use are lower. This being so because we use country specific emission factors for dairy cows which are generally lower than Asman's. For fertilizer consumption our estimates are lower chiefly because we take into account the difference in nitrogen loss between calcium ammonium nitrate and ammonium nitrate (Buijsman et al., 1987). Estimates for industry are higher since we include both fertilizer and

**Table 7.** $NH_3$ estimates according to source (kton $NH_3$)

| Estimate | Buijsman et al. (1987) | IIASA | IIASA | Asman (1990) |
|---|---|---|---|---|
| Source | early 80 | 1980 | 1987 | 1987 |
| Cattle | 2749 | 2813 | 2624 | 3560 |
| Pigs | 422 | 868 | 949 | 897 |
| Poultry | 325 | 339 | 344 | 436 |
| Sheep | 640 | 441 | 468 | 395 |
| Horses | 58 | 72 | 63 | 64 |
| Fertilizer | 881 | 867 | 959 | 1538 |
| Industry | 102 | 104 | 101 | 13 |
| Other | 0 | 172 | 188 | 0 |
| Total | 5178 | 5676 | 5696 | 6903 |

Note: excluding the USSR.
Due to rounding, total might differ from sum.

ammonia production as sources. Figures for pigs, poultry, sheep have the same order of magnitude but somewhat different for pigs and poultry since we use recent data on the emission coefficients (Van der Hoek, 1989; Hannessen, 1991).

In sum, estimates for total European NH3 emissions are 10 - 20 per cent higher or lower than the IIASA estimates. Estimates for specific countries, and for specific source categories, however, can show considerably greater (up to 40%) or smaller divergences (smaller than 5 per cent).

# 7 Ammonia emissions in 2000

The forecasts on livestock population and fertilizer consumption necessary to project future ammonia emissions are, as far as possible, based on national forecasts from various agricultural research institutes or universities (Klaassen, 1991). Country specific estimates were not available in international studies (Alexandratos, 1990) or were outdated (Politiek and Bakker, 1982). If no country specific data were available, trends as observed in the period 1979-1988 were extrapolated. Where necessary trends were adjusted to bring the forecasts in line with regional forecasts of the OECD (Boonekamp, 1990) and the EC (Schäfer, 1990). Fertilizer production in 2000 was based on trend extrapolation. If necessary, trends were adapted to reflect national estimates on future consumption patterns. Forecasts on human population were based on the UN medium scenario (UN, 1989b). The resulting ammonia emissions for the year 2000 are shown in Table 8. They are based on the assumption that emission coefficients for ammonia remain constant over time. This may imply an underestimation of the emissions since yields/animal and consequently nitrogen excretion and ammonia emissions per animal might increase over time. According to Table 8, total ammonia emissions in Europe will increase from 8000 kilotons in 1980 to more than 8600 kilotons in 2000. Over the 1987 level this implies an increase of 8 per cent.

The regional trend in ammonia emissions is as follows. Emissions are expected to decline or stabilize in the EC-North (Belgium, Denmark, France, FRGermany, Ireland, Luxembourg, Netherlands and the United Kingdom), Scandinavia (Finland, Norway, Sweden) and Alpine countries (Austria, Switzerland). Mediterranean countries show a diffuse picture (Albania, Yugoslavia and Turkey). An increase is generally expected in EC-South (Greece, Italy, Portugal, Spain) and in Eastern Europe (Bulgaria, Czechoslovakia, Germany DR, Hungary, Poland, Rumania and the USSR).

Table 9 and Figure 1 show that the slight increase in ammonia up to the year 2000 results from two opposing trends. Emissions from cattle, especially dairy cows, will decrease considerably. By contrast, emissions from other livestock animals, pigs, poultry and sheep and especially from fertilizer use, will increase if no abatement measures will be taken.

**Table 8.** Future $NH_3$ emission estimates per country (kton $NH_3$)

| Year<br>Country | 1980 | 1987 | 2000 | Index<br>2000/1980 |
|---|---|---|---|---|
| Albania | 25 | 27 | 33 | 134 |
| Austria | 79 | 79 | 80 | 101 |
| Belgium | 102 | 105 | 90 | 89 |
| Bulgaria | 122 | 120 | 141 | 115 |
| CSFR | 200 | 197 | 191 | 95 |
| Denmark | 116 | 103 | 81 | 70 |
| Finland | 56 | 49 | 39 | 70 |
| France | 679 | 650 | 637 | 94 |
| FRG | 529 | 533 | 541 | 102 |
| GDR | 228 | 239 | 176 | 77 |
| Greece | 88 | 100 | 125 | 143 |
| Hungary | 156 | 155 | 161 | 103 |
| Ireland | 128 | 128 | 156 | 122 |
| Italy | 359 | 366 | 371 | 103 |
| Luxembourg | 5 | 5 | 5 | 100 |
| Netherlands | 224 | 239 | 209 | 93 |
| Norway | 37 | 47 | 31 | 83 |
| Poland | 570 | 528 | 476 | 84 |
| Portugal | 66 | 65 | 62 | 94 |
| Romania | 297 | 340 | 422 | 142 |
| Spain | 251 | 317 | 409 | 163 |
| Sweden | 66 | 59 | 59 | 89 |
| Switzerland | 64 | 60 | 52 | 81 |
| Turkey | 532 | 476 | 414 | 78 |
| UK | 482 | 492 | 509 | 105 |
| USSR | 2288 | 2446 | 2935 | 128 |
| Yugoslavia | 214 | 217 | 218 | 202 |
| EUROPE | 7961 | 8143 | 8620 | 108 |

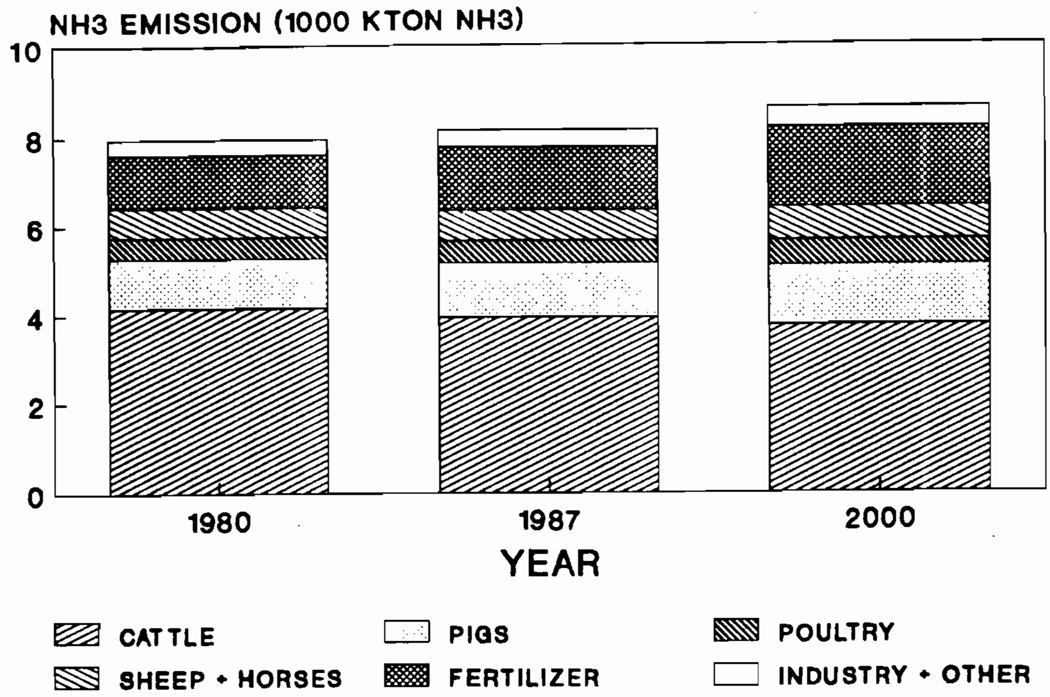
Note: Due to rounding, totals might differ from the sum.

**Table 9.** Future $NH_3$ emissions by source (kton $NH_3$)

| Source | 1980 | 1987 | 2000 |
|---|---|---|---|
| Dairy cows | 2373 | 2228 | 1928 |
| Other cattle | 1783 | 1703 | 1825 |
| Pigs | 1125 | 1223 | 1338 |
| Laying hens | 270 | 272 | 270 |
| Other Poultry | 212 | 237 | 293 |
| Sheep | 545 | 571 | 629 |
| Horses | 127 | 120 | 110 |
| Fertilizer | 1170 | 1393 | 1781 |
| Industry | 140 | 156 | 202 |
| Other | 219 | 240 | 240 |
| Total | 7961 | 8143 | 8620 |

Note: Due to rounding, total might differ from sum.

**Figure 1.** $NH_3$ by source and year



There exists uncertainty in these estimates. Not only because differences exist in estimates of emissions coefficients. In addition, there is uncertainty in the forecasts. This is not only due to major uncertainties on the continuation of EC-policy (especially regarding dairy cows and potential new member states) but also due to structural changes in Eastern Europe. For the former GDR, for example, the national expert projection shows a drastic reduction in the number of livestock animals and fertilizer use. Using estimates of Nikolov (1990) for livestock population, based on long term trend extrapolation, would generally show that ammonia emissions in Eastern Europe would rise even more. Emissions in Mediterranean countries (especially Turkey) might even increase.

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# REGIONAL SCALE AMMONIA EMISSION INVENTORY FOR HUNGARY

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## Abstract

Ammonia emissions have been estimated for four source categories: animals, fertilizer, industry and soil exhalation. $NH_3$ emissions released in Hungary, using available county statistical data and emission factors reported by Dutch researchers, were determined for years 1980, 1985 and 1987. The total values for the country of about 150 kt $NH_3$ $a^{-1}$, show satisfactory agreement with emissions estimated by other authors. Emitted $NH_3$ quantities originated 89 per cent from agricultural (animals and fertilizer) sources. The ammonia emission inventories used as input for the regional model were produced with a resolution of 20km x 20km.

# 1 Ammonia sources and emission factors

Ammonia in the atmosphere is emitted mostly from agricultural sources near to the ground level. The important sources are mainly livestock wastes and fertilizer applications with a small proportion.

Buijsman et al. (1985) computed the ammonia emission in Europe by using livestock statistics and emission factors for different categories of domestic animals. Emission factors applied by us originate from Buijsman and are shown in Table 1. These are in satisfactory agreement with values of other authors (Bonis et al. 1980, ApSimon et al. 1987). Factors for domestic animals were extended to wild animals.

Emission originating from fertilizer application of $NH_3$ content depends on the type of fertilizer. Emission factors are between 2 and 15% of the quantity used. The average value is 10%. This agrees with the per cent given for fertilizer $NH_4NO_3$ by Buijsman et al. (1985).

The ammonia emission can be released from industrial processes too. Buijsman et al. (1985) took into account the ammonia factories where the applied emission factor is 0.8 kg $NH_3$ per t of produced ammonia.

Anthropogenic sources, such as automobile traffic and coal combustion, release ammonia in small proportions. Since this part approximately corresponds to the quantity absorbed by atmospheric oxidation of $NH_3$ gas, these emissions were neglected.

An important natural source is the ammonia exhalation of some soils due to the decomposition of nitrogenous organic matter in terrestrial ecosystems. According to Bonis et al. (1980) the average rate of this soil emission is equal to 10 $\mu g\ m^{-2}\ hr^{-1}$.

# 2 County and country-wide ammonia emissions for Hungary

Hungary's ammonia emissions were determined for the years 1980, 1985 and 1987. At first the county emissions were calculated. The sums of these values gave the total emissions for the country. Four source categories were discriminated:

Anthropogenic sources

(a) animals } agriculture
(b) fertilizer }
(c) industry

Natural sources

(d) soil exhalation

(a) Animals

Yearly $NH_3$ emissions were estimated for five different kinds of the domestic animals and for three types of wild animals from livestock statistics using the corresponding emission factors in Table 1. The county stock of game was determined from total numbers of wild animals for Hungary distributed according to the rate of the county forest territories. The country-wide totals are given in Table 1.

(b) Fertilizer

County statistical data regarding chemical fertilizer quantities of nitrogen content are known. According to estimates made by specialists about 73% of these fertilizers in Hungary had ammonium content, mainly as $NH_4NO_3$. As it was mentioned in the previous section the emission factor of 10% was used in this case.

Table 1 Ammonia emission factors for animals and ammonia emissions by Hungarian animals in 1980, 1985 and 1987

| Animal type | | Emission factors kg $NH_3$ $a^{-1}$ $animal^{-1}$ | Emissions, kt $NH_3$ $a^{-1}$ | | |
|---|---|---|---|---|---|
| | | | 1980 | 1985 | 1987 |
| Domestic animal | cattle | 18 | 34.5 | 31.8 | 29.9 |
| | pigs | 2.8 | 23.3 | 23.2 | 23.0 |
| | sheep, goat | 3.1 | 9.6 | 7.7 | 7.3 |
| | horse, buffalo ass, mule | 9.4 | 1.2 | 1.1 | 0.9 |
| | poultry, rabbit | 0.26 | 17.9 | 17.1 | 18.0 |
| Wild animal | deer, fallow-deer, moufflon | 9.4 | 2.2 | 2.8 | 2.7 |
| | wild hog | 2.8 | 0.0 | 0.1 | 0.1 |
| | hare, pheasant, partridge | 0.26 | 0.7 | 0.5 | 0.5 |
| Total | | | 89.4 | 84.3 | 82.4 |

Ammonia emissions of Hungary originating from fertilizers are the following:

| Year | Emissions, kt $NH_3$ $a^{-1}$ |
|---|---|
| 1980 | 47.6 |
| 1985 | 50.3 |
| 1987 | 51.2 |

(c) Industry

The Hungarian factories report on their yearly emissions which are summed in sectors for the counties. The yearly values change only slightly, and are approximately equal to 6.4 kt $NH_3$ $a^{-1}$.

(d) Soil exhalation

The $NH_3$ quantity released by soil exhalation was calculated from total territory of the country using emission factors given in section 1. It resulted in 9.9 kt $NH_3$ $a^{-1}$. The country-wide sum was distributed according to land-use practices of the counties.

In Table 2 ammonia emissions of Hungary calculated by different authors are tabulated. It appears that emission totals in this paper are in good agreement with other estimations, especially with $NH_3$ emissions considered for model calculations of EMEP MSC-W (Iversen et al. 1990) and IIASA (Klaassen, 1990).

Table 2 Ammonia emissions of Hungary according to source categories

| Reference | Year of determination | Emissions, kt $NH_3$ $a^{-1}$ | | | | | |
|---|---|---|---|---|---|---|---|
| | | Agriculture | | Industry | Soil Exhalation | $NH_3$ Oxidation | Total |
| | | Animals | Fertilizer | | | | |
| Present examination | 1980<br>1985<br>1987 | 89.4<br>84.3<br>82.4 | 47.6<br>50.3<br>51.2 | 6.4<br>6.4<br>6.4 | 9.9<br>9.9<br>9.9 | | 153<br>151<br>150 |
| Bónis (1981) | 1976 | 68—105 | 16—32 | 0.5—0.7* | 5—20 | —(0.7—3) | 89—154 |
| Buijsman et al. (1985) | 1982 | 83 | 42 | 4 | | | 130 |
| ApSimon et al. (1987) | 1980 | 69 | | | | | |
| Iversen et al. (1990) | 1988 and 1989 | | | | | | 151 |
| Klaassen (1990) | 1980<br>1987 | 91<br>99 | 51<br>55 | 4+3**<br>4+3** | | | 149<br>160 |

*Originating from coal combustion and traffic
**Originating from other anthropogenic sources

# 3 Ammonia emission inventory for Hungary

The ammonia emission inventories for years 1980, 1985 and 1987, which are used as input for the regional model, were produced for a grid with a resolution of 20km x 20km.

To determine grid by grid distributions of agricultural and industrial $NH_3$ emissions, we set out from corresponding values estimated for counties. The following data were used from map series titled "Planning economic regions of Hungary":

- numbers of cattle, pigs, sheep and horse,
- territories of forest,
- chemical fertilizers employed,
- industrial productions with $NH_3$ emission.

County emissions by soil exhalation were distributed in proportion to soil territories.

Emission contributions from different sectors and respective counties added yearly. In this way ammonia emissions were produced for territorial units of 20km x 20km in years 1980, 1985 and 1987. Finally the results were converted into an emission density of t $NH_3$ $km^{-2}a^{-1}$.

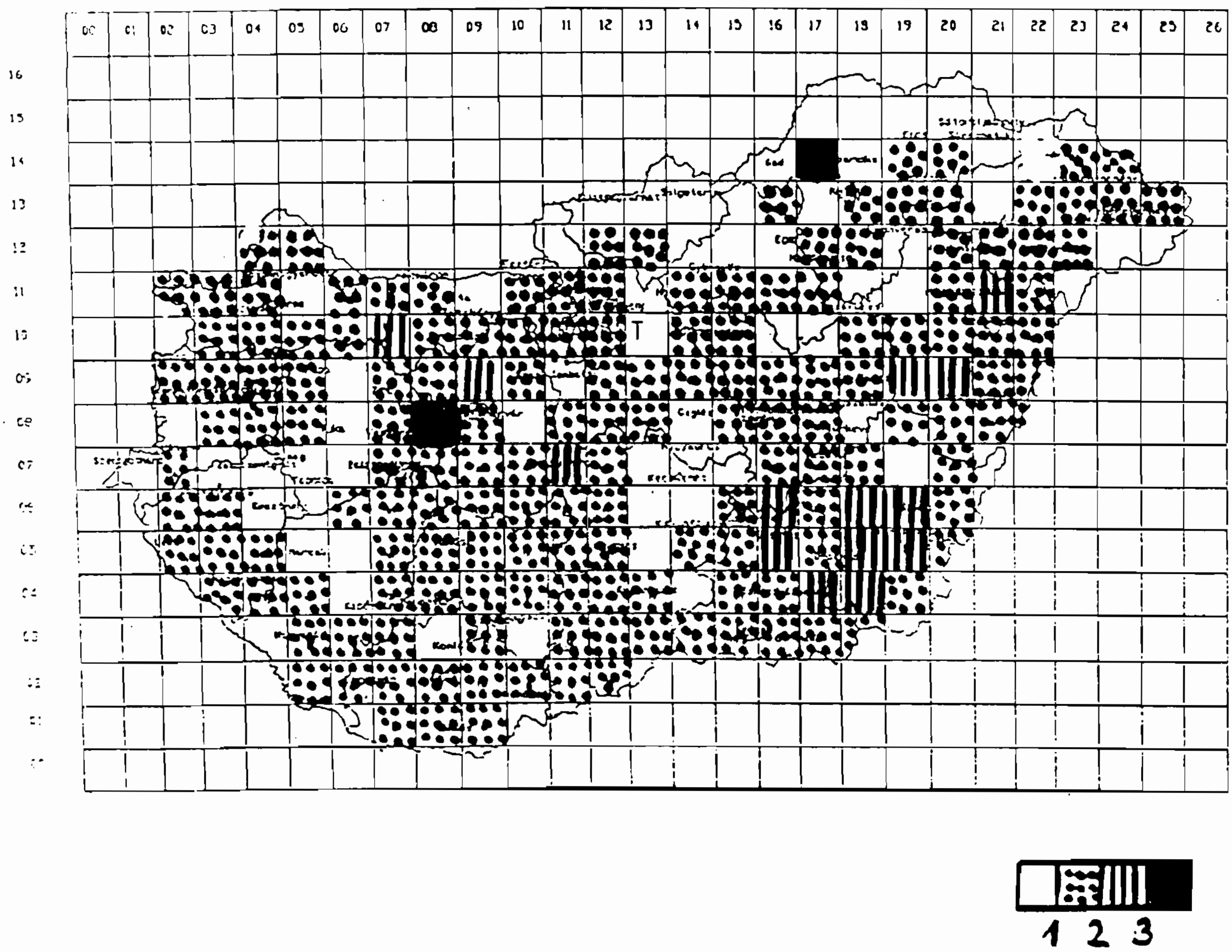


Figure 1 Regional scale ammonia emission density, t $km^{-2}a^{-1}$, of Hungary in 1987

For example, Figure 1 shows regional scale ammonia emission density of Hungary in 1987. Similar distributions are presented for years 1980 and 1985. The highest emission density (> 3t $km^{-2}a^{-1}$) is found in two grid cells where ammonia factories are located. Major contiguous areas of emission density between 2 and 3 t $km^{-2}a^{-1}$ originating from agricultural sources are in the south-east regions of the country.

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# AMMONIA EMISSIONS IN FINLAND

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## Abstract

The main source of ammonia emissions is agriculture, especially livestock manure. Industrial emissions contribute little to the total emissions, but can be locally significant. In Finland, fur farms are an important local source of ammonia. The agricultural emissions are calculated from statistical data on animal numbers, use of artificial fertilizers, and emission factors determined by combining Finnish agricultural practices with foreign ammonia evaporation data. The emission factors for fur animals are estimated assuming that 50 % of the nitrogen in the manure is emitted. Data on industrial emissions are obtained from notifications from the industry. Totally the calculated emissions for Finland amount to 43 kt $NH_3$ in 1985/86 and the emissions are expected to decrease in the future. The ammonia emissions in Finland are small compared to most countries in Europe. Even though, the deposition of ammonium compounds in Finland is estimated to account for nearly half of the total nitrogen load. Most of the deposition is assessed to originate from abroad, only about 35 % is estimated to come from domestic sources.

## 1 Introduction

The modelling of ammonia emissions has been a part of the Finnish Acidification Research Project (HAPRO) and included in the Finnish Integrated Acidification Model (HAKOMA). The Finnish model has been developed by the Technical Research Centre of Finland in co-operation with several other Finnish institutes and universities. The RAINS-model of IIASA has been used as a basis for the Finnish model, which contains modules for emissions ($SO_2$, $NO_x$ and $NH_3$), atmospheric dispersion and transformation, deposition and impacts on forest soils and lakes. A module for estimation of abatement costs is also included. The ammonia module contains data on Finnish and European emissions and deposition in Finland for the period of 1950 to 2000. The potential acidifying effect of ammonium deposition is considered in the model, but no other impacts or costs of emission reduction. European emissions are assumed to be the same as the ones used in the EMEP project (Iversen et al., 1990).

Ammonia emissions originate mainly from agriculture: livestock manure and use of artificial fertilizers. In Finland about 90 per cent of the emissions come from agriculture. Industrial emissions contribute little to the total emissions, but can be significant locally. Fur farms are also an important local source of ammonia, although their importance has decreased during the last two years because of large reductions in the animal numbers due to economical difficulties.

# 2 Finnish Ammonia Emissions

## 2.1 Emission Factors

Livestock

The most important source of ammonia is livestock manure. Ammonia evaporates from manure in the stables, during storage and application and from the meadows during the grazing period. The emissions are influenced by many factors such as:

- animal type, age and weight
- the amount and composition of nitrogen in the feed
- the length of the meadow (grazing) period
- management of manure in the stable and during storage and application
- weather conditions: temperature, wind and humidity.

In the Finnish model the emissions for cattle, pigs and poultry (laying hens and chickens) are calculated using municipal data on animal numbers and emission factors determined on the basis of estimates made by Niskanen et al. (1990). Their estimates are based on combining data of Finnish agricultural practices with foreign data on ammonia evaporation from manure. Factors like animal type and age, length of the grazing period and management of manure in the stable and during storage and application are accounted for in the estimate, whereas the differences in the nitrogen content of the feed and weather conditions are omitted.

Emission factors used in the model are 20 kg $NH_3$/animal/year for cattle, 3.5 kg $NH_3$/animal/year for pigs and 0.25 kg $NH_3$/animal/year for poultry.

The total ammonia emission from livestock for the year 1985/86 was about 38,000 t $NH_3$.

The emission factors used are averaged over animal type, age and manure management practices. A more detailed description on how these factors are determined is given for cattle. Table 1 is a comparison between emission factors used in the Finnish model and in the IIASA model (Klaassen, 1990) for cattle. The differences in the total emission factors are rather small (4% for dairy cows and 16% for other cattle) whereas the differences in the emission factors for storage and application are greater.

Table 1. Emission factors used in the Finnish HAKOMA-model compared with those used in the IIASA-model.

| Model | Emission factors/kg $NH_3$/animal/year | | | |
|---|---|---|---|---|
| | stall/storage | application | meadow | total |
| **Dairy and calf cows** | | | | |
| IIASA | 9.3 | 15.2 | 7.3 | 31.8 |
| HAKOMA | 13.8 | 11.5 | 7.9 | 33.2 |
| **Other Cattle** | | | | |
| IIASA | 2.1 | 3.9 | 3.8 | 9.8 |
| HAKOMA | 4.8 | 3.9 | 2.7 | 11.4 |
| **Cattle (all)** | | | | |
| HAKOMA | 8.3 | 6.9 | 4.8 | 20.0 |

These differences can be explained by the differences in manure treatment in central Europe and in Finland. In Finland most of the manure is stored as solid (about 80 % of cattle, 40 % of pig and almost all poultry and fur animal manure). According to Niskanen et al. (1990) the emissions during stall and storage are greater in solid manure treatment than for liquid manure. In application of manure in the fields the opposite is valid. The emissions during application could be even lower than given because most of the manure is spread in the fields and only a small fraction on grass and the manure is usually ploughed down (directly or within 24 hours).

Artificial fertilizers

In Finland mostly complex multinutritive fertilizers are used. They contain about 85 % of the nitrogen in artificial fertilizers sold in Finland. Other types of nitrogen fertilizers used in Finland are calcium/magnesium ammonium nitrates (12.1%) and urea (2.1%) and small amounts of ammonium sulphate. The total amount of nitrogen in the artificial fertilizers was about 205,000 t in 1985/86.

The emission factor used in the model (1.5 % of the nitrogen content in the fertilizers) was taken from estimates made by Buijsman et al. (1986). The ammonia emission was estimated to be about 3,700 t $NH_3$ in 1985/86. In Finland artificial fertilizers are often incorporated in the soil together with the seeds and the emissions are smaller than when applied on the surface.

Fur farms

In 1988 there were about 5,700 fur farms in Finland and the number of furs produced was to about 6.5 million. The fur farms are situated very close to each other and the animal densities in these regions are very large. The fur animals are mostly minks and foxes and their manure (dung) is very rich in nitrogen. The estimated (Helin 1982) amount of nitrogen in manure per fur produced is 1.5 kg for a fox and 0.9 kg for a mink. According to Ferm et al. (1988) 40 - 50% of the nitrogen evaporates to the atmosphere. From these figures an overall estimate of 0.65 kg $NH_3$/fur produced was obtained and used as emission factor in the model.

The emissions from fur farms were estimated to be about 4,000 t $NH_3$ in 1985/86. The emission densities in the fur animal regions were the highest in Finland in 1985/86 and the emissions have caused local disturbances in the growth of forests near the farms (Ferm et al., 1988).

## 2.2 Industrial emissions

Fertilizer and ammonia production plants are the main sources of industrial emissions. Smaller amounts of ammonia are emitted from mineral and glass-wool factories. Industry emits much less ammonia than does agriculture. In Finland the industrial emissions of ammonia are estimated to be about 1,300 t $NH_3$/a. The data is obtained from the notifications from the industry to the Ministry of Environment. The largest local sources of ammonia are fertilizer industries in Uusikaupunki (about 400 t/$NH_3$/a) and Kokkola and Oulu (each about 200 t $NH_3$/a).

## 2.3 Total emissions and emission densities

The total ammonia emissions calculated in the model for the reference year 1985/86 are given in Table 2. The emissions were calculated from municipal data on animal numbers, fertilizer use and industrial emissions. The emission density (municipal emissions divided by the area of the municipality) in Finland is given in Figure 1. The emissions are largest in the southwestern and western Finland. More than half of the pig and poultry farms, about 25 % of the cows and almost all the fur farms are located in these areas.

Table 2. Ammonia emissions in Finland in 1985/86.

| Source | Emission t $NH_3$ |
|---|---|
| Cattle | 28,200 |
| Pigs | 4,400 |
| Poultry | 1,500 |
| Artificial fertilizers | 3,700 |
| Fur farms | 3,900 |
| Industry | 1,300 |
| Total | 43,000 |

The emission inventory in the model does not contain emissions from broilers, sheep and horses. The emissions from these sources are estimated to be rather small, together about 1,200 t $NH_3$/a. According to Klaassen (1990) human population also emits ammonia, about 0.3 kg $NH_3$/human/a. In Finland this would amount to about 1,500 t $NH_3$/a. Other possible sources of ammonia are sewage treatment and sludge, but according to present knowledge the emissions are small, only hundreds of tons a year.

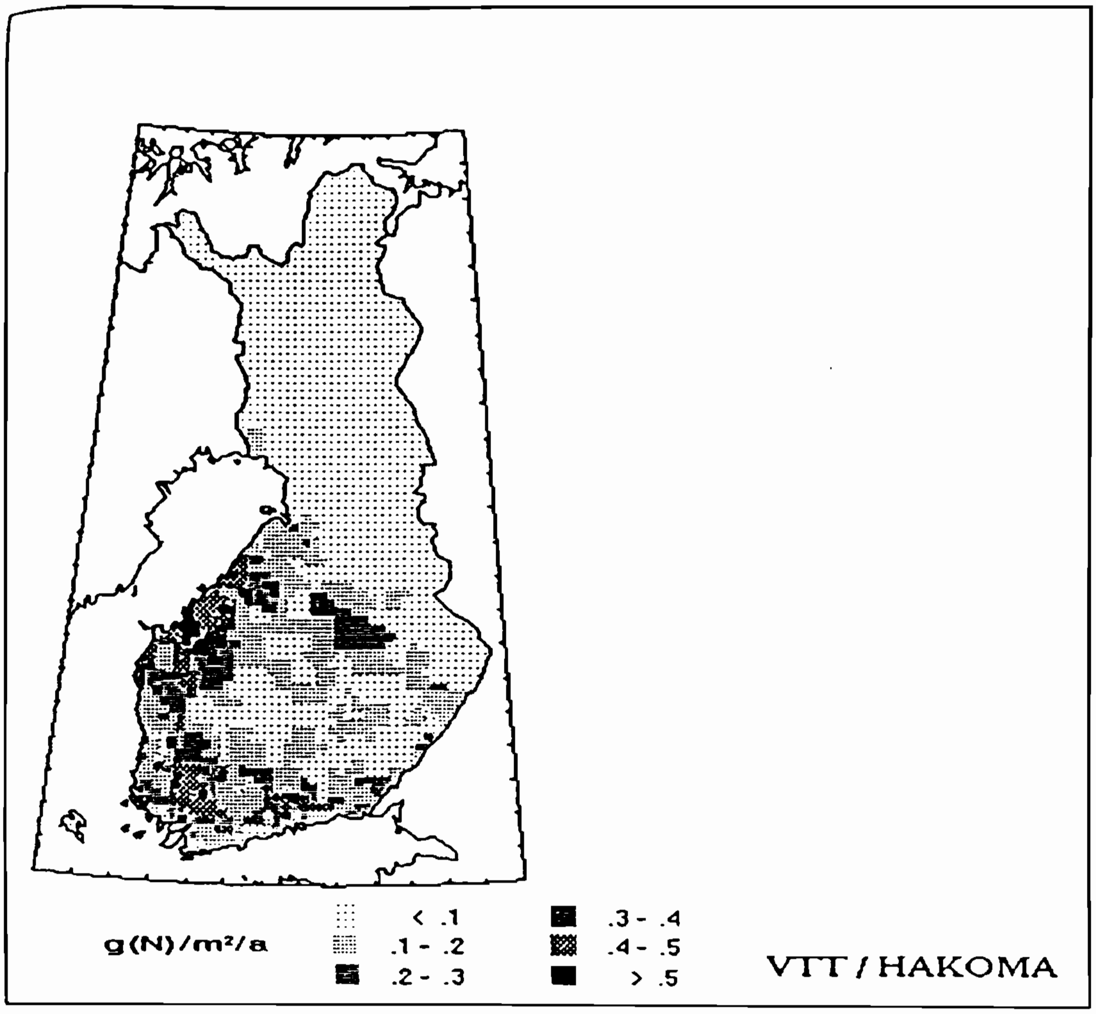


Figure 1. The geographical distribution of ammonia emissions in Finland in 1985/86

## 2.4 Historical and future emissions

The historical emissions (since 1950) have been calculated using the same emission factors as for the reference year 1985/86. It is possible that the historical emissions are overestimated because of this. Since 1950 the breeding of livestock has been intensified, for instance the milk production per cow a year is more than doubled. Some earlier manure treatment measures as covering the manure piles by clay also decreased ammonia losses from the manure.

At the end of the 1970s the number of cattle decreased by about 20% and has been decreasing ever since. The number of pigs and poultry has increased slightly since 1950. The production of furs is relatively new in Finland. The establishment of farms began in the late 1960s and the production of furs was most intensive in the late 1980s. Industrial emissions have been steadily growing since 1950, although they have not changed much during last decade.

In the future, the ammonia emissions are estimated to decrease by 25% on all sectors. According to Finnish agricultural forecasts the number of livestock will decrease. The number of dairy cows is expected to decrease by some 20 % before the year 2000. The reduction in the number of pigs and poultry is expected to be less. During the last few years the fur farms have had severe economic difficulties. The prices of furs have gone down and many farms have closed. The reduction in the fur animal numbers has been even larger than estimated in the model. Industrial emissions will be reduced because of rationalization of the production. Technical abatement of the emissions is not expected as most of the plants are already using the best available techniques for emission reduction.

A schematic presentation of the historical and future emissions in Finland is given in Figure 2.

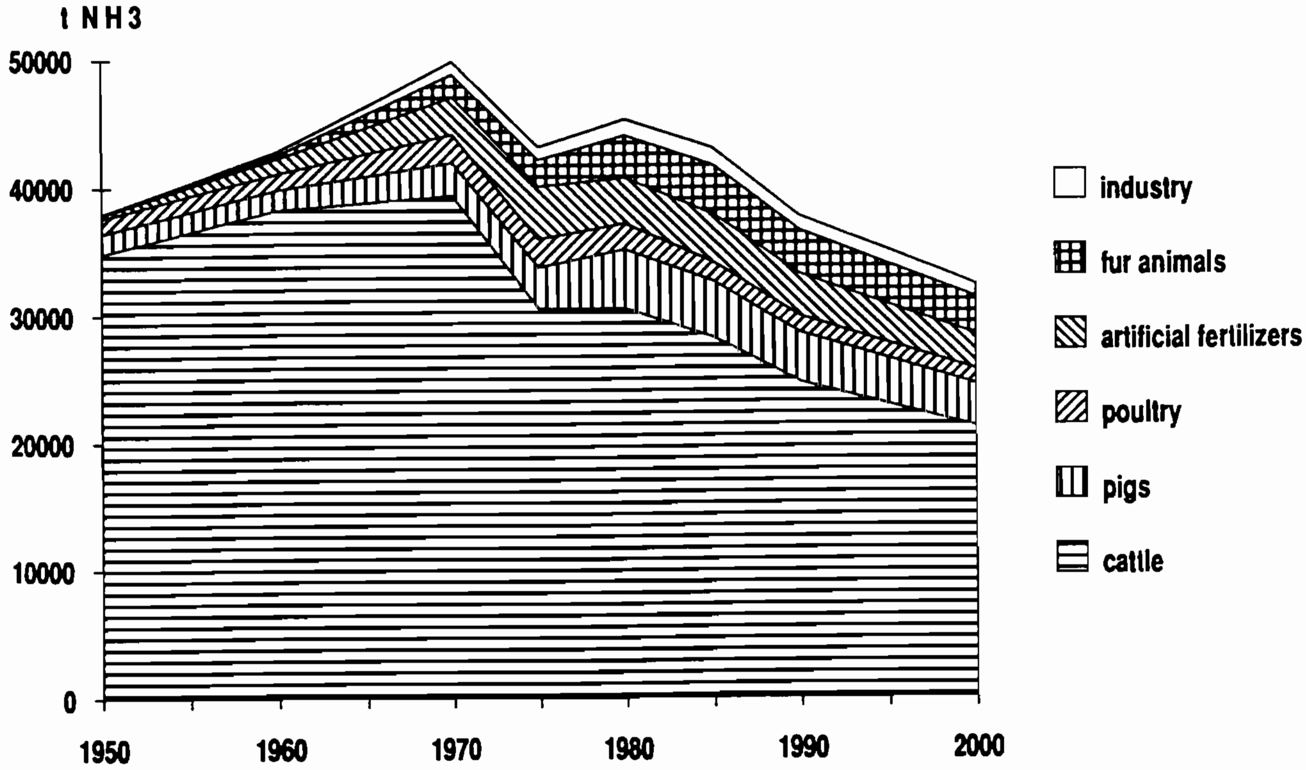


Figure 2. Historical and future emissions in Finland

# 3 Deposition in Finland

The calculated ammonium deposition in Finland is presented in Figure 3. The deposition from Finnish emissions is low (< 0.02 - 0.08 $g(N)/m^2/a$) in most parts of Finland. In the fur animal region and in the southwest of Finland where the pig and poultry densities are high, the deposition values are also higher. The total deposition is much higher (0.02 - 0.4 $g(N)/m^2/a$). According to the latest EMEP calculations, only about 35 % of the deposition in Finland is originating from domestic sources (Figure 4). The total ammonium deposition in Finland is estimated to account for about 40 % of the total nitrogen deposition. The nitrogen load in Finland, though smaller than in many other European countries, is estimated to cause disturbancies in the forests in the long run.

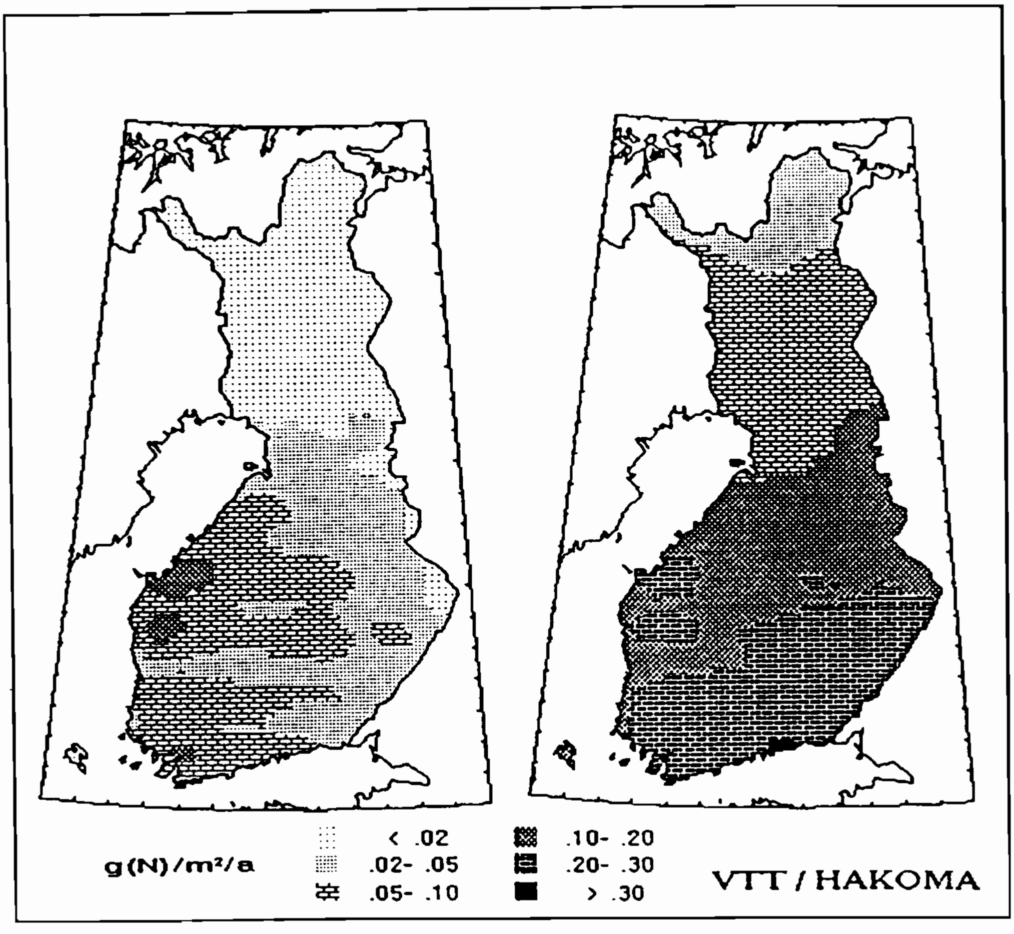


Figure 3 Nitrogen deposition from ammonia emissions in Finland in 1985/86. The left map shows the deposition from Finnish emissions, the right the total deposition.

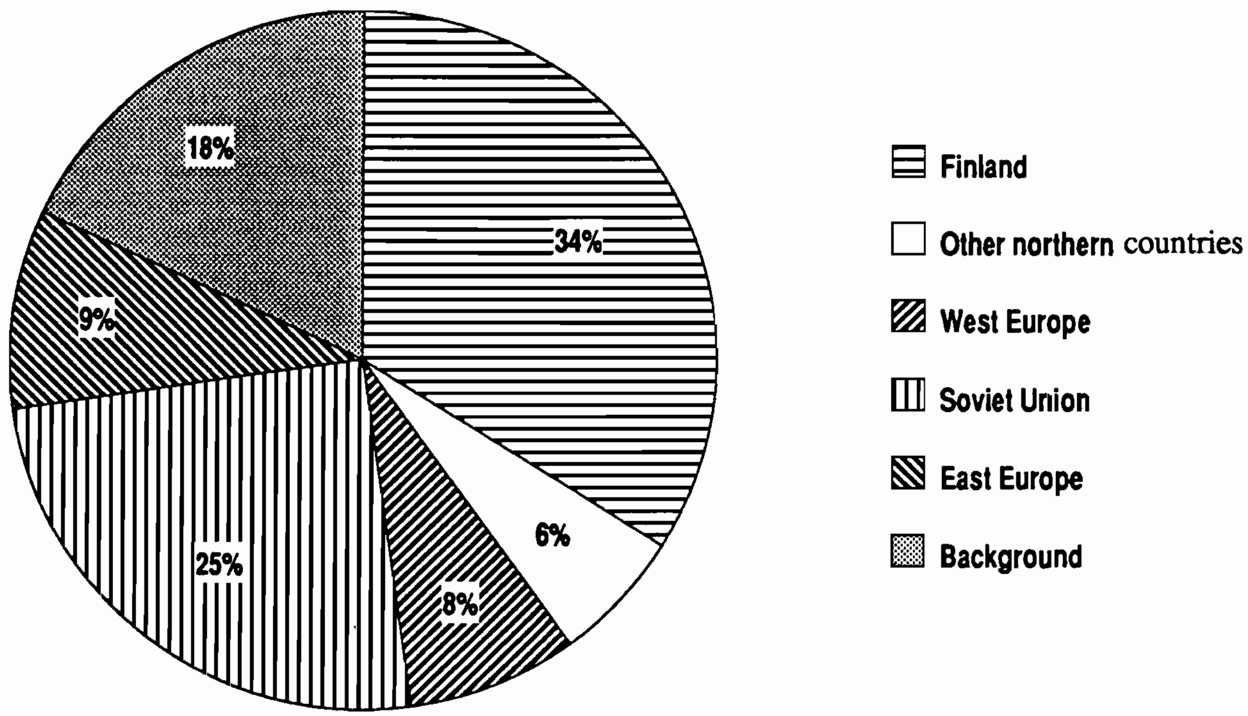


Figure 4 The origin of ammonium deposition in Finland in 1989 (Iversen et al. 1990)

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# AMMONIA BUDGET FOR THE FORMER GDR

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## Abstract

The $NH_3$ emissions in the former GDR (the new German countries) have been estimated to be 350 kt N $a^{-1}$. The uncertainty lies in the order of 40 %. Between 1950 and 1970 the $NH_3$ emission increased by 70 % and remains nearly constant up to 1990. The total deposition amounts to approximately 225 kt N $a^{-1}$. Around 1/3 of the east German ammonia emission goes into the long range transportation outside the territory.

# 1 Introduction

The role of ammonia in the atmospheric chemistry is well known, especially in buffering the liquid phase as an alkaline reagent and in an enhanced removal of $SO_2$ from the atmosphere (e.g. Möller and Schieferdecker, 1985). Ecological impacts, such as plant damage and nitrogen deposition, are widely discussed. Based on these facts, it is a little surprising that $NH_3$ is missing in most pollutant transport/transformation models. One reason (beside the "uninteresting" air chemistry) should be the uncertainties of the emission figures. In this paper our emission and deposition estimates for the former GDR (Möller and Schieferdecker, 1982; 1985; 1989; 1990) are presented comprehensively.

# 2 Emission

The estimation of mean emission factors for the biogenic $NH_3$ emission from soils and animal excrements is a serious problem and the reason for the large

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differences in emissions found in literature. This is most relevant for livestock farms with a wide range of living and processing conditions (Table 1). Variations of the $NH_3$ emission, based on the N turnover by animal, are given between 10 and 50 %.

Due to the relatively low share of mineral fertilizer application in the total $NH_3$ emission, variations in emission factors are of minor importance. Our estimation, taking into account different fertilizer types, is 0.08 % of the N content.

Industrial processes (coking, coal combustion, travel, $NH_3$ synthesis, N fertilizer production) are, at present and in the future, negligible in comparison to the $NH_3$ emission from animals. This is given due to the development in low-waste technologies.

It is very likely that our earlier estimations of the "natural" $NH_3$ emission from soils (estimated to be 3 kg N $ha^{-1}$ $a^{-1}$) as well as human emission (sewage, direct exhaust) of 1.3 kg N $man^{-1}$ $a^{-1}$ are underestimated.

Table 1 N turnover and mean emission factor for animals from livestock in the former GDR.

| Animal | N production in kg N $a^{-1}$ | $NH_3$ emission factor in kg N $animal^{-1}$ $a^{-1}$ |
|---|---|---|
| cattle | | 26.1 |
| milk cow | 100 | 35.0 |
| fattening cow | 44 | 15.4 |
| pigs | 17.2 | 5.2 |
| poultry | 0.73 | 0.22 |
| horses | 50 | 15 |
| sheep | 12 | 3 |

All emissions are summarized in Table 2. Biogenic processes amount to 95 % to the total $NH_3$ emission with the biggest share from livestock (2/3). The contribution from mineral fertilizers continuously increased from 7% (begin 1950's) up to 20% (end 1980's).

Table 2 $NH_3$ emission in the former GDR in kt N $a^{-1}$

| Year | Cattle | Pigs | Poultry | Other animal | 'Natural' and man | Fertilizer | Industry | Sum |
|---|---|---|---|---|---|---|---|---|
| 1950 | 87.5 | 29.4 | 5.0 | 19.0 | 50 | 15.0 | 3 | 209 |
| 1955 | 99.1 | 46.6 | 6.0 | 18.1 | 50 | 16.8 | 6 | 243 |
| 1960 | 114.8 | 42.9 | 8.1 | 14.2 | 50 | 18.8 | 12 | 259 |
| 1965 | 115.9 | 45.8 | 8.4 | 10.5 | 50 | 33.1 | 16 | 280 |
| 1970 | 122.3 | 50.0 | 9.5 | 7.2 | 50 | 40.9 | 18 | 298 |
| 1975 | 127.5 | 59.3 | 10.4 | 6.9 | 50 | 54.2 | 25 | 333 |
| 1980 | 130.0 | 66.4 | 11.4 | 7.3 | 50 | 60.0 | 30 | 355 |
| 1985 | 130.0 | 66.5 | 11.1 | 8.1 | 50 | 61.6 | 17 | 345 |
| 1988 | 130.0 | 64.0 | 10.9 | 8.1 | 50 | 69.6 | 17 | 350 |

# 3 Deposition

Taking into consideration the problems with sampling and storage of rain water samples, the estimation of regional wet ammonium deposition is easy. $NH_4^+$ in precipitation water is homogeneously distributed within the former GDR; no trends have been observed from 1982 for selected stations. Based on all figures, an average of 1.8 mg $NH_4^+$-N $l^{-1}$ seems to be representative. This figure, however, should be corrected by a 20% increase due to the ammonium instability in the first 2 weeks between sampling and analysis. In this way the total wet deposition can be assessed to be 130 kt N $a^{-1}$. The uncertainty is less than 10 %.

A hard problem, however, is the estimation of the $NH_3$ dry deposition, because the $NH_3$ gas concentration is widely unknown and the selection of mean dry deposition velocities $v_T$ is very difficult. Following the discussion in Möller and Schieferdecker (1990) a figure of 0.3 cm $s^{-1}$ should be appropriate for agricultural areas, whereas variation for the actual $v_T$ between 0 (emitting soil) and 2 (no emitting wet soil) should be taken into account. The dry deposition into forest ecosystems is more efficient and a $v_T$ between 2 and 3 cm $s^{-1}$ seems to be reasonable.

First measurements of $NH_3$ and $NH_4^+$ were taken near our institute in the southern part of Berlin using the denuder-filter technique. The results are 4.5 +- 2.5 $\mu$g $NH_3$ $m^{-3}$ and 3.8 +- 1.4 $\mu$g $NH_4^+$ $m^{-3}$. Calculated figures from Asman (1990) are 2.5 $\mu$g $NH_3$ $m^{-3}$ and 4.7 $\mu$g $NH_4^+$ $m^{-3}$, resp., based on GDR's $NH_3$ emission of 274 kt $a^{-1}$.

Assuming a figure of 3 $\mu$g $NH_3$ $m^{-3}$ as representative one for the whole territory a total dry deposition of 95 kt N $a^{-1}$ is following, taking into account a forest area share of 27 % and the above discussed $v_T$ figures. The uncertainty is at least in the order of 50%. The particulate $NH_4^+$ dry deposition seems to be totally negligible (Möller and Schieferdecker, 1990).

# 4 Budget

The ratio of wet deposition to emission $D_w/Q$ amounts to 0.37, which is very similar to the estimation of 0.41 by Buijsman and Erisman (1986) and Asman and Janssen (1987). The following budget equation is valid.

$$\frac{D}{Q}=\frac{D_w}{Q}+\frac{D_d}{Q}=1-\frac{EX-IM}{Q} \tag{1}$$

where EX and IM denote the export and import of the atmospheric trace constituent. From our dry deposition estimation it follows $D_d/Q = 0.27$ and according to Equation (1), $D/Q = 0.64$. This result is in agreement with the finding by Asman (1990) for GDR's $D/Q = 0.64$ (= 175/274 in kt) by using a European transport/transformation model despite different absolute emission and deposition figures. From Equation (1) it follows that EX - IM = 125 kt, i.e. around 1/3 of the East German $NH_3$ emission will be exported outside the country.


# Acknowledgements.

I would like to express my thanks to Dr. Karin Acker for her $NH_x$ measurements.


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# THE AMMONIA EMISSION REGISTER FOR THE FIVE NEW "LÄNDER" IN GERMANY

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## Abstract

The $NH_3$ emissions have been evaluated for the territory of the five new "Länder" for 1989. Thereby, the emissions of the animal production, fertilizer losses, municipal wastewater and industrial emissions have been taken into consideration. Hence follows a total value of about 300 kt N/y. For the first time, these emissions were recorded in a register on the basis of 10*10 $km^2$ grids and were thus integrated into the Länder and counties by areas.

## 1 Introduction

After sulfur dioxide in the late eighties, atmospheric nitrogen compounds among the air pollutants have become more and more important. Within the framework of UNECE an agreement on the reduction of nitric oxide emissions was signed on November 1, 1988, as successor of the $SO_2$ protocol.

Beside $NO_x$ also reduced nitrogen compounds, particularly ammonia, contribute to the atmospheric deposition of compounds containing nitrogen with about 50%. The role ammonia plays in the acidification of soil and its contribution to the recently arising forest damages is regarded as uncontested.

Consequently, a registration by single areas is necessary in order to include the reduced atmospheric nitrogen compounds into the calculations of the emission, long range transport and deposition. In the following, for the first time an ammonia emission register covering the entire area of the five new "Länder" is represented in a grid of 10*10 $km^2$.

Table 1 shows the present ammonia emission values for the territory of the former GDR. The determining large range of values has been suggested to evaluate again the emissions for 1989 because of the new specific emission factors.

**Table 1.** Ammonia emissions for the five new "Länder" (literature)

| | kt N | Year |
|---|---|---|
| Möller (1982) | 423 | 1980 |
| Möller (1985) | 150 | 1980 |
| Möller (1989) | 345 | 1985 |
| Buijsman (1987) | 170 | beginning 1980s |
| ApSimon (1987) | 122* | 1980 |

* only for livestock animals

# 2 Anthropogenic sources

The most important sources of ammonia are the facilities for animal production, fertilizer losses, municipal wastewater as well as plants producing fertilizers. Table 2 gives an overview of all the animal quantities kept at the territory of the five new "Länder" (Staatliche Zentralverwaltung für Statistik, 1990a). The figures for the animals are subdivided by districts, counties and large farms.

**Table 2.** Livestock animal population (October 1989) in the five new "Länder" (SZS, 1990a)

| | Sum | In large farms* | Number of large farms | Share of large farms |
|---|---|---|---|---|
| Cattle | 5,724,421 | 2,081,677 | 448 | 36.4 |
| Pigs | 12,012,723 | 3,015,864 | 143 | 25.1 |
| Sheep | 2,602,692 | 1,030,690 | 387 | 39.6 |
| Hens | 24,865,817 | | | |
| Broilers | 10,925,166 | | | |
| Ducks | 800,105 | | | |
| Geese | 2,306,215 | | | |
| Turkeys | 727,304 | | | |

* for cattle >3,000 animals, pigs >10,000 animals and sheep >1,500 animals

The specific emission factors are summarized in Table 3. Values by Erisman have been used for the evaluations. The evaluation of ammonia emissions from municipal wastewater proceeds from the assumption that the total population amounts to 16.2 millions and the specific emission is 3.4 kg N/capita per year.

**Table 3.** Specific emission coefficients for ammonia from livestock animals (kg N/y)

| | (1) | (2) | (3) | (4) |
|---|---|---|---|---|
| Cattle | 14.9 | 22.1 | | 24.0 |
| Fattening cows | | 15.4 | | |
| Dairy cows | | 35.0 | 29.25 | |
| Horses | 7.76 | 15.0 | 10.29 | |
| Sheep, goats | 2.53 | 3.0 | 3.35 | |
| Pigs | 2.33 | 5.2 | 5.06 | 5.14 |
| Poultry | 0.21 | 0.22 | 0.3 | 0.36 |
| Broilers | | | 0.231 | |
| Ducks, geese | | | 0.096 | |
| Turkeys | | | 0.706 | |

(1) Asman (1987)
(2) Möller (1989)
(3) Erisman (1989)
(4) Isermann (1990), using 36 kg N/large animal unit (Großvieheinheit)

As far as industrial enterprises are concerned, the nitrogen producing factory Piesteritz with an emission of 2800 t N/y (Kraftcik, 1990) and the fertilizer plant in Rostock with 58 t N/y (Boehm, 1990) were taken into consideration.

With respect to the mineral fertilizers the total nitrogen yield was classified by districts and separated by arable land and permanent grassland (Staatliche Zentralverwaltung für Statistik, 1990b). According to the usual fertilizer assortment the complete nitrogen quantity was subdivided in the former GDR. Considering the fact that losses from ammonium sulphate do only occur at strongly calcareous soils (Thuringian region, 390,000 ha) the nitrogen losses were calculated out of it. The utilized fertilizer as well as the appropriate nitrogen losses are represented in Table 4.

**Table 4.** Consumption of nitrogen fertilizer and losses (IfD, 1990)

| | Share of total N consumption (%) | N loss (%) |
|---|---|---|
| Ammonium sulphate | 22 | 10 |
| Ureum | 23 | 3.4 |
| Ammonium nitrate | 48 | 0.5 |
| Ammonium nitrate-urea | 5 | 1.1 |
| Complex fertilizer | 2 | 0.5 |

# 3 Natural sources

The most important natural sources are wild living animals and nitrogen losses of unmanured soil. That is why an ammonia emission of 2.3 kt N/y is used for wild living animals. The specific emissions have been estimated because of their weight taking into account that one animal unit corresponds to about 500 kg of life weight (Isermann, 1990).

**Table 5.** Ammonia emissions for the five new "Länder" (1989)

| | Emission factor (kg N/y) | Emissions (kt N/y) | Share (%) |
|---|---|---|---|
| Cattle | 26.14 | 149.52 | 50 |
| Pigs | 5.06 | 60.77 | 20.5 |
| Sheep | 3.35 | 8.71 | 3 |
| Laying hens | 0.3 | 7.46 | |
| Broilers | 0.231 | 2.52 | |
| Ducks, geese | 0.096 | 0.298 | |
| Turkeys | 0.706 | 0.51 | |
| Poultry | | 10.788 | 3.5 |
| Fertilizer losses | | 10.238 | 3.5 |
| Industry | | 2.858 | 1 |
| Municipal wastewater | | 55.08 | 18.5 |
| Sum | | 297.964 | 100 |

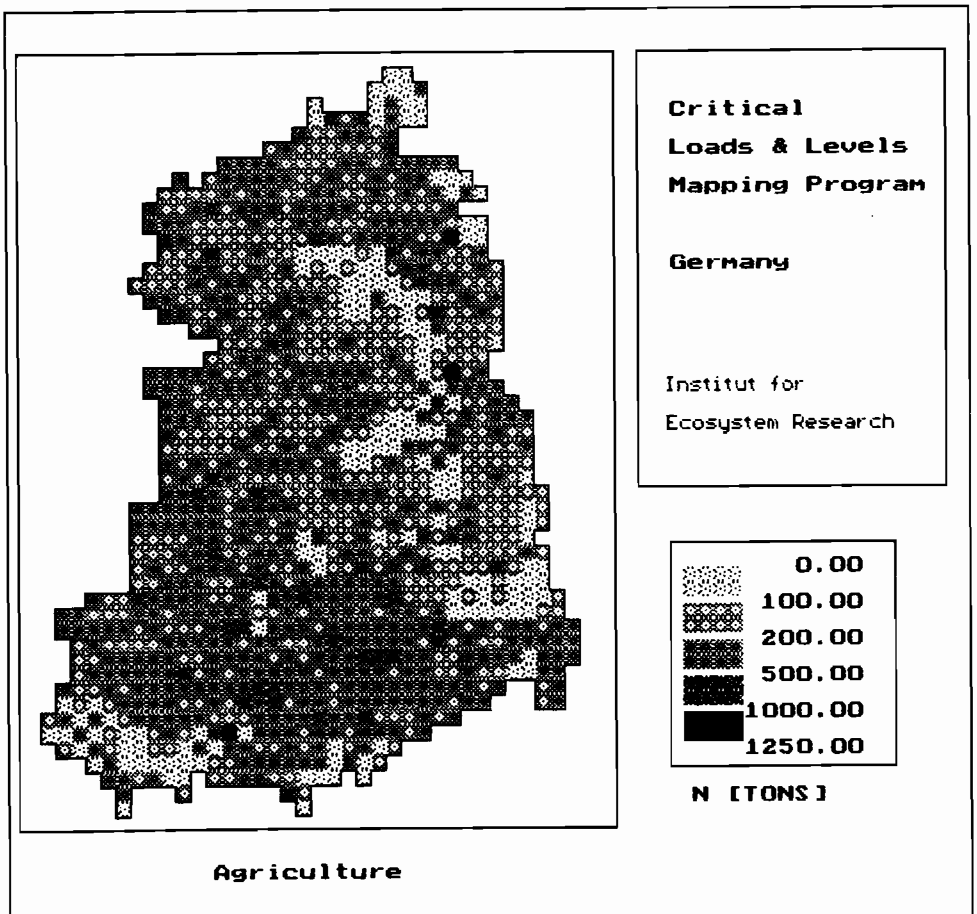
The emissions of natural soils can be calculated with 0.09 kg N/ha*y for an effective forest area of about 2,963,100 ha. This implies a total quantity of 219,618 kg N/year.

The natural emissions are not considered in the emission register because they are not very high and cannot be classified into the grids. In Table 5 all ammonia emissions are depicted for 1989.

# 4 The ammonia emission register

For the territory of the five new "Länder" a register exists on the basis of 10*10 $km^2$ grids. The single shares of forest (detached by deciduous and coniferous forest), arable land, permanent grassland, water, cultivated areas as well as the residential population are recorded for each grid. The grid areas are integrated into the districts and counties. The separately presented livestock farms were classified to one particular grid under the assumption that the entire ammonia emission (stable, storage and spreading of the animal wastes) develops in this grid. Those animals which are not kept in large plants were evenly distributed in circles on the corresponding grids.

Figure 1. Ammonia emission from agriculture



In the same way, emissions from municipal wastewater (by the distribution of the residential population) and nitrogen losses emerging from fertilization (by the distribution of the agricultural area) respectively industrial plants can be integrated in the equivalent grid. The Figures 1 - 3 show the territorial distribution of the selected sources respectively of the total emission.

Figure 2. Ammonia emissions from communities

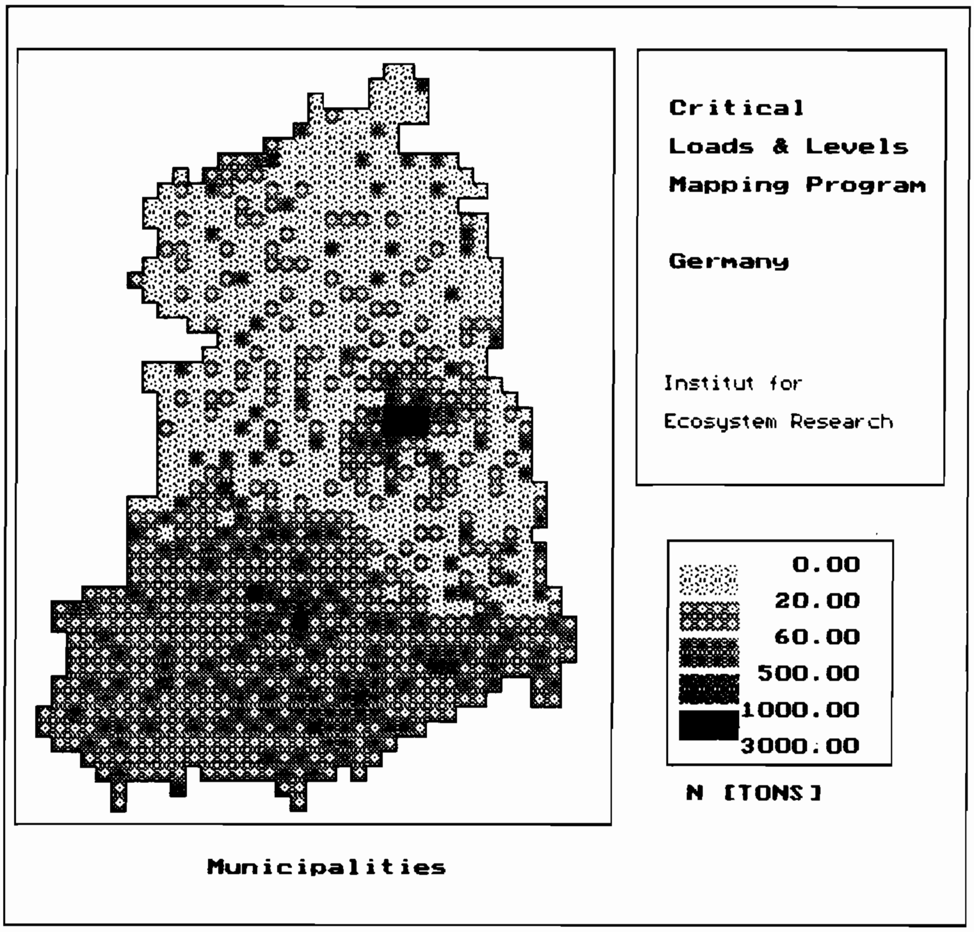
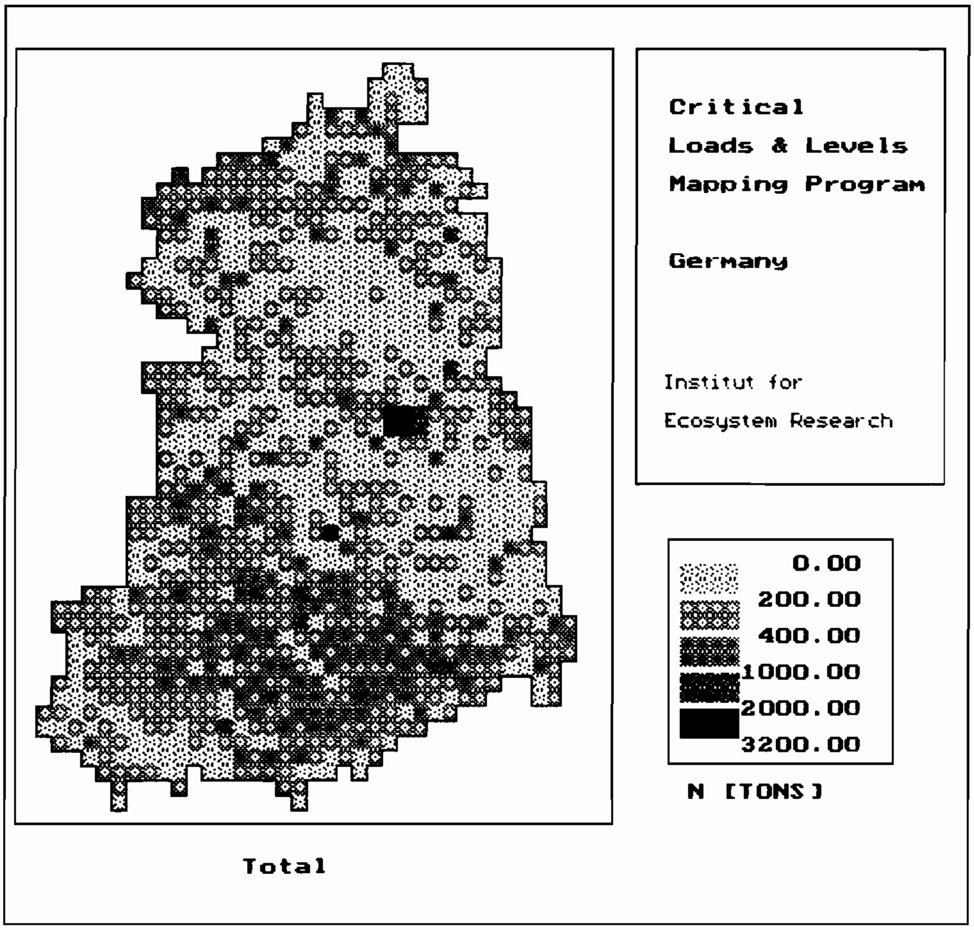


Figure 3. Total ammonia emissions

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# AMMONIA EMISSIONS IN THE FEDERAL REPUBLIC OF GERMANY

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## Abstract

Scientists recently pointed out the negative effects of increased ammonia emissions, the main source being livestock farming. In Western Europe this problem seems to be specially prominent in the Netherlands. The average emission density in the Federal Republic of Germany has a level of about 15 kg $NH_3$ /hectare referring to the total area. About 290,000 tons per year or 78 per cent of the total ammonia emissions in the Federal Republic of Germany are contributed by livestock farming. Depending on animal species, between 20 and more than 50 per cent of the excreted nitrogen is emitted as ammonia in the air. Other factors affecting emissions include stable systems, storage and application of the excrements. In the Federal Republic of Germany, cattle contribute approximately 60 per cent, pigs 13 per cent, poultry 2 per cent to the total emissions and other animals, such as horses and sheep, 2 to 3 per cent. Chemical fertilizers contribute approximately 15 per cent. Other non-agricultural sources, such as internal combustion engines, combustion processes and municipal waste are of relatively minor importance.

# 1 Introduction

During the establishment of the factors causing forest damage, evidence arose that ammonia emissions contribute to the possible damage of forests as well as to the degree of contamination of soils and ecosystems. Ammonia emissions in Europe amount to 7 million tons of which livestock farming contributes 80 per cent. Among the countries compared, the Netherlands have the highest emission density and France the highest total amount of ammonia emissions. Based on the total area, the emission density of the Federal Republic of Germany is at an average of 15 kg $NH_3$ /hectare.

Ammonia emissions have both agricultural and non-agricultural sources. Due to its relative importance, this paper directs its attention solely to agricultural ammonia emissions. A monitoring network for ammonia emissions across the whole Federal Republic of Germany does not exist at present. For this reason, different sources of information have been analyzed for data concerning the amount of ammonia emissions.

The majority of the data was provided by Dutch authors since the problems were first noticed in the Netherlands due to their high settlement and high livestock rates. Although agriculture in the Netherlands is more intensive than in the Federal Republic of Germany, the degree of self-sufficiency in the FRG has increased by more than 100 per cent. During the same time, there was a decrease in population growth and agricultural land use. The area necessary for the production of food for one person has decreased from 0.3 to 0.2 hectare today. In the course of this development, the average size of the farms increased. Specialization in agricultural production (when related to total agricultural area) has led to arable land increasing from 55 per cent in 1970 to 60 per cent today while conversely the area of grassland has receded.

These structural changes in turn led to an increase in livestock numbers over the last 30 years from 11.8 (BML, 1958) to 14 million LU[1] (BML, 1988). At the same time, the number of livestock owners decreased, thus livestock farming became more concentrated. Traditional livestock farming changed, becoming more region specific with areas specializing in raising cattle or pigs. Therefore, some areas in the North and Northwest of the Federal Republic of Germany have comparable conditions to those in the Netherlands.

# 2 Sources of ammonia emissions

## 2.1 Introduction

An important source of agricultural ammonia emissions is livestock farming. Storage and application of excrements contribute to a high ammonia release. In addition, the application of artificial fertilizers contributes to emissions. Non-agricultural sources such as industry, traffic, combustion processes, and municipal waste are of minor importance.

The ammonia emissions caused by the different sources are given at the end of this paper.

## 2.2 Sources of agricultural emission

### 2.2.1 Emissions from livestock farming

Ammonia is formed by microbiological decomposition of organic materials such as excrement and fodder and is therefore a traceable element in stables. The

[1]LU = Large Animal Unit ("Großvieheinheit"), see Table 4.

amount of ammonia released from animals depends on the nitrogen content of the fodder. Feeding more proteins than the animals can digest leads to a high content of $NH_3$ and urea in their excrements. Of particular concern is feed for pigs. When the proportion of protein is too high, the surplus protein is decomposed and metabolized into urea by the liver and then excreted (Gädeken, 1986).

In ruminants, high amounts of ammonia are also built through microbiological decomposition of non-protein-nitrogen. The direct release (via respiration) of ammonia from animals is not of importance. It cannot be altered distinctly by feeding. Indirectly however, the excretion of nitrogen via manure and urine can be relevant and can lead to $NH_3$ concentrations harmful to the environment.

Different stable systems are much more important for ammonia emissions than feeding (Table 1). With straw on the stall floor, more nitrogen is absorbed and less $NH_3$ released than with manure slurry (Sauerbeck, 1986). Housing systems of the Federal Republic of Germany are becoming progressively more comparable to those of the Netherlands. Therefore the application of parameters from the Netherlands to German conditions seems valid.

Table 1: Ammonia emissions from different housing systems for fattening pigs.

| Animal type/housing system | Per animal | | Per LU |
|---|---|---|---|
| | mg $NH_3$/h | kg $NH_3$/a | kg $NH_3$/a |
| Fattening pigs: | | | |
| -Danish system | 214 | 1.870 | 11.7 |
| -Partly slatted floor | 396 | 3.468 | 21.7 |
| -Fully slatted floor | 220 | 1.927 | 12.0 |

Reference: Eerden et al. (1981)

The noticeable trend is confirmed by the Kuratorium für Technik und Bauwesen in der Landwirtschaft, KTBL (Hüffmeier, 1977) (Table 2).

Table 2: Gas concentrations inside and outside pig stables.

| Average gas concentrations inside pig stables | |
|---|---|
| Deep stable with litter | 28 ppm |
| Solid dung stable with litter | 14 ppm |
| Manure slurry stable | 7 ppm |
| Average gas concentrations outside pig stables (in relation to 40 $m^3$ of waste air per animal and hour) | |
| Deep stable | 36.0 ppm |
| Solid dung stable | 18.0 ppm |
| Manure slurry stable, underground drainage | 13.5 ppm |
| Manure slurry, side drainage | 10.5 ppm |

Source: Hüffmeier (1977)

There are great differences between the ammonia emissions of different animal species (compare Table 3, 4 and 5). These differences will be discussed in more detail in the last section of this paper.

Table 3: Average N production and $NH_3$ emissions from animal excretion per annum.

| Animal Type | N Production | | $NH_3$ Emission | | $NH_3$ -N Emission |
|---|---|---|---|---|---|
| | N kg/animal | N kg/LU | $NH_3$ kg/animal | $NH_3$ kg/LU | In % of N production |
| Cattle | 64.0 | 64.0 | 18.0 | 18.0 | 23.2 |
| Pigs | 13.0 | 81.3 | 2.8 | 17.5 | 17.7 |
| Poultry | 0.48 | 24.0 | 0.26 | 13.0 | 44.6 |
| Horses | 34.0 | 34.0 | 9.4 | 9.4 | 22.8 |
| Sheep/Goats | 12.0 | 120.0 | 3.1 | 31.0 | 21.3 |

Source: BEF (1989) based on Buijsman (1985) and Buijsman et al. (1985)

Table 4: Definition of LU

| | |
|---|---|
| Horses (3 years and older) | 1.100 |
| Calves and young cattle < 1 year | 0.300 |
| Cattle 1 - 2 years | 0.700 |
| Dairy cows and cattle > 2 years | 1.000 |
| Piglets | 0.020 |
| Breeding pigs 20 - 50 kg | 0.060 |
| Fattening pigs > 50 kg | 0.160 |
| Breeding sows and boars | 0.300 |
| Sheep (≥ 1 year) | 0.100 |
| Poultry all | 0.004 |

Source: BML (1988)

Table 5: Ammonia emissions of different species (kg $NH_3$ /a)

| Animal type | Stall and Storage per | | Application per | | Meadow per | | Total per | | As % of N-Production |
|---|---|---|---|---|---|---|---|---|---|
| | Animal | LU | Animal | LU | Animal | LU | Animal | LU | |
| Cattle | 6.7 | 6.7 | 8.5 | 8.5 | 9.7 | 9.7 | 24.9 | 24.9 | 23.3 |
| Fattening calves | 1.0 | 3.3 | 2.6 | 8.7 | - | - | 3.6 | 12.0 | 50.0 |
| Fattening pigs | 1.8 | 11.3 | 1.5 | 9.4 | - | - | 3.3 | 20.7 | 18.0 |
| Laying hens | 0.15 | 7.5 | 0.12 | 6.0 | - | - | 0.27 | 13.5 | 36.0 |
| Sheep (≥ 1 year) | - | - | - | - | 3.4 | 34.0 | 3.4 | 34.0 | 45.0 |
| Horses (≥ 3 years) | 5.0 | 4.5 | 4.0 | 3.6 | 3.5 | 3.2 | 13.0 | 11.3 | 22.6 |

Source: BEF (1989): based on Janssen (1985)

Another source of $NH_3$ emissions is the preservation of fodder with ammonia. This source contributed only 1 per cent and is consequently not further discussed (Küntzel and Pahlow, 1980).

The combined emissions from storage, conditioning and application of solid dung and slurry are high. The amount is influenced by the housing system, the time that the manure remains in the stall, temperature and wind conditions, the conditioning technology, the kind of application and other factors of minor importance. These emissions cover a wide range between 0 and 90 per cent (Sauerbeck, 1985) (Table 6).

Table 6. Losses of manure-nitrogen.

| Process | Storage<br>Preparation<br>Stabilization | Application<br>Spreading | In the soil<br>Washing out<br>Denitrification |
|---|---|---|---|
| % Loss: | 30 — 90 | 0 — 50 | 0 — 50 |
| Form: | $NH_3$, $N_2$ | $NH_3$ | $NO_3$, $N_2$, $N_2O$ |

Reference: Sauerbeck (1985)

We have no detailed results of emissions from other organic components used by agriculture, such as sewage sludge, waste-sewage-sludge-compost and similar.

### 2.2.2 Ammonia emissions from fertilizers

The total amount of nitrogen fertilizer applied in the Federal Republic of Germany in 1987/88 was 1.6 million tons (BML, 1988). This corresponds to an average quantity of 134 kg N per hectare based on the total area of land in agricultural use. Depending upon fertilizer type, the amount of emitted nitrogen ranges between 2 and 15 per cent of the total nitrogen.

With a loss of 2 per cent calcium ammonium nitrate has a low emission factor (Table 7). Because calcium ammonium nitrate is the most widely used nitrogen fertilizer in the Federal Republic of Germany it is pleasant that the emissions are low. In relation to calcium ammonium nitrate, emissions from urea application are high. In the Federal Republic of Germany they amount to 50 per cent of those from calcium ammonium nitrate application but the consumed quantity of urea is only 10 per cent of calcium ammonium nitrate consumption. At the moment, urea consumption is fortunately decreasing and therefore we hope that in the future $NH_3$ emissions from fertilizers will decrease.

Table 7: Emission levels of different fertilizers

| Fertilizer type | Consumption 1987/88 in 1000 tons nitrogen | Emission factor in % of the N content | $NH_3$ Emission 1987/88 in tons |
|---|---|---|---|
| Calcium ammonium nitrate | 1,100 | 2 | 22,000 |
| Ammonium sulfate | 4 | 15 | 600 |
| Ammonium nitrate | 2 | 10 | 200 |
| Urea | 110 | 10 | 11,000 |
| Calcium cyanimide | 10 | 10 | 1,000 |
| Ammonia | 50 | 10 | 500 |
| Compound fertilizer | 383 | 5 | 19,150 |
| Total | 1,614 | - | 54,450 |

Source: BEF (1989); based on BML (1988); Buijsman et al. (1985)

## 2.3 Non-agricultural emission sources

Böttger et al. (1978) separated non-agricultural emission sources into:

- natural ones from microbiological processes in soil
- internal combustion engines
- industry
- combustion processes.

Industrial and municipal waste, such as domestic cleaner, must be mentioned, as well as ammonia that is emitted by waste water. Wild animals also contribute to ammonia emissions. The amount of $NH_3$ released from the oceans is of significance, but is not further treated, especially as the territorial waters of the Federal Republic of Germany play only a minor role on the whole.

Emissions from internal combustion engines are important particularly in a highly motorized country. Otto engines release an average of 11 mg $NH_3$ per liter petrol. Petrol consumption of cars has decreased, but catalytic converters increase the emission of $NH_3$ (Pundt, 1987). Lies (1988) calculated that for cars with catalytic converters $NH_3$ emissions are 40 times higher than for cars without them. Diesel engines released 20 mg per liter (Oetting et al., 1984).

Industrial emissions occur from ammonia and nitrogen fertilizer production. Production of urea, sulfonamides, sodiumcyanide, sodium carbonate

(Solvay-process) and synthetic fibers lead to ammonia emissions. Ammonia is an indispensable component for nitric acid and nitrate production. Here ammonia storage enhances the $NH_3$ content in the air. These examples are accounted for with 20 kg per hectare. More information about emissions from combustion processes, especially if lignite is burned, were found by measurements in the German Democratic Republic.

In the waste air of carcass disposal plants ammonia is also one of the main harmful gases. The main contamination results from waste-dump percolating water. Ammonia emissions here may have less importance.

# 3 Magnitude of ammonia emissions from livestock farming

Only 5 per cent of ammonia emissions have natural causes and 95 per cent result from human activities. An average of 81 per cent of ammonia emissions in Europe result from livestock farming. Emissions in the Federal Republic of Germany exceed this average with 89 per cent (Fabry et al., 1990). These calculations are based on the assumption that 30 per cent of nitrogen content of excrements is released as $NH_3$ during storage and application (Böttger et al., 1978). Regionally, $NH_3$ emissions depend mostly on respective types of livestock farming.

Livestock farming is not evenly distributed by region and location (see Figure 1, based on Bundesamt für Ernährung und Forstwirtschaft quoted in: Pfeiffer and Werschnitzky, 1988). In the North and West of the Federal Republic of Germany, bigger stocks are more common than in other regions. Coastal regions and regions of high consumption in the Northwest of the Federal Republic of Germany have especially high stocks of pigs and poultry. Opposite to that, a high cattle stock is more common in traditional grassland regions. Especially remarkable is the concentration of poultry keeping in the administrative district of Vechta with 73 LU per hectare of agricultural area and the concentration of pig keeping in the administrative districts of Vechta, Kloppenburg and Coesfeld with more than 100 LU per hectare.

Animal specific ammonia emissions were calculated (Fabry et al., 1990) on the basis of emission factors and livestock at an administrative area level (emission factors are given in Tables 3 to 5).

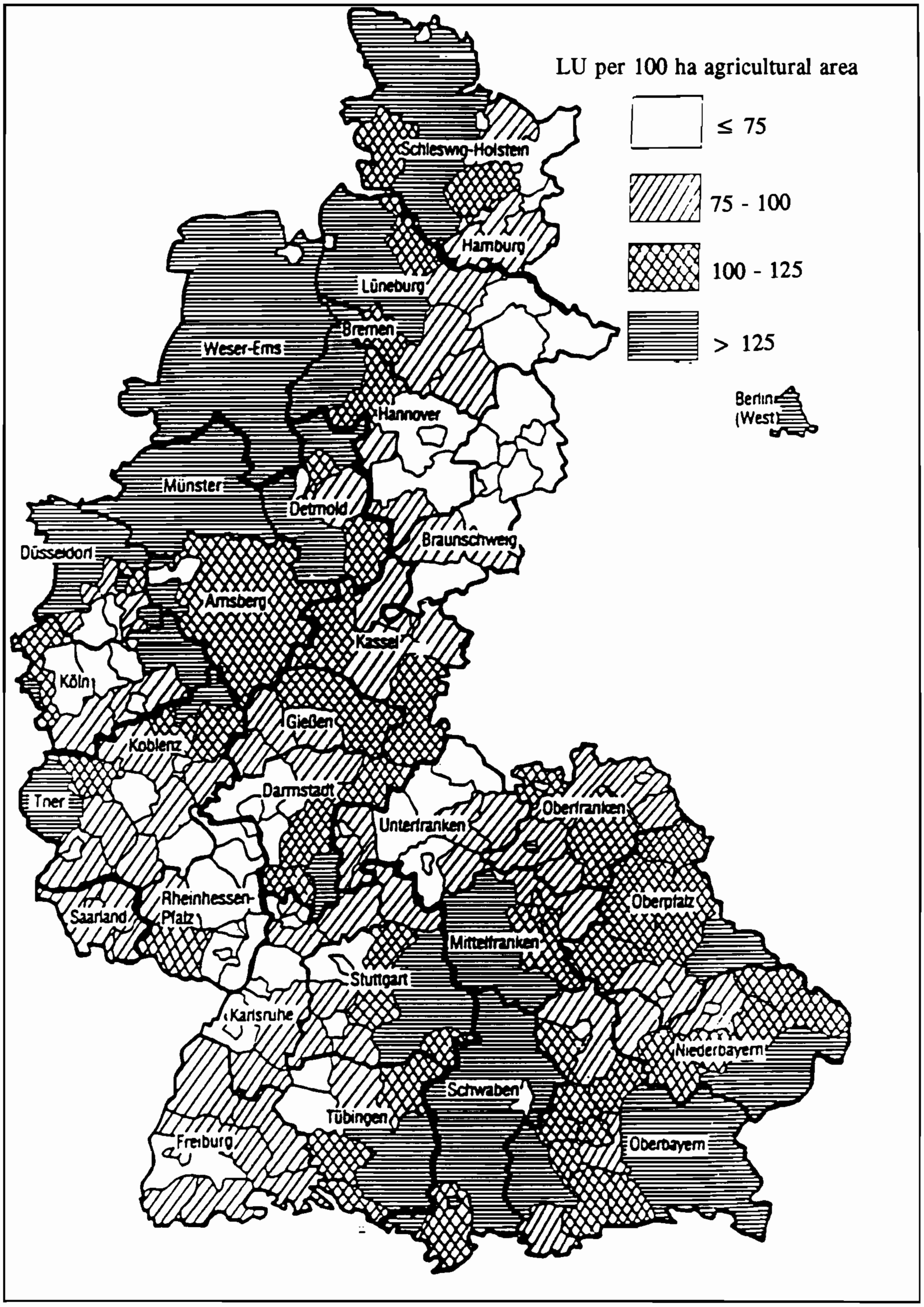



**Figure 1** Livestock numbers in administrative districts in large animal units (LU) per 100 ha agricultural area in 1987.

Table 8: Ammonia emissions of livestock farming in 1988.

| Region<br>Administrative district (RB) | Ammonia Emission in t | | | | | | Emission density | |
|---|---|---|---|---|---|---|---|---|
| | Horses | Cattle | Pigs | Sheep | Poultry | Total | kg/ha agricultural area | kg/ha total area |
| Emission factor (kg $NH_3$/LU/year | 10 | 22 | 19 | 32 | 25 | | | |
| **Schleswig-Holstein** | 331 | 21,960 | 3,229 | 588 | 342 | 26,420 | 24.4 | 16.8 |
| **Hamburg** | 29 | 174 | 13 | 8 | 6 | 230 | 15.5 | 3.0 |
| **Niedersachsen** | 769 | 47,293 | 16,509 | 612 | 3,524 | 68,707 | 25.1 | 14.5 |
| Braunschweig | 111 | 2,972 | 976 | 128 | 88 | 4,275 | 11.0 | 5.3 |
| Hannover | 151 | 6,117 | 3,032 | 159 | 374 | 9,833 | 18.9 | 10.9 |
| Lüneburg | 275 | 14,887 | 2,954 | 175 | 376 | 18,667 | 22.5 | 12.2 |
| Weser-Ems | 232 | 23,317 | 9,547 | 150 | 2,686 | 35,932 | 36.1 | 24.0 |
| **Bremen** | 10 | 253 | 9 | 1 | 3 | 276 | 27.5 | 6.8 |
| **Nordrhein-Westfalen** | 810 | 28,284 | 13,201 | 566 | 1,220 | 44,081 | 27.5 | 12.9 |
| Düsseldorf | 191 | 4,808 | 1,619 | 107 | 155 | 6,880 | 28.3 | 13.0 |
| Köln | 172 | 4,933 | 401 | 117 | 137 | 5,760 | 18.8 | 7.8 |
| Münster | 176 | 8,999 | 6,387 | 72 | 415 | 16,049 | 37.9 | 23.3 |
| Detmold | 117 | 5,163 | 3,423 | 122 | 384 | 9,209 | 25.4 | 14.1 |
| Arnsberg | 152 | 4,381 | 1,372 | 148 | 129 | 6,182 | 23.2 | 7.7 |
| **Hessen** | 316 | 11,384 | 2,287 | 428 | 339 | 14,754 | 19.0 | 7.0 |
| Darmstadt | 158 | 3,112 | 519 | 135 | 156 | 4,080 | 17.1 | 5.5 |
| Gießen | 72 | 3,230 | 509 | 136 | 83 | 4,030 | 20.3 | 7.5 |
| Kassel | 86 | 5,042 | 1,258 | 157 | 100 | 6,643 | 19.5 | 8.0 |
| **Rheinland-Pfalz** | 199 | 8,900 | 1,172 | 362 | 290 | 10,923 | 15.2 | 5.5 |
| Koblenz | 84 | 3,486 | 545 | 156 | 102 | 4,373 | 17.1 | 5.4 |
| Trier | 32 | 3,817 | 326 | 82 | 31 | 4,288 | 22.7 | 8.7 |
| Rheinl.-Pfalz | 83 | 1,597 | 301 | 124 | 157 | 2,262 | 8.2 | 3.3 |
| **Baden-Württemberg** | 515 | 25,604 | 4,612 | 741 | 601 | 32,073 | 21.3 | 9.0 |
| Stuttgart | 163 | 8,184 | 2,185 | 263 | 235 | 11,030 | 22.3 | 10.5 |
| Karlsruhe | 113 | 2,325 | 401 | 108 | 93 | 3,040 | 15.2 | 4.4 |
| Freiburg | 103 | 5,297 | 583 | 167 | 90 | 6,240 | 17.9 | 6.7 |
| Tübingen | 136 | 9,798 | 1,443 | 203 | 183 | 11,763 | 25.7 | 13.2 |
| **Bayern** | 625 | 77,534 | 8,506 | 1,070 | 1,266 | 89,001 | 25.8 | 12.6 |
| Oberbayern | 227 | 22,017 | 1,151 | 247 | 233 | 23,875 | 28.2 | 13.6 |
| Niederbayern | 81 | 11,275 | 2,301 | 133 | 391 | 14,181 | 24.7 | 13.7 |
| Oberpfalz | 52 | 9,269 | 708 | 79 | 310 | 10,418 | 24.6 | 10.8 |
| Oberfranken | 50 | 5,897 | 709 | 80 | 59 | 6,795 | 20.8 | 9.4 |
| Mittelfranken | 54 | 8,060 | 1,265 | 187 | 113 | 9,679 | 27.1 | 13.4 |
| Unterfranken | 58 | 3,967 | 1,136 | 181 | 72 | 5,414 | 15.4 | 6.4 |
| Schwaben | 103 | 17,048 | 1,235 | 163 | 88 | 18,637 | 33.1 | 18.7 |
| **Saarland** | 37 | 1,062 | 83 | 42 | 31 | 1,225 | 18.5 | 4.9 |
| **Berlin (West)** | 35 | 13 | 7 | 6 | 6 | 67 | 50.8 | 1.4 |
| Total | 3,676 | 222,461 | 49,627 | 4,425 | 7,627 | 287,816 | 24.1 | 11.6 |

Reference: BEF (1989)

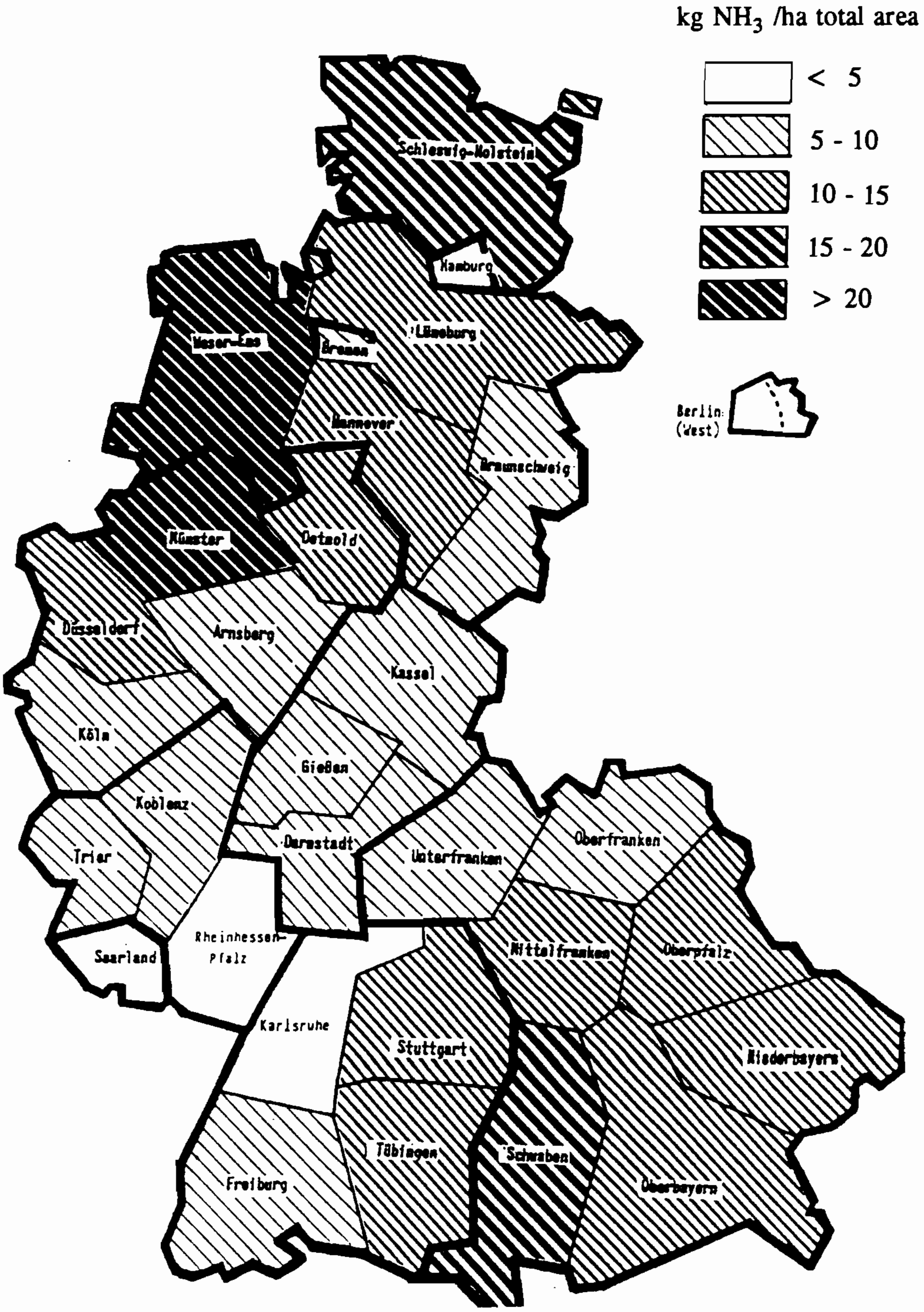
Emission factors used are average values calculated from Buijsman (1985) and Janssen (1985). As stated before, similar conditions in the Netherlands lead to a usage of values from this country. Cattle, which contribute most to ammonia emissions , were based on a value of 22 kg $NH_3$ per LU and year. This value was multiplied with the number of animals and related to the size of the area.

The harmful effects of $NH_3$ emissions have been based on the total area (Fabry et al., 1990). The calculated amount of about 290,000 tons $NH_3$ per year for the Federal Republic of Germany corresponds to reports of Isermann (1986). In the report of the experts council for the environment (SR-U, 1985) the estimated amount of ammonia emissions from livestock farming is slightly higher at 0.4 to 0.5 million tons per year.

The maximum emission densities of 30 kg $NH_3$ per hectare agricultural area were observed in the administrative districts of Münster, Weser-Ems and Schwaben (Figure 2). In Schwaben, ammonia emissions are a result of cattle keeping. In the other two regions (near the border to the Netherlands) swine raising is also a cause of emissions and in the region of Weser-Ems, poultry raising is of significance.

Based on total livestock farming, cattle keeping contributes 67 per cent to ammonia emissions. Swine and poultry keeping, assumed to be the main source, contribute only 17 and 3 per cent respectively (Fabry et al., 1990). These values are confirmed by Buijsman et al. (1985) and Vetter (1988).

The effects of ammonia emissions are also related to the area. In the Federal Republic of Germany, cattle keeping is land dependent and therefore emission density cannot be as high as from swine and poultry keeping, which are land independent (Fabry et al., 1990). Emissions have an enhanced effect close to the emitter. Therefore local emission levels are of importance. Table 9 shows ammonia emissions from livestock farming based on selected administrative districts. The administrative district of Vechta has by far the highest emission density. This occurs mainly from pig and poultry keeping. This is similar in the administrative districts of Northwest Germany where pig and poultry keeping contribute more to emission than in East Allgäu and Schwaben where cattle keeping is more dominant. In the administrative district of Münster, livestock farming and emissions are well spread over the entire area, but in other administrative districts livestock farming is more concentrated in certain rural districts.




Source: BEF (1989)

**Figure 2** Emission density per hectare in 1988.

Table 9: Ammonia emissions from livestock farming based on selected administrative districts (1988).

| Administrative District Rural District | LU Total per 100 ha LF[2] | Ammonia Emissions in t | | | | | | Emission density | |
|---|---|---|---|---|---|---|---|---|---|
| | | Horses | Cattle | Pigs | Sheep | Poultry | Total | kg/ha Agricultural Area | kg/ha Total Area |
| Emission Factor (kg $NH_3$/LU/Year | | 10 | 22 | 19 | 32 | 25 | | | |
| WESER-EMS | 171 | | | | | | | 36.1 | 24.0 |
| Vechta | 308 | 21 | 1,314 | 1,746 | 2 | 1,148 | 4,239 | 65.3 | 52.2 |
| Cloppenburg | 220 | 20 | 2,357 | 1,898 | 6 | 296 | 4,577 | 45.5 | 32.3 |
| Oldenburg | 147 | 20 | 1,494 | 793 | 11 | 177 | 2,495 | 36.0 | 23.5 |
| Graf. Bentheim | 191 | 12 | 1,649 | 586 | 4 | 307 | 2,558 | 40.9 | 26.1 |
| DÜSSELDORF | 137 | | | | | | | 28.2 | 13.0 |
| Kleve | 179 | 28 | 2,046 | 792 | 20 | 41 | 2,927 | 37.5 | 23.8 |
| MÜNSTER | 184 | | | | | | | 37.9 | 23.3 |
| Borken | 229 | 22 | 2,982 | 1,451 | 11 | 88 | 4,554 | 47.8 | 32.1 |
| Coesfeld | 185 | 32 | 1,251 | 1,365 | 11 | 104 | 2,763 | 37.4 | 24.9 |
| Steinfurt | 174 | 34 | 2,503 | 1,497 | 12 | 76 | 4,122 | 35.9 | 23.0 |
| DETMOLD | 123 | | | | | | | 25.4 | 14.1 |
| Gütersloh | 167 | 27 | 1,298 | 600 | 12 | 194 | 2,119 | 34.8 | 21.9 |
| TÜBINGEN | 120 | | | | | | | 25.7 | 13.2 |
| Ravensburg | 165 | 30 | 3,230 | 139 | 15 | 22 | 3,436 | 35.7 | 21.0 |
| SCHWABEN | 153 | | | | | | | 33.1 | 18.7 |
| Unterallgäu | 199 | 14 | 3,353 | 64 | 18 | 12 | 3,461 | 43.6 | 27.1 |
| Ostallgäu | 172 | 15 | 2,966 | 37 | 8 | 5 | 3,031 | 37.6 | 21.7 |

Source: BEF (1989)

Comparisons are difficult to make because the boundaries of the different counties in the Federal Republic of Germany are no longer related to landscape. For this reason a separation between arable plains and low mountain locations is not possible. To obtain better structured results, a calculation should be carried out on rural district levels, or a monitoring network should be built up across the entire Federal Republic of Germany. However, cost and the federal structure of the Federal Republic of Germany's administration make this difficult.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[2]LF = agricultural land

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# AN IMPROVED UK AMMONIA EMISSION INVENTORY

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## Abstract

There have been previous estimates of emissions of ammonia from the UK. However these estimates have not been entirely satisfactory. Some have been limited spatially to England and Wales. There have also been suggestions that the inventories do not include all sources of ammonia because the results of modelling studies using these inventories predict lower levels than those observed experimentally. This study examines possible emission sources for the entire United Kingdom and attempts to improve earlier studies. An assessment of the uncertainties of the results is made. The total UK emission of ammonia is estimated to be about 550 kTonnes compared with 300 - 450 kTonnes in earlier studies.

# 1 Introduction

There have been several earlier estimates of UK emissions of Ammonia. They are of two types. Either they concentrate on UK emissions or they are part of a wider European inventory. The work by Kruse et al. (1989) is an example of the first type. This gives details of animal emissions. Examples of the second are Asman(1990) and Buijsman et al. (1987). The existing inventories have been criticised as inadequate since modelling studies based on them underpredict the observed levels of ammonia deposition and its distribution in the UK (Metcalfe et al. 1989). However a recent report by Asman and van Jaarsveld (1990) gives reasonable agreement.

Table 1 summarizes estimated emissions from these studies. The large emissions from farm animals gives a very different geographic distribution of the emissions from other pollutants. Sulphur dioxide and nitrogen oxides are emitted mainly from fuel combustion and the geographic distribution of their emissions is thus roughly related to population.

Table 1. Emission estimates of ammonia

| | Asman (1990) | | | Kruse (1989) | |
|---|---|---|---|---|---|
| | Europe | UK | | England + Wales | |
| | kT | kT | % | kT | % |
| Cattle | 3,838 | 258 | 57 | 182 | 60 |
| Pigs | 872 | 31 | 7 | 19 | 6 |
| Poultry | 469 | 34 | 8 | 25 | 8 |
| Horses | 69 | 2 | | 2 | |
| Sheep | 358 | 41 | 9 | 62 | 21 |
| Fertilizer application | 1,339 | 84 | 19 | 12 | 4 |
| Industry | 11 | 1 | | - | |
| Total | 6,956 | 451 | | 302 | |

Neither of the two studies described above has paid much attention to other, admittedly small, sources. This study aims to include all likely sources of ammonia and to describe the likely confidence limits of the resultant estimate. For each emission source upper and lower estimates of the emission have been made together with a central estimate. Thus the likely spread of emissions can be estimated. However it is extremely unlikely that the true emissions will all be the lower or upper estimate. Thus the best estimates should reflect the true emissions better. This is discussed in greater depth below.

The results do not produce any great surprises. However the inclusion of all emissions and the errors associated with them do show that ammonia emissions from the UK could be substantially larger than earlier thought.

# 2 Emission sources

The largest source is cattle. Other farm animals are also large sources. The factors and methodology presented here follow those of Kruse and ApSimon (1989) and Erisman (1989). Emission factors and calculation methods have been developed for other sources. The details are given below.

## 2.1 Animals

### 2.1.1 Cattle

The ammonia emissions from cattle come from both urine and the manure. The method of estimating these emissions has been explained in ApSimon et al. (1987), Asman (1990) and Erisman (1989). In the UK cattle spend about half of their time outside in pastures and the remainder under cover. If the cattle are not outside the manure has to be collected, stored and disposed of (usually by spreading on land). Thus there are three emission factors to be considered: From pasture, from storage and from manure spreading. Table 2 shows the factors used in this study and compares them with other factors from the literature. The different factors used give a range and thus an upper and lower estimate can also be derived. Experimental measurements (for example Lockyer et al.; 1989) show a wide range of results which appear to depend on a variety of factors including temperature, time of year, humidity and details of animal husbandry. Until this is better understood this area of the inventory will have large uncertainties.

### 2.1.2 Sheep

Emissions from sheep are estimated in a similar way to those from cattle. In the UK sheep are kept outside for almost all the year and so only emissions from this source need be considered. Table 2 shows the emission factors used.

### 2.1.3 Other farm animals

In the UK pigs are mainly kept indoors. However some do spend time outside. The approach is again similar to cattle and Table 2 shows the factors used. Poultry, ducks, geese, turkeys, horses and goats are all considered. The factors used are shown in Tables 2 and 3. As most of the emissions come from urine which, when deposited on surfaces, is broken down with the help of a natural enzyme, the factors for humans are lower than for other animals. The factors used come from Kruse and ApSimon (1989) and Erisman (1989) with additional information from Jarvis (1990) and Lockyer (1989).

Table 2 Emission factors used for estimating emissions from farm animals

Comparison of some emission factors expressed as Equivalent Emission Factors kg $NH_3$/year (assuming animal spends all year in pasture or inside)

| Data used in this study (after Kruse,1989) | | Kruse (1989) | Buijsman (1987) | Erisman (1989) |
|---|---|---|---|---|
| **Cattle** | | | | |
| kg N produced per year | 45.5 | | | |
| % year outside | 50 | | | |
| % N lost as $NH_3$ | | | | |
| pasture | 35 | 15.9 | 7.2 | 12.3 |
| storage | 20 | 9.1 | 4.9 | 14.4 |
| spreading | 30 | 13.65 | 6.3 | 14.4 |
| **Sheep** | | | | |
| kg N produced per year | 10 | | | |
| % year outside | 100 | | | |
| % N lost as $NH_3$ from pasture | 21 | 2.1 | 3.1 | 2.1 |
| **Pigs** | | | | |
| kg N produced per year | 5.5 | | | |
| % N lost as $NH_3$ from | | | | |
| storage | 20 | 1.1 | 1.5 | 2.0 |
| spreading | 56 | 3.1 | 1.3 | 4.1 |
| **Poultry** | | | | |
| kg N produced per year | 10 | | | |
| % N lost as $NH_3$ from | | | | |
| storage | 10 | 1.0 | 0.11 | 0.18 |
| spreading | 13 | 1.3 | 0.15 | 0.19 |

However the data needed to estimate emissions from these animals is not available in a consistent form. Some years the UK Ministry of Agriculture, Fisheries and Food (MAFF) have combined counts of differing groups of these animals. This leads to greater uncertainty in the emission estimate.

Table 3 Emission factors used for estimating emissions from other animals.

| | kg $NH_3$/animal/year |
|---|---|
| Horses | 9.4 |
| Ducks | 0.12 |
| Turkeys | 0.86 |
| Goats | 6.4 |
| Humans | 0.29 |

### 2.1.4 Other animals

Emissions from pets have been estimated following Erisman (1989).Cats and dogs are estimated separately. There is great uncertainty in the total number of these animals as there are no central statistics. However, on the basis of numbers of animals used in this study, they do not amount for a large percentage of the total emissions.

An estimate of emissions from wild animals has also been made. Due to the large uncertainties in the numbers of wild animals and their emissions this is a rough estimate derived from the data in Erisman (1989). However it does show that this amounts to 18 kT and is a minor contribution to the total emission.

### 2.1.5 Humans

Humans are another source and Table 3 shows the factor used. This was derived from Erisman (1989). It does not include sewage which is accounted for below.

## 2.2 Fertilizer Use

The use of fertilizers is a potentially large source. Kruse et al. (1989) estimate 12 kTonnes for England & Wales while Asman (1990) gives 84 kTonnes for the United Kingdom. The differences come from the emission factors used. Recent reports (Jarvis & Pain (1990) and Smith & Arah (1990), favour higher emission rates. Buijsman et al (1987) and Erisman (1989) use the factors given in Table 4. These have been used here. There is considerable uncertainty about the exact nature of the fertilizers used in the UK while the total amounts are known. Here it was assumed that the fertilizer used is mainly ammonium sulphate, ammonium nitrate and other compounds. The amount of urea used increased significantly through the period 1980-1988. The decision to allocate a large fraction of known fertilizer use to the "other" group gives it an emission factor at the low end of the possible range. This is probably the area of the inventory that needs most work to quantify more precisely the changes in time in fertilizer use. The time and method of application of fertilizer has a large effect on the emission of ammonia and this should also be examined in more detail for the UK situation.

Table 4 Emission factors used for estimating emissions from fertilizer use.

| | % N lost as $NH_3$ |
|---|---|
| Ammonium sulphate | 15 |
| Ammonium nitrate | 10 |
| Urea | 10 |
| Other (incl. combined) | 1 |

## 2.3 Industrial Processes

Several industrial processes may emit ammonia. Those included here are the production of ammonia and nitric acid and the manufacture of fertilizers. The factors are given in Table 5. The quantities produced are not known precisely. For some sectors the data is kept confidential as there are only a few producers and the publication of the data may give information to competitors. Here it has been necessary to estimate some outputs from the capacity of the plants.

## 2.4 Sewage

The spreading of sewage sludge is another source of ammonia. It is estimated that about 15 million tonnes of sludge are produced each year. Of this about 50% is spread on agricultural land each year. The emission rate is dependent on the method of application but the single simple factor used indicates that this is a very small source.

Table 5 Emission factors used for estimating emissions from industrial processes.

| | kg $NH_3$ /T produced |
|---|---|
| Ammonia | 0.8 |
| Nitric acid | 0.01 |
| NPK fertilizer | 12.5 |
| Ammonium nitrate | 5.0 |
| Ammonium sulphate | 5.0 |
| Ammonium phosphates | 5.0 |

## 2.5 Landfill

There are small emissions of ammonia from landfill along with much larger amounts of methane and carbon dioxide. It has been estimated that the ratio of nitrogen emissions to methane is about 7.3%. Of this about 10% is ammonia. In this study this ratio has been used together with estimates of UK national emissions of methane (Munday, 1990).

## 2.6 Road Transport

Road vehicles are another small source. Diesels emit small amounts. Here a factor of 3 mg/km has been used. Petrol engined cars emit negligible amounts. However the use of three-way catalysts will increase the emissions from this sector. A factor of about 60 mg/km appears to be reasonable. Currently, in the UK, there are few cars fitted with three-way catalysts but their number is expected to grow rapidly in the future. The current estimate of emissions from road transport is 0.2 kT. If all cars had three-way catalysts the total would be 17 kT (0.04% and 3% of UK total emissions respectively).

## 2.7 Other

An estimate of emissions from natural soils based on Erisman (1989) has been included for completeness. The total is small and together with the estimate from wild animals indicates the predominance of anthropogenic emissions.

# 3 Results

Table 6 gives results for the years 1980 to 1988. These figures are central estimates. The accuracy and uncertainty of the results will be discussed below. There is little overall trend. In 1988 43% of the emissions came from cattle, 16% from sheep and 22 % from fertilizer use. In total agriculture accounts for 87% of total UK emissions. While emissions from cattle have declined by 12% those from sheep have increased by 30%.

Table 6 UK emissions of ammonia (kTonnes)

| | 1980 | 1981 | 1982 | 1983 | 1984 | 1985 | 1986 | 1987 | 1988 | 1988 as % |
|---|---|---|---|---|---|---|---|---|---|---|
| Cattle | 260 | 254 | 256 | 257 | 256 | 250 | 242 | 235 | 230 | 43 |
| Sheep | 66 | 67 | 69 | 71 | 73 | 74 | 77 | 81 | 86 | 16 |
| Other farm animals | 33 | 33 | 34 | 34 | 32 | 33 | 33 | 33 | 34 | 6 |
| Other animals | 19 | 19 | 19 | 19 | 19 | 19 | 19 | 19 | 19 | 4 |
| Natural | 12 | 12 | 12 | 12 | 12 | 12 | 12 | 12 | 12 | 2 |
| Fertilizer use | 107 | 102 | 113 | 128 | 124 | 131 | 126 | 138 | 116 | 22 |
| Sewage | 4 | 4 | 4 | 4 | 4 | 4 | 4 | 4 | 4 | 1 |
| Humans | 17 | 17 | 17 | 17 | 17 | 17 | 17 | 17 | 17 | 3 |
| Process | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 1 |
| Landfill | 4 | 4 | 4 | 5 | 5 | 5 | 5 | 5 | 5 | 1 |
| Total | 536 | 527 | 542 | 561 | 558 | 560 | 551 | 560 | 538 | |

Figure 1 plots the emissions for each year. There is little difference in total emissions between years.

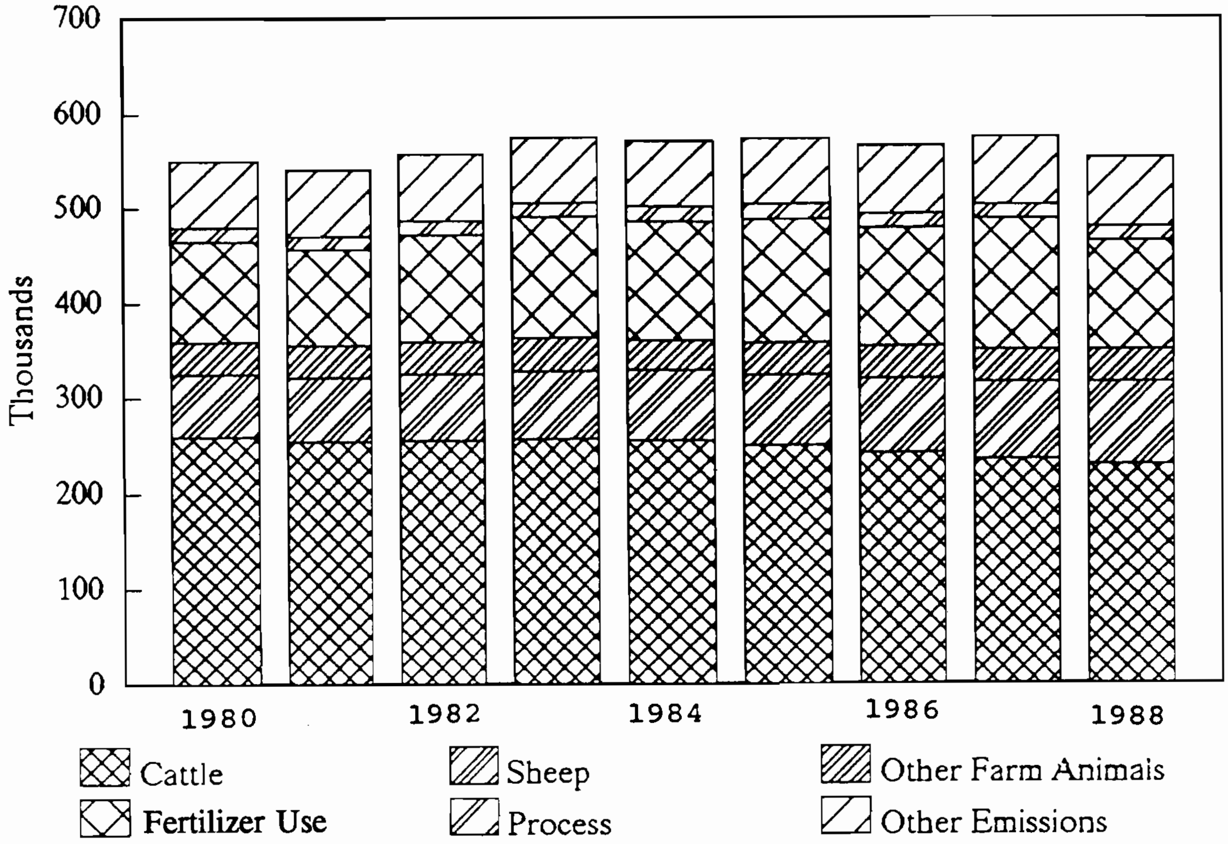


Figure 1 UK emissions of ammonia

# 4 Comparison with other estimates

Comparing Tables 1 and 6 shows that these estimates are larger than earlier estimates. This is due to two factors. Firstly the emissions from fertilizer use are larger. This is due to higher emission factors in some cases and to higher estimates of urea use in others. Secondly there are a number of small sources included which, while individually small, increase the total significantly.

When the accuracy of the estimates is considered the different studies seem broadly comparable. Due to the large uncertainties in all these figures (see below) there is no clear difference between them.

# 5 Accuracy

To assess the uncertainties of the final estimate a Monte-Carlo approach has been used. This approach is explained in more detail in Eggleston (1990). For each input variable (emission factor or surrogate statistic) an assessment of the likely range of possible values was made. The literature gives a range of values from different measurements and these form the basis for this information. Then an estimate of the total emission was made with data allowed to randomly take a value in the range centered on the 'central' values discussed above. This was repeated 4000 times and the resulting 4000 totals give a frequency distribution of the possible totals for the emission inventory.

Figure 2 shows a few of these distributions. It shows the number of times each total emission was predicted. Thus it shows the probability of the true emission having a particular value. The curves for different years overlap and therefore the emissions for the different years are indistinguishable.

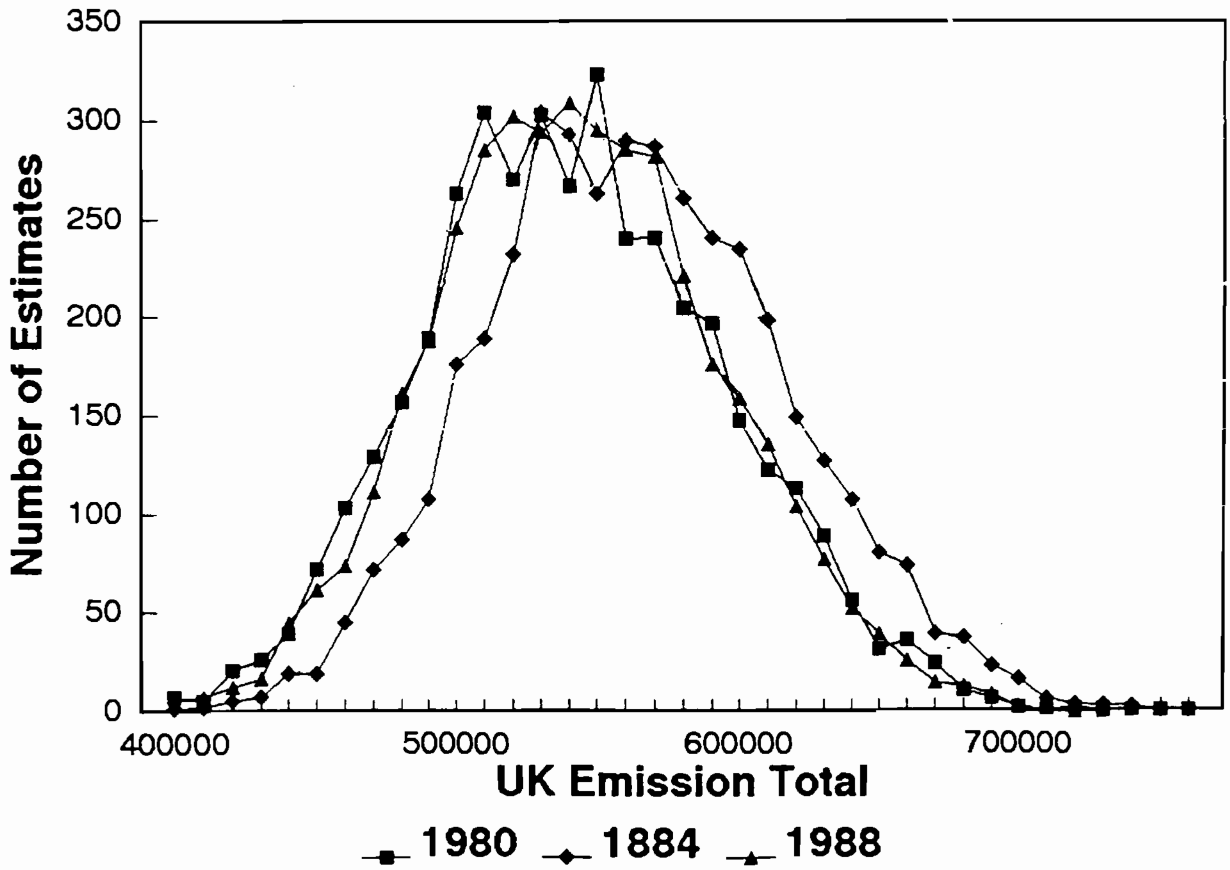


Figure 2 Frequency plot of $NH_3$ emission estimates

Table 7 shows the results. The standard deviation is an indication of the width of the distribution.

Table 7 Uncertainty of estimates of emissions of ammonia

| | 1980 | 1981 | 1982 | 1983 | 1984 | 1985 | 1986 | 1987 | 1988 |
|---|---|---|---|---|---|---|---|---|---|
| Total (kT) | 536 | 527 | 542 | 561 | 558 | 560 | 551 | 560 | 538 |
| Standard deviation | 52 | 52 | 52 | 53 | 53 | 52 | 52 | 51 | 50 |
| Minimum | 375 | 364 | 375 | 398 | 386 | 394 | 384 | 395 | 378 |
| Maximum | 721 | 704 | 721 | 752 | 739 | 747 | 727 | 737 | 702 |

Figure 3 shows these values plotted for each year. It can be seen that when uncertainty is taken into account that there is no clear trend in emissions.

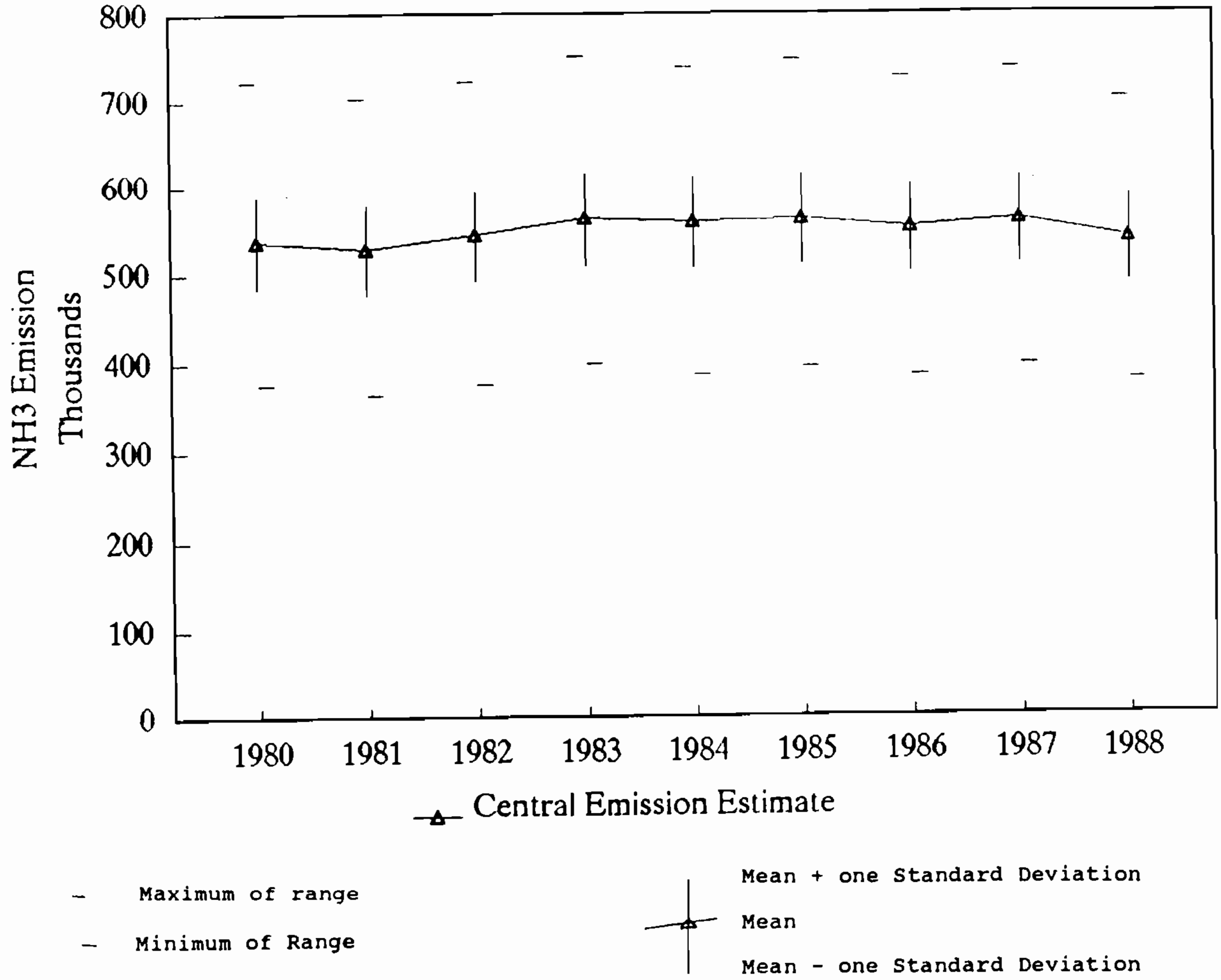


Figure 3 Ammonia emissions

# 6 Conclusions

Emissions of ammonia from the UK are about 500-550 kT per year, with a central estimate of 538 kT in 1988.

The estimates have a range of 340-350 kT with a standard deviation of 50 kT. This is so wide that year to year variations in Table 7 are not significant.

More work needs to be done to refine estimates of emissions from fertilizer use. Better data is needed particularly for types of fertilizer used and manufacture of fertilizers.

It seems likely that earlier estimates of ammonia from the UK underestimated the true figure. However given the uncertainties is the current understanding of these emissions it is not possible to be absolutely certain.

## Acknowledgement

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# A NEW APPROACH TO ESTIMATE AMMONIA EMISSIONS IN SWEDEN

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## Abstract

The $NH_3$ emission from storage and application of animal manure has been parameterized in the expression: $(C_{eq}-C_a)\cdot K_Z$. All three terms can be measured in the field and an example is given in the text. $C_{eq}$ is the $NH_3$ equilibrium concentration in the air at the air/soil interface. $C_{eq}$ depends on the manure, slurry or urine used, the temperature and if it has been applied to a soil. It also depends on soil type, humidity, spreading technique and time. $C_a$ is the ambient concentration and $K_Z$ the mass transfer coefficient. $C_a/C_{eq}$ and $K_Z$ depend on wind speed and wind stability. The $NH_3$ emission can be estimated by measuring $C_{eq}$ as a function of the parameters mentioned and by using a range of values for $C_a/C_{eq}$ and $K_Z$. By parameterizing the expression for $NH_3$ emission, climate variations and agricultural practice can be taken into account in an $NH_3$ emission inventory.

# 1 Introduction

A literature review of papers dealing with studies of interaction and transport mechanisms for $NH_3$ in soil and air has previously been published (Ferm, 1983). $NH_3$ can rapidly be transported in the soil as $NH_4^+$ ions with the soil solution. In the air it can fairly rapidly be transported by turbulent diffusion. The transport in the air at the air/soil interface is, however, slow and is the rate-determining mechanism. This interface is often referred to as the laminar boundary layer. It is of the order of a few millimeters thick. There $NH_3$ is transported by molecular diffusion. The thickness of this layer decreases with increasing wind speed. The $NH_3$ emission will therefore increase with increasing wind speed and consequently cannot be measured with equipment that affects the wind. The difference between the gaseous $NH_3$ concentrations at the soil surface (Ceq) and the air above the soil ($C_a$) is the driving force for the transport (emission or deposition). $C_{eq}$ is determined by the equilibrium between $NH_4^+$, $NH_3(g)$ and $H_3O^+$. The transport rate (emission or deposition), $\varnothing_{NH3}$ can be calculated from the driving concentration difference and the mass transfer coefficient ($K_Z$).

$$\varnothing_{NH3}=(C_{eq}-C_a)\cdot K_Z \tag{1}$$

If the concentration in the air ($C_a$) just above the laminar boundary layer is used, $K_Z$ can be called the surface transfer coefficient. The expression can be used both for the $NH_3$ loss during storage and spreading but probably not for the emission from livestock buildings. It is believed that the losses from storage and spreading make up more than half of the total losses, if no $NH_3$ emission control is used. These losses are very dependent on climate and agricultural practices and should therefore be different in Sweden than, for instance, in central Europe. An inventory for a country can be performed by measuring a series of $C_{eq}$ values in the laboratory. $C_{eq}$ is then measured using manure or slurry having different $NH_4^+$ contents and pH at different temperatures and times after application etc. $C_a$ is a function of $C_{eq}$ and the height above the soil. The ratio $C_a/C_{eq}$ depends on the stability of the air. $K_Z$ depends on the wind speed and the reference height (which should be the same as for $C_a$). $C_a/C_{eq}$ and $K_Z$ can be determined in field experiments or perhaps be taken from micro-meteorological formulas combined with wind statistics.

By integrating many combinations of $C_{eq}$, $C_a/C_{eq}$ and $K_Z$ using equation 1, the total $NH_3$ emission from storage and spreading can be estimated for a country. For this purpose a large number of measurements must be made, requiring that simple and inexpensive measurement techniques are available. This paper discusses a relatively simple and inexpensive measurement technique which might be simplified even further by measuring $K_Z$ directly instead of obtaining it from measurements of $\varnothing_{NH3}/(C_{eq}-C_a)$.

# 2 Experimental design

## 2.1 Manure spreading

Liquid cattle manure was spread with a low trajectory surface spreader of 3.0 m spreading width, equipped with splash plates at the outlet of each hose, equidistant at 250 mm. In this way a uniformly broadcast experimental area of 12.3 m by 14.5 m was achieved. The pH was 7.5, the dry matter content 4.0%, the $NH_4^+$-N content 0.13 % and the organic-N content 0.08 %. The manure was spread at a rate of 45 tonnes per hectare.

## 2.2 Manure storing

The manure was stored in two 29 m diameter concrete slurry tanks. The tanks were uncovered and situated approximately 200 m from the experimental field. Passive flux measurements were carried out in one of these storage tanks as well.

## 2.3 Soil and weather conditions

The field surface was barley stubble on water-saturated clay soil. Soil temperature during the 24-hour experimental period was on the average +7.6°C. The wind speed varied between 0 and 5 m/s and was on the average 1.9 m/s. There was no precipitation during the period. Air temperature varied between 0°C and +14°C and was on the average 6.8°C. Variations in meteorological conditions were measured by means of a Vicon WS-8 recorder and stored in a computer. Figure 1 gives the temperature of the air, ground surface, and soil together with variations in wind velocity.

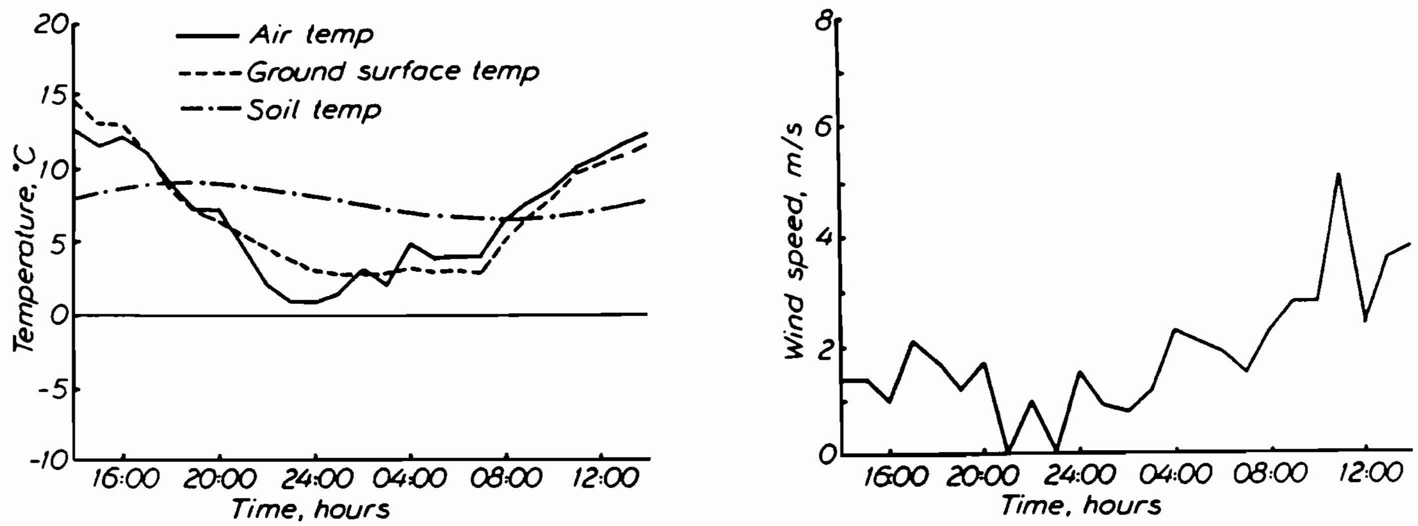


**Figure 1** Data obtained by a Vicon WS-8 weather recorder at the experimental field showing temperature and wind variations during the first 24 hours following slurry application.

## 2.4 The closed dynamic chamber technique for estimating the equilibrium concentration

The closed dynamic chamber technique, introduced by Watkins et al. (1972) was used. According to Marshall and DeBell (1980), ammonia losses estimated by this technique are more representative of ammonia losses under field conditions, than other variants of the chamber technique. As will be shown below, the results obtained with this technique have been interpreted in a different way.

A steel frame with an open area of 0.275 $m^2$ was placed on the experimental area after spreading and a chamber was placed on the frame. See Figure 2. Air was sucked through the chamber by means of a channel fan. The air in the chamber was mixed thoroughly by means of a stirring fan. Air from the chamber was extracted continuously for ammonia analysis by a TGM-555 air monitor from CEA Instruments Inc.

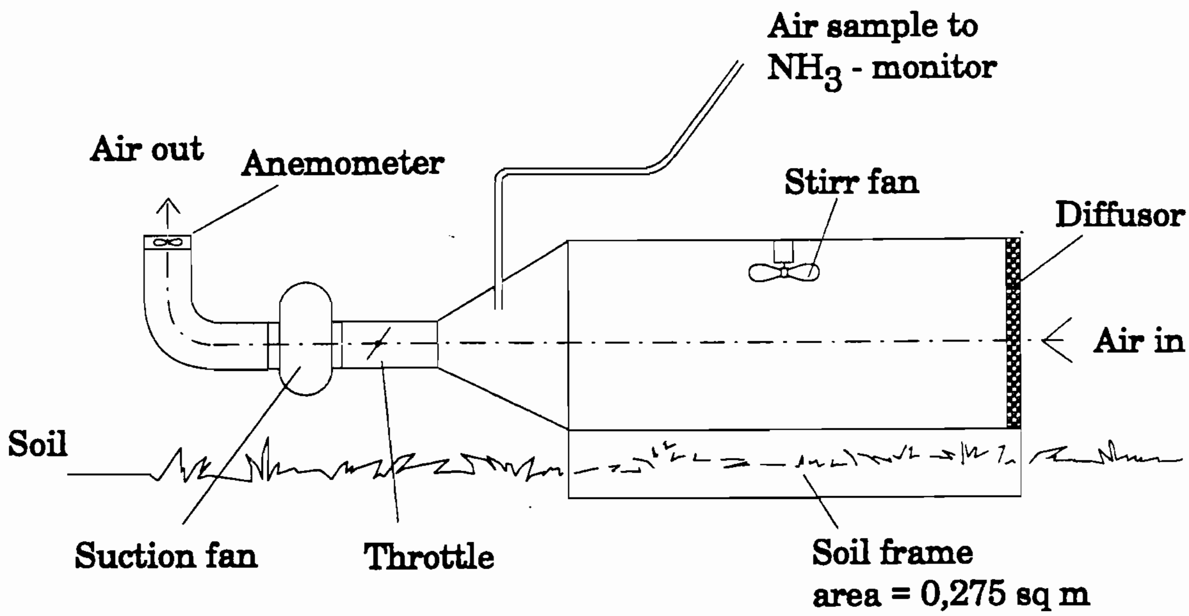


**Figure 2** Chamber assembly for ammonia analysis in air according to the closed dynamic method.

The analytical method is based on a colorimetric procedure in which an emerald green color is produced by the reaction of sodium salicylate, sodium nitroprusside and sodium hypochlorite with ammonia which is absorbed in a dilute sulfuric acid and then buffered in an alkaline solution. The intensity of the color formed is measured spectrophotometrically at a shoulder at 550 nm and is directly proportional to the concentration of ammonia absorbed. The instrument was calibrated for operation in the range of 0 - 25 ppm. Ammonia concentration was continuously recorded by means of a linear recorder. The air flow rate through the chamber was chosen so as to obtain a ratio between the ventilation flow F and the covered area A of 19 mm/s.

The $NH_3$ emission from the chamber, i.e. the $NH_3$ concentration in the outgoing air multiplied by its flow and divided by the covered soil area, is different from an uncovered area of soil. The $NH_3$ emission from the chamber depends on the ventilation flow and the stirring velocity of the fan. If the stirring fan is efficient the mass transfer coefficient ($K_Z$ in eq. 1) inside the chamber will be high (Ibusuki and Aneja, 1984). The ammonia flux from the chamber ($\varnothing_{NH3,\ ch}$) is calculated from the ammonia concentration in the outlet ($C_{ch}$) and the inlet air ($C_a$), the ventilation flow (F) and the covered area (A).

$$\varnothing_{NH3,ch} = (C_{ch} - C_a) \cdot F/A \tag{2}$$

By combining equation 1 applied to the chamber with equation 2 we obtain:

$$\varnothing_{NH3,ch} = (C_{eq} - C_{ch}) \cdot K_{ch} = (C_{ch} - C_a) \cdot F/A \tag{3}$$

or

$$C_{ch} = \frac{(C_{eq} \cdot K_{ch} + C_a \cdot F/A)}{(K_{ch} + F/A)} \tag{4}$$

Equation 4 shows that the concentration in the outgoing air ($C_{ch}$) approaches $C_{eq}$ when the laminar layer is thin due to efficient stirring ($K_{ch}$ is high) and the ventilation flow (F) is low.

### 2.4.1 A simple chamber for estimating the equilibrium concentration

A very simple chamber consisting of a 40 x 20 cm polyethylene container with a height of 20 cm was equipped with a battery-operated fan. The battery was small and could run for several days without being recharged. The $NH_3$ concentration was measured with diffusional samplers attached to the inside by double-sided self-adhesive foam pads just before the container was placed on the soil and the fan was started.

## 2.5 Ammonia emission from flux measurements

A simple passive sampler that can measure the horizontal ammonia flux, i.e. the product of the ammonia concentration, the wind speed and cosine for the angle between the wind direction and the axes of the sampler, has been developed (Ferm et al., 1990). By measuring the horizontal fluxes that have passed the field at four different heights and at four different points enclosing the field, and subtracting the fluxes that enter the field from the surroundings at the same 16 points, the vertical $NH_3$ emission from the field can be calculated from the mass balance. The horizontal ammonia flux is higher near the ground and decreases approximately exponentially with increasing height. Equal distances between the samplers have been used so far (see also Ferm and Christensen, 1987). In this study, the height intervals increased exponentially. The horizontal flux ($\mu$g N/m$^2$) at each point was calculated from:

$$\varnothing = \frac{(X_1+X_2)}{2\cdot 0.7\cdot\pi\cdot(0.5\cdot 10^{-3})^2} \tag{5}$$

$X_1$ and $X_2$ are the amounts of $NH_3$ ($\mu$g N) in the two tubes facing the same direction. $0.5\cdot10^{-3}$ is the radius of the hole (m). 0.7 is an empirically determined correction term for the air turbulence behind the hole.
The emission (vertical ammonia flux) $\varnothing NH_3$ was calculated from:

$$\varnothing_{NH3}\cdot A\cdot t = \sum_{m=1}^{m=4}\sum_{n=1}^{n=4} W\cdot\Delta H_n(\varnothing_{F,n}-\varnothing_{s,n}) \tag{6}$$

A is the area between the masts, t the sampling time, W the width of the field, ΔH the height interval that the sampler represents. Index m denotes the masts, n the heights, F the direction from the field and S from the surroundings. The choice of heights and height intervals is shown in Figure 3.

If the field has a rectangular shape and the masts are centered on its sides, the measured value will to some extent be affected by the angle between the wind direction and the field. A circular field is ideal because the estimated emission will be independent of the wind direction (Ferm et al., 1990). Figure 4 shows the distance that the air has passed over a rectangular plot before it reaches the sampler and the wind component that is proportional to the flow that enters the passive flux samplers.

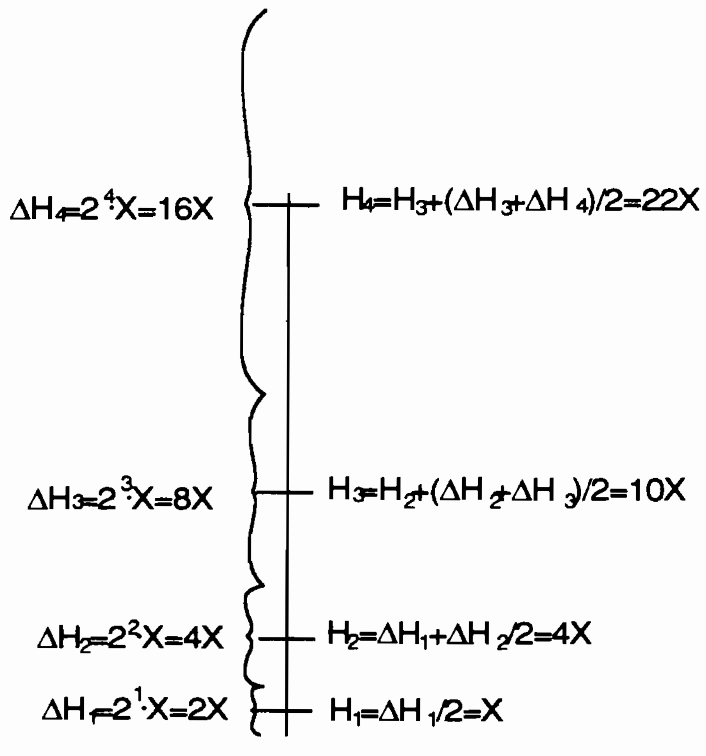


**Figure 3** Heights above ground and height intervals used during the sampling.

The formula for the wind component is only valid for wind directions between arctan a/2b and arctan b/2a, where a and b are the dimensions of the rectangle. If a large number of masts and measuring points are used the measured emission will be correct. If only four masts are used and placed in the middle of the sides, they will over-represent the flux at certain wind directions. This is schematically shown in Figure 5, where the over-estimation of the emission as a function of wind direction is shown. The average over-estimation is 1.11.

## 2.6 Diffusional samplers for measuring $NH_3$ concentrations and deposition velocities

This passive sampler, which is now referred to as a diffusional sampler, was first used by Palmes and Gunnison (1976) to measure $NO_2$ in indoor air. The gas to be measured is trapped on an impregnated filter that acts as a perfect sink. The concentration gradient in front of the filter is the driving force. This gradient is made constant by placing the filter in a tube. The gas should only be transported by molecular diffusion inside the tube. The trapped amount of gas

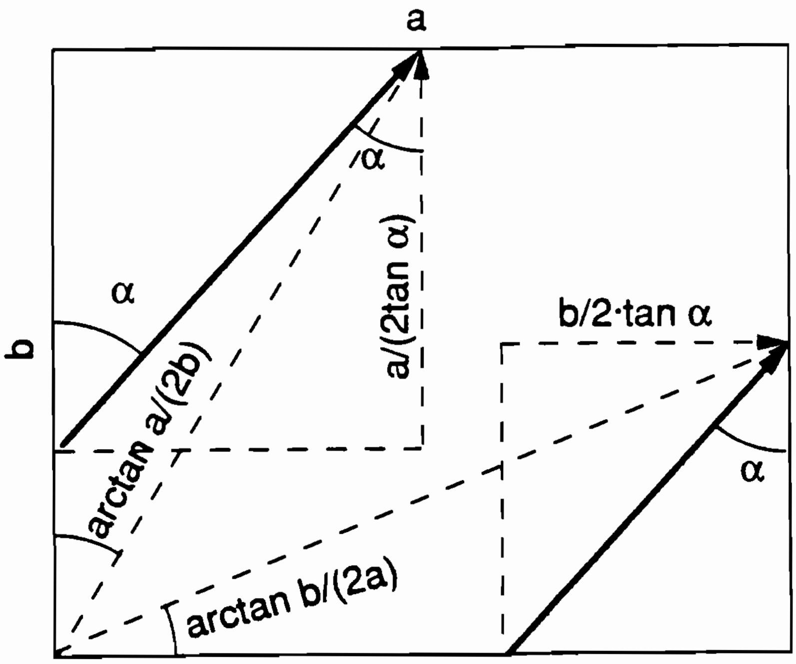


**Figure 4** Wind trajectories passing over a rectangular field and wind components toward the samplers when placed in the middle of each side.

is analyzed on the filter. The amount is proportional to the sampling time, the dimensions of the tube and the concentration of the gas in the ambient air. Since the tube is open at one end a turbulent flow will occur if it is exposed outdoors. This turbulence can be avoided by mounting a net or a membrane filter in front of the open end. The most sensitive version of the sampler consists of a polypropylene ring. The filter (Whatman 40) which acts as a perfect sink for $NH_3$, is impregnated with 1 % oxalic acid in methanol. The filter is attached by putting on a snap-on cap. A Teflon filter (Fluoropore® FALP Ø 25 mm) is placed under the lower opening. The Teflon filter is attached by another snap-on cap. A 20 mm centered hole is punched out in this cap. During storage and transport the sampler is kept in a polypropylene vial with a similar but wider snap-on cap see Figure 6.

The concentration is derived from Fick's law Ø = -D·dC/dL. Where Ø is the flux of the gas (the amount that is net transported per unit time and area), D is the diffusion coefficient for $NH_3$ in air which is $2.54 \cdot 10^{-5}$ $m^2s^{-1}$ at 25°C (Coulson and Richardson, 1954). The temperature dependence of D can be set to $T^{1.5}$ where T is the absolute temperature (Gilliland, 1934). dC/dL is the concentration gradient which is negative in the direction of the flow. The flux

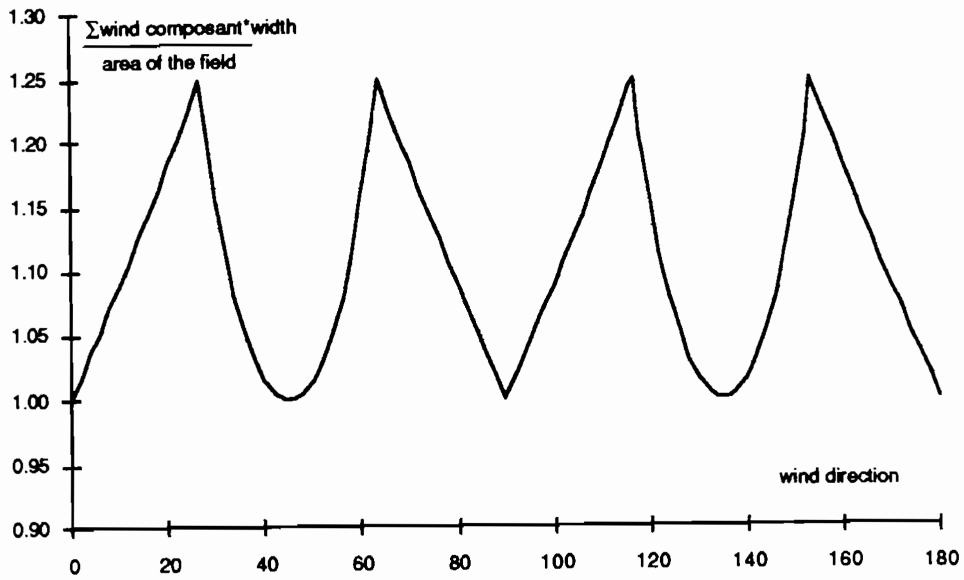


**Figure 5** Over-estimation of the emission as a function of wind direction. The calculation represents a quadratic field with four centered masts on its sides.

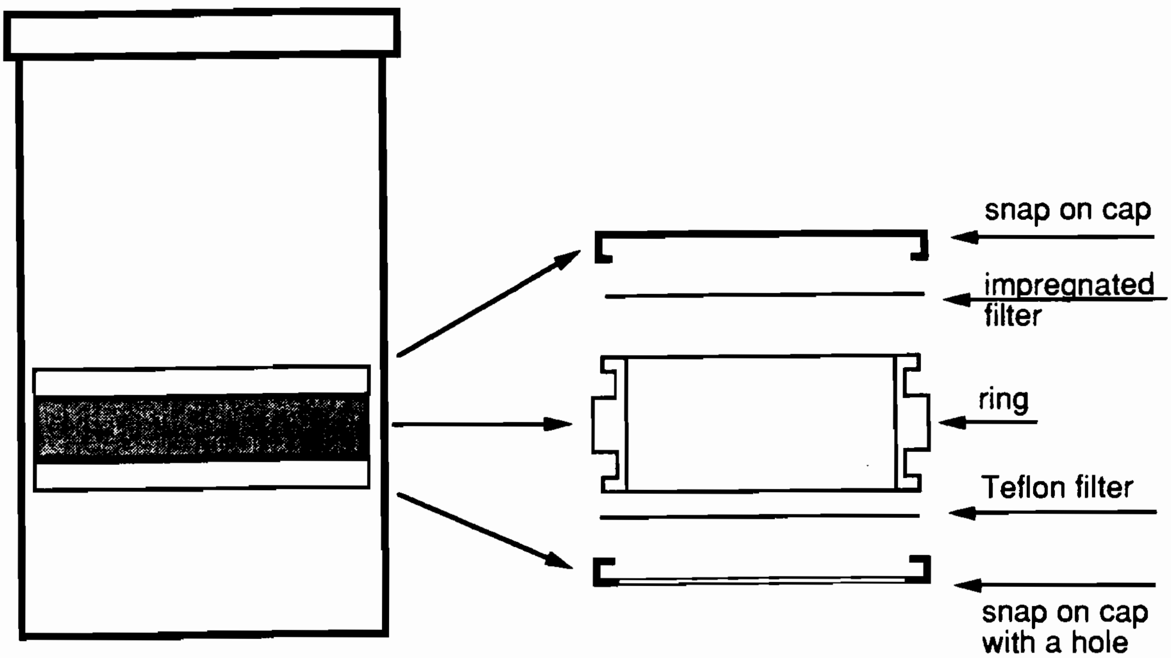


**Figure 6** Storage container with diffusional sampler.

is calculated from X/(A·t) where X is the amount trapped on the filter. A is the cross section area and t the exposure time. The concentration difference over a section of the sampler is then calculated from:

$$\Delta C = -\frac{X \cdot \Delta L}{t \cdot D \cdot A} \tag{7}$$

Gases are transported by molecular diffusion in the laminar boundary layer that surrounds all objects. The transport distance is therefore somewhat longer than the length of the tube.

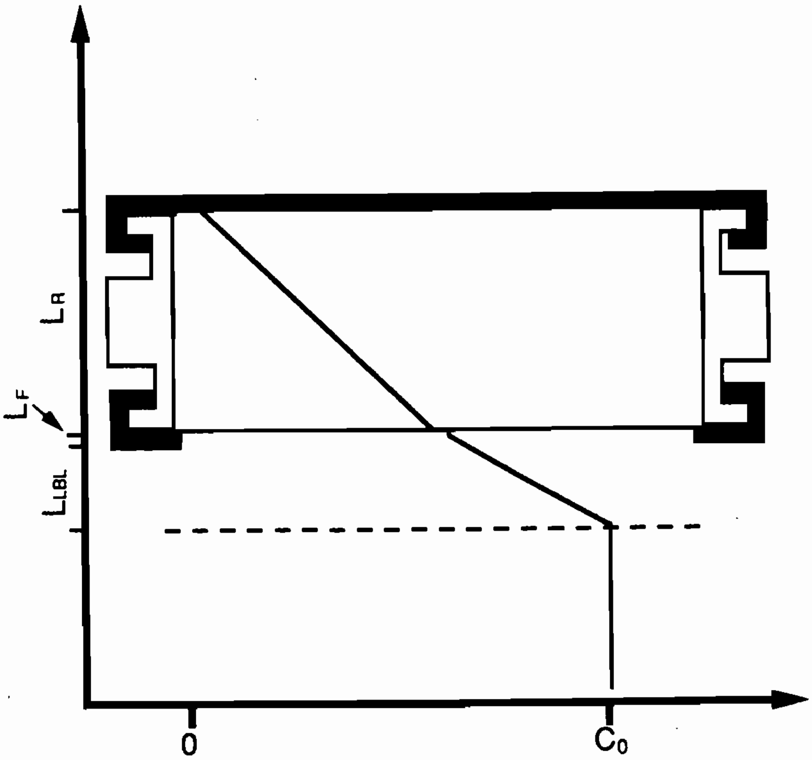


**Figure 7** The concentration gradient outside and inside the sampler.

The thickness of the laminar boundary layer below the inlet is here denoted $L_{LBL}$. The thickness of the Teflon filter is denoted $L_F$. $L_R$ is the length of the tube. The cross section areas are denoted A and have the same indexes. During sampling, the fluxes of the gas through all sections are equal, i.e. nothing is adsorbed on the walls, which is a prerequisite. The ambient concentration $C_0$

can be calculated by adding all concentration differences over all sections, i.e.:

$$C_o = \frac{X}{t \cdot D} \left( \frac{L_R}{A_R} + \frac{L_F}{A_F} + \frac{L_{LBL}}{A_R} \right) \tag{8}$$

$A_F$ is the total area of all the pores in the filter. The Fluoropore filter has 85% porosity and a thickness of 175 $\mu$m. The thickness of the laminar boundary layer can be measured during the sampling period. An extra sampler with an impregnated filter directly exposed to the air is then used. If the sorbed amount on this filter is Y the ambient concentration is:

$$C_0 = \frac{Y}{t \cdot D} \left( \frac{L_{LBL}}{A_R} \right) \tag{9}$$

$C_0$ is obtained by solving equation systems 8 and 9, i.e.

$$C_0 = \frac{X}{t \cdot D} \cdot \frac{Y}{(Y-X)} \cdot \left( \frac{L_R}{A_R} + \frac{L_F}{A_F} \right) \tag{10}$$

$L_{LBL}$ is then calculated from equation 9 and the surface transfer coefficient ($K_{Z=0}$) from $D/L_{LBL}$.

# 3 Results

## 3.1 Estimates from the closed dynamic chamber technique

As can be seen the outgoing $NH_3$ flux from the stirred chamber is approximately proportional to the flow (Figure 8).

This implies that the $NH_3$ concentration in the outgoing air was constant during the experiment, i.e. equal to $C_{eq}$ in equation 1. The $NH_3$ emission cannot, however, directly be obtained with the chamber technique. The emission can only be estimated if the ambient concentration and the mass transfer coefficient for the uncovered soil are known at the same height (see equation 1).

The ammonia volatilization from the chamber during the first 24 hours was calculated at 21.5% of the original ammonia content in the manure. During the following 24 hours another 5% volatilized. This volatilization rate is higher than that obtained from the flux measurements. The reason for this is that the F/A ratio (equation 2) is higher than the average transfer coefficient $K_Z$ (equation 1) above the uncovered soil.

The concentration of ammonia decreased exponentially during the experiment from approximately 5 ppm at the start to below 1 ppm after 16 hours (see Figure 9). The mean ammonia concentration for the first 24 hours was calculated at 1.30 ppm, which equals 746 $\mu g$ $NH_4^+$ -N/$m^3$.

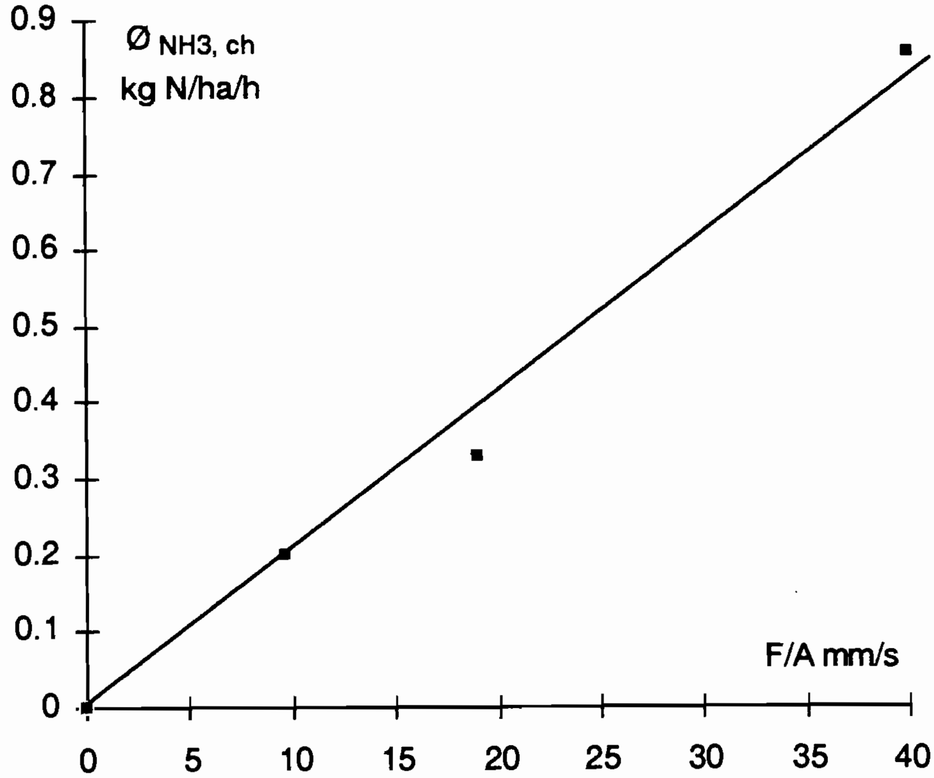


**Figure 8** Relationship between ammonia flux from a closed, stirred, dynamic chamber some hours after slurry application as a function of the F/A ratio.

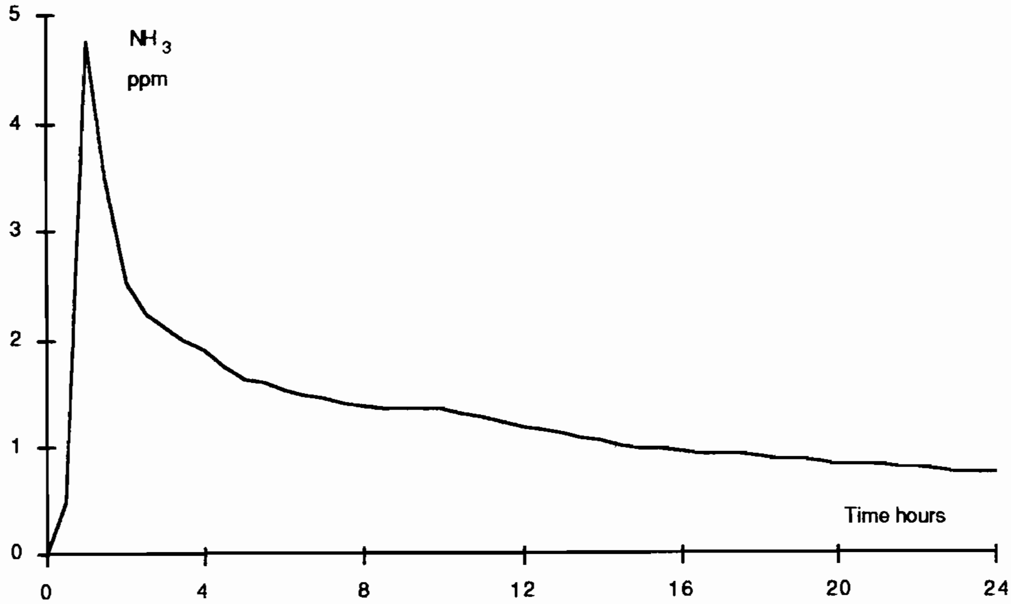


**Figure 9** Data obtained using the stirred chamber technique showing the ammonia concentration in air from cattle slurry during the first 24 hours following application.

## 3.2 Emission estimates using the passive flux technique

The total emission, estimated from the mass balance technique (passive flux), during the first 24 hours after the spreading was 87 g N, corresponding to 4.4 kg N/ha or 7.5 % of applied $NH_4^+$-N (the correction term 1.11 for a quadratic field was used). All the fluxes and horizontal transport rates are given in Table 1.

**Table 1** Fluxes in g N/m$^2$ and horizontal transport rates in g N during the first 24 h after slurry application. F indicates from the field and S from the surroundings.

| | North | | East | | South | | West | | Σ in g N | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Height | F | S | F | S | F | S | F | S | Out | In | Out-In |
| 180 | 2.36 | 0.11 | 0.22 | 0.84 | 0.29 | 1.96 | 1.05 | 0.22 | 66.8 | 53.3 | 13.5 |
| 82 | 4.22 | 0.11 | 0.36 | 0.65 | 0.51 | 1.89 | 1.31 | 0.18 | 53.4 | 23.8 | 29.6 |
| 33 | 6.25 | 0.11 | 0.51 | 0.69 | 0.98 | 1.49 | 2.51 | 1.29 | 43.7 | 11.2 | 32.5 |
| 8 | 3.27 | 0.11 | 0.4 | 0.44 | 0.84 | 0.84 | 2.69 | 0.47 | 15.2 | 4.0 | 11.3 |
| Total | 103 | 3.27 | 10.9 | 26.4 | 14.3 | 54.2 | 50.6 | 8.35 | 179.1 | 92.2 | 86.9 |

The total emission during seven days after the first 24 h was 57 g N, corresponding to 2.9 kg N/ha or 4.9 % of applied $NH_4^+$-N (the correction term 1.11 for a quadratic field was used). The figures are listed in Table 2.

**Table 2** Fluxes in g N/m$^2$ and horizontal transport rates in g N during 2nd to the 8th day after slurry application. F indicates from the field and S from the surroundings.

| | North | | East | | South | | West | | Σ in g N | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Height | F | S | F | S | F | S | F | S | Out | In | Out-In |
| 180 | 7.71 | 0.73 | 0 | 5.09 | 0.51 | 8.69 | 6.8 | 0 | 260.4 | 247.8 | 12.5 |
| 82 | 10 | 0.51 | 2 | 0.8 | 1.24 | 8.69 | 1.2 | 1.64 | 119.6 | 96.2 | 23.4 |
| 33 | 8.04 | 2.33 | 0.44 | 1.31 | 2.33 | 4.25 | 3.02 | 0.47 | 58.4 | 35.1 | 23.3 |
| 8 | 0 | 0.91 | 1.35 | 0.55 | 1.64 | 3.53 | 1.31 | 0.84 | 9.4 | 11.9 | -2.5 |
| Total | 236 | 26.9 | 23.4 | 112 | 30.6 | 233 | 158 | 19.6 | 447.8 | 391.1 | 56.7 |

Diffusional samplers were also used during the first 24 hours after the slurry spreading. The concentrations of $NH_3$, measured with two 50 mm high and diameter 12 mm diffusional samplers, which were attached to a plastic plate that was placed directly on the soil, were 0.81 and 0.86 ppm. A more sensitive sampler which was only 10 mm high and 20 mm in diameter gave concentrations of 0.82 and 0.84 ppm. The two different types gave similar results despite the fact that they measured the concentrations at 1 and 5 cm above the soil surface, respectively. A directly exposed impregnated filter received an $NH_3$ amount corresponding to $L_{LBL}$ = 3.3 mm and $K_Z$ = 6.5 mm/s. The equilibrium concentration in the simple chamber was measured using the long diffusional sampler synchronously with the passive flux samplers. Two samplers were used and they both gave a concentration of 1.9 ppm.

If the emission from the flux samplers (4.4 kg N/ha/24 h), the ambient (0.83 ppm= 0.51 mg N/m$^3$) and the equilibrium concentration in the simple chamber (1.9 ppm= 1.16 mg N/m$^3$) are used, a $K_{Z<0.05}$ value of 7.8 mm/s is obtained (see eq. 1). This transfer coefficient is close to the value obtained by the directly exposed impregnated filter (6.5 mm/s).

One measurement of the $NH_3$ emission from the slurry tank was also performed. Since the slurry tank was so wide, masts that were twice as high were used (X=16 cm). The tank was continuously stirred during the whole measurement (27 h). The net flux (Out-In) did not decrease with height to the highest point of measurement as it did on the field (Tables 1 and 2). This indicates that the heights of the masts should have been greater in this case. The results are shown in Table 3.

**Table 3** Fluxes in g N/m$^2$ and horizontal transport rates in N from the slurry tank. F indicates from the field and S from the surroundings.

| | North | | East | | South | | West | | Σ in g N | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | F | S | F | S | F | S | F | S | Out | In | Out-In |
| 360 | 0.55 | 1.02 | 3.56 | 0.11 | 2.55 | 0.91 | 0.58 | 0.8 | 549.8 | 215.5 | 334.3 |
| 164 | 1.85 | 1.89 | 5.64 | 0.18 | 5.6 | 0.65 | 0.95 | 0.91 | 529.2 | 137.1 | 392.1 |
| 66 | 2.18 | 2.04 | 10.6 | 0.36 | 12 | 0.65 | 0.91 | 1.45 | 492.1 | 86.3 | 405.8 |
| 16 | 4.8 | 2.22 | 13.9 | 3.56 | 18.9 | 1.24 | 1.78 | 0.76 | 365.1 | 71.9 | 293.2 |
| Total | 198 | 208 | 815 | 54.8 | 810 | 118 | 114 | 130 | 1936.2 | 510.8 | 1425.4 |

Since the diameter of the slurry tank was 29 m, the flux equals $1.425 \cdot 10^4/29^2/27$ = 0.63 kg N/ha/h. This is almost four times more then when spreading. The loss is, however, small in comparison to the total N content of the slurry tank.

# 4 Discussion

The measurements using the stirred dynamic chamber on a field with recently applied slurry have shown that a constant $NH_3$ concentration is obtained inside the chamber when the ratio of the ventilation flow to the covered area is small. This concentration is most likely equal to the equilibrium concentration of gaseous $NH_3$ above the soil surface. The ambient $NH_3$ concentration and the ammonia emission from a small experimental plot can be measured using two different passive sampling techniques. By combining these values with the equilibrium concentration, the mass transfer coefficient for $NH_3$ from the soil can be calculated. Conversely, the emission can be estimated if the ambient, the equilibrium concentration and the mass transfer coefficient for $NH_3$ are known. The emission is more laborious to measure than the air concentrations. If the mass transfer coefficient could be estimated by another technique, the emission from (or deposition to) a field close to a strong $NH_3$ source and the emission from a huge field could be estimated.

The transfer coefficient to a small surface acting as a perfect sink can easily be measured. If this value can be used as an approximation for the transfer coefficient for the soil surface, very simple and inexpensive measurements of the $NH_3$ emission from arable land can be performed. In the single experiment presented here $K_Z$ to the artificial surface was close to $K_Z$ for the soil. This will be of subject for further investigations.

# Acknowledgement

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# TENTATIVE EMISSION INVENTORY OF $NH_3$ IN FRANCE IN 1985

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## Abstract

The CITEPA (Centre Interprofessionnel Technique d'Etudes de la Pollution Atmosphérique) has carried out an inventory of $NH_3$ emissions in France in 1985 with a financial participation of the French Ministry of the Environment. $NH_3$ emissions from industrial sources, from agricultural sources and from nature have been estimated. $NH_3$ emissions amount to about 800,000 tonnes. 90% of these emissions are caused by agricultural activities.

# 1 Introduction

Ammonia is one of the main acid neutralizing compounds in the atmosphere. Its presence in the atmosphere is caused by the decomposition of nitrogen containing organic materials, by spreading of natural or chemical fertilizers and by some industrial or domestic sources. Using relevant emission factors, $NH_3$ emissions in France in 1985 have been estimated with a financial participation of the French Environment Ministry. $NH_3$ emissions from the main industrial and domestic sources have been estimated.

# 2 $NH_3$ emissions from industrial and domestic sources

## 2.1 Ammonia production: $NH_3$

Ammonia is produced through an operation of synthesis from $N_2$ and $H_2$. Its production involves the three following steps:

- Production of synthetic gases, $N_2$ and $H_2$ from raw materials (natural gas or naphtha) by steam reforming.

- Purification of synthetic gases.

- $NH_3$ synthesis occurring in a reactor at about 450°C and 100 to 300 bar.

There are no significant $NH_3$ emissions from the defined sources because plants work in total recycling method.

However, fugitive $NH_3$ emissions can occur by leaks on valve, pump and compressor seals. According to the leak detection and repair program, losses can vary. An average emission factor of 0.8 kg $NH_3$/t $NH_3$ produced has been taken into account (Buijsman et al., 1986). Total $NH_3$ emissions can be estimated at about 1,900 tonnes.

## 2.2 Nitric acid production: $HNO_3$

All units of $HNO_3$ are presently based on the operation of two steps:

- oxidation of $NH_3$ into NO
- oxidation of NO into $NO_2$ and absorption in water

"Single pressure" or "double pressure" processes are used (according to pressure of the second step, respectively equal or higher than the pressure of the first step). Fugitive $NH_3$ emissions can occur during storage and handling of ammonia. The emission factor varies from 0.01 to 0.1 kg $NH_3$/t N produced (Istas et al., 1988), or 0.0022 to 0.022 kg $NH_3$/t $HNO_3$ produced. Total emissions of $NH_3$ from $HNO_3$ production can be estimated at about 8 to 80 t.

## 2.3 Nitrogen containing fertilizer production

### a) Ammonium Nitrate Production: $NH_4NO_3$

Ammonium nitrate is produced through the reaction of $NH_3$ (gaseous) and $HNO_3$ (liquid). Ammonium nitrate obtained at a concentration of 81 to 83% is then concentrated by evaporation to produce low density (95 to 96% concentrated) or high density (99.5 to 99.8% concentrated) ammonium nitrate. By cooling, ammonium nitrate is crystallized and then granulated or prilled. $NH_3$ emissions can occur on the neutralizer and on the evaporator. According to measurements carried out on different plants (CITEPA, 1982), $NH_3$ emissions vary from 0.026 to 0.85 kg/t $NH_4NO_3$ produced. According to material balances (CITEPA, 1982), $NH_3$ emissions are estimated at about 3.1 kg/t $NH_4NO_3$ produced. The average emission factor presented by Buijsman (Buijsman et al., 1986) has been taken into account: 5 kg $NH_3$/t N produced (or 1.75 kg $NH_3$/t $NH_4NO_3$ produced). Total emissions of $NH_3$ can be estimated at about 3,590 tonnes in 1985.

### b) Urea production: $NH_2CONH_2$

Urea is produced through the reaction of $NH_3$ and $CO_2$. The reaction takes place in a continuously operated synthesis reactor, at high temperature and high pressure (Environmental Resources Limited, 1986). The urea solution produced is then cooled by air stripping, dried by vacuum evaporation and then prilled. Fugitive $NH_3$ emissions and $NH_3$ emissions due to unreacted $NH_3$ in vents from the reactor occur. According to Environmental Resources Limited (1986), fugitive $NH_3$ emission can be of about 5 kg $NH_3$/t urea produced with a good maintenance program. $NH_3$ emissions arising from unreacted $NH_3$ in purge gases (Environmental Resources Limited, 1986) are estimated at 0.1 to 1.5 kg/t urea produced. An average emission factor of 5 kg $NH_3$/t urea produced has been taken into account. Total $NH_3$ emissions in 1985 can be estimated at 1,520 tonnes.

### c) Mixed fertilizer production: NP, NK, NPK

According to processes and types of mixed fertilizer produced, $NH_3$ emission factors could vary between 0.12 to 0.7 kg $NH_3$/t mixed fertilizer produced (VDI, 1990). An average emission factor of 0.5 kg $NH_3$/t mixed fertilizer produced has been taken into account in these estimations. It is in agreement with the 12.5 kg $NH_3$/t N produced given by study (Buijsman et al., 1986). (Mixed fertilizer must contain at least 3% in weight of nitrogen. Considering this content, 12.5 kg $NH_3$/t N produced is equivalent to about 0.4 kg $NH_3$/t mixed fertilizer produced). 6,780 kt of mixed fertilizer have been produced in France in 1985. The resulting $NH_3$ emissions can be estimated at about 3,400 tonnes.

## 2.4 Other industrial sources

### a) Coal combustion

$NH_3$ emissions can be generated by coal combustion (Istas et al., 1988). An average emission factor of 12.5 g $NH_3$/t coal burned has been taken into account (10 to 15 g N/t of coal burned) (Istas et al., 1988). In 1987, the coal consumption has been 23,7.$10^6$t. $NH_3$ emissions could be of about 360 tonnes.

**b) Road Traffic**

Istas' study (Istas et al., 1988) presents an emission factor of 0.025 g $NH_3$/km. Taking into account vehicle statistics and average mileage, $NH_3$ emissions can be estimated at 9,400 tonnes.

# 3 $NH_3$ emissions from agriculture and farming

## 3.1 Domestic animals

Animal wastes by way of faeces and urine are a source of $NH_3$ emissions.

The annual production of wastes is estimated at (CITEPA, 1987):

| | |
|---|---|
| 320 | million tons from cattle, |
| 20 | million tons from pigs, |
| 16 | million tons from sheep, |
| 3 | million tons from poultry, |

After analysis of literature data, $NH_3$ emission factors given by Buijsman et al. (1986) have been taken into account (see Table 1).

Table 1. Emission factors

| Animal type | $NH_3$ emission factor (kg $NH_3$/year animal) |
|---|---|
| cattle | 18.00 |
| pig | 2.80 |
| poultry | 0.26 |
| horse | 9.4 |
| sheep | 3.1 |

Total emissions have been estimated by the author cited above, taking into account the number of animals, their age and their weight, for the year 1982. The $NH_3$ emissions in 1982 are presented in Table 2 (Buijsman et al., 1986). They are assumed to be the same in 1985. These estimations take into account the emissions during housing of animals and the emissions when animals are at grass.

Table 2. Emissions of $NH_3$ in 1982

| | |
|---|---|
| Cattle | 447.5 kt |
| Pigs | 25.6 kt |
| Poultry | 48.8 kt |
| Horses | 2.6 kt |
| Sheep | 44.6 kt |
| TOTAL | 569.2 kt |

## 3.2 Application of fertilizers

Nitrogen containing fertilizers are a source of $NH_3$ emissions. After their application on soils, $NH_3$ emissions depend on the temperature, the humidity, the pH and the nature of soils. Emission factors of Buijsman et al. (1986) have been retained. They are as follows (% of N applied):

Table 3. Emission factors for fertilizer

| | |
|---|---|
| Ammonium sulfate | 15 |
| Ammonium nitrate | 10 |
| Urea | 10 |
| Ammonia | 10 |
| Ammonium and calcium nitrate | 2 |
| Other | 1 |

Taking into account the distribution of fertilizers applied in France in 1985 (Ministère de l'Agriculture, 1986), an average $NH_3$ emission factor of 6.1 % of nitrogen applied has been taken into account. In 1985, 2,407 kt N have been applied (Ministère de l'Agriculture, 1986). $NH_3$ emissions can be estimated at about 145 kt.

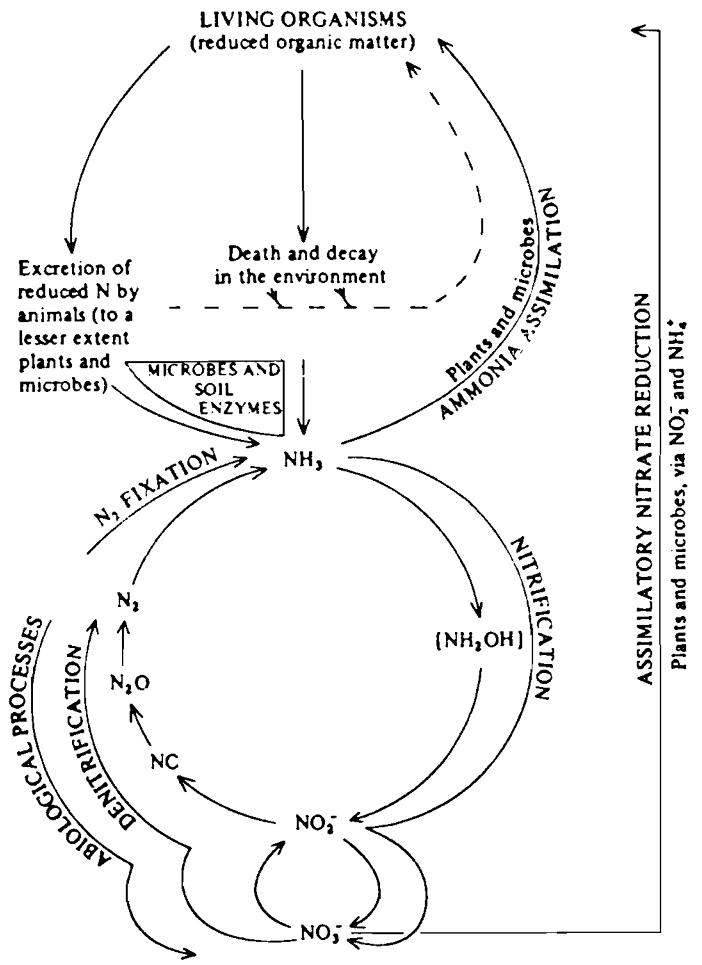


Figure 1. Natural emissions

# 4 Natural emissions

In natural soils, the transformation of organic materials by micro-organisms or the direct fixation of atmospheric nitrogen by micro-organisms involve $NH_3$ emissions. The complex reactions involved are presented in Figure 1 (J. Sprent, 1987). Natural $NH_3$ emissions are estimated to about 0 to 20 $\mu g.\ m^{-2}.h^{-1}$. Taking into account an average emission factor of 10 $\mu g.\ m^{-2}.h^{-1}$, total natural $NH_3$ emissions can be estimated at 48,000 t. (All the French area has been assumed emitting).

# 5 Conclusions

Total $NH_3$ emissions in France in 1985 can be estimated at about 780,000 tons. The distribution of these emissions is as follows:

| | |
|---|---|
| Industrial and domestic sources | 20,250 t (2.5%) |
| $NH_3$ production | 1,900 t |
| $HNO_3$ production | 80 t |
| $NH_4NO_3$ production | 3,590 t |
| Urea production | 1,520 t |
| Nitrogen mixed fertilizers production | 3,400 t |
| Coal combustion | 360 t |
| Traffic | 9,400 t |
| Agriculture and farming | 714,200 t (91.3%) |
| Farming | 569,200 t |
| Fertilizer application | 145,000 t |
| Natural emissions | 48,000 t (6.2%) |
| TOTAL | 782,450 t (100%) |

The main $NH_3$ emission sources are agriculture and farming which represent together more than 90% of total $NH_3$ emissions. Industrial sources give very low $NH_3$ emissions. Many uncertainties still exist regarding these estimates.

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# SPECIAL CONDITIONS INFLUENCING AMMONIA EMISSION FACTORS IN SWITZERLAND

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## Abstract

So far ammonia emission factors for Switzerland were derived from studies carried out in other countries. To work out specific recommendations for the abatement of ammonia emissions on individual farms, we need differentiated emission factors which allow the assessment of the emissions under specific conditions. Current projects on agricultural ammonia emissions will concentrate on questions particular for Swiss agriculture. Some of these are: the complex topography, variable meteorological conditions, stable systems which produce solid manure plus urine-rich liquid manure, mixing of cattle and pig slurry, heavy slurry dilution, high proportion of grass in cattle feed rations, low proportion of nitrogen excreted by dairy cows during grazing, and relatively low N-fertilizer input. The main objective of the projects is to produce differentiated norm values of ammonia emissions to facilitate an optimal management of manure and the abatement of ammonia emissions.

# 1 Introduction

Different Swiss authors have measured nitrogen losses together with other variables (e.g. Besson et al., 1990, 1985; Häni et al., 1990; Gisiger, 1968; Liechti und Ritter, 1910; Neftel et. al., 1990; Ott, 1990a), but these measurements cannot be extrapolated to global ammonia ($NH_3$) emissions. So far, assessments of ammonia emissions in Switzerland (Stadelmann, 1990) were therefore based on the ammonia emission inventory of Buijsman et al. (1985, 1987). Projects to determine specific $NH_3$ emission factors are still in their initial phases. To be able to produce the data required to evaluate emissions and possible abatement measures within reasonable time, we have to concentrate our work on specific Swiss conditions and combine the results with general information from the literature.

When deriving the global or national ammonia emission, emission models will probably give better results than detailed measurements. However, specific information on emission factors under a wide variety of conditions is necessary for a detailed inventory of regional emission and for giving recommendations to farmers on the optimal utilization of animal excretion and on measures to reduce ammonia emissions (Häni et al., 1990). These factors have to be derived from experiments in the stables and in the field. They are likely to vary strongly from farm to farm and from country to country due to the specific conditions of production (production intensity, composition of the feed ration, manure management, climatic conditions etc.). As natural conditions, technical infrastructure and methods of production in Switzerland are, in many cases, different from those in the countries where most ammonia studies have been conducted so far (Netherlands, UK, Germany), Swiss ammonia emission factors too, could differ from those found in literature.

To be of use for fertilization recommendations specific emission factors for fertilizers and manure should be accurate to about $\pm$ 10 per cent. This accuracy is generally difficult to achieve.

# 2 General ammonia emission factors and total ammonia emissions in Switzerland

The emission factors per animal used in the assessments of the ammonia emission in Swiss agriculture (Buijsman et al., 1987; Stadelmann, 1990) are summarized in Table 1. These emission factors are today considered as too low (Asman 1990; Möller and Schieferdecker, 1990) so that the total agricultural emission of ammonia in Switzerland is likely to be higher than the 56 ktons per year calculated by Stadelmann (1990), even though the number of cattle and pigs has decreased during the past few years. The emission factors of Asman (1990) combined with the number of livestock in 1988 and the mineral fertilizer emission factors of Buijsman et al. (1987) give a total agricultural emission of 63 ktons of ammonia. This is clearly higher than the 55 ktons given by Klaassen (1990).

Systematic measurements of ammonia emissions are still preliminary in our country and it is, therefore, difficult to evaluate the accuracy of the estimates listed in Table 1. Recent studies carried out at our institute have shown that the average annual N production per cow or per pig has increased significantly during the last decades due to changes in the production intensity (Flückiger et al., 1989; Menzi, 1991). The average amount of N excreted by those animals is therefore higher than the norm values used so far. This would also imply that N losses were probably underestimated. On the other hand, some of the values suggested by Buijsman et al. (1987) could probably be reduced when specific conditions of Swiss agricultural production systems are considered.

Table 1. Ammonia emission factors and total agricultural ammonia emissions for Switzerland given by different authors

| Emission factor ($NH_3$ kg/animal/year) | | | | |
|---|---|---|---|---|
| | Buijsman et al. (1987) | Stadelmann (1990) | Asman (1990) | Klaassen (1990) |
| Cattle | 18 | 21.3 | 25.1 | 12.5 - 24.9 |
| Pigs | 2.8 | 2.1 | 4.8 | 4.8 |
| Poultry | 0.26 | 0.26 | 0.32 | 0.28 - 0.33 |
| Horses | 9.4 | 11.1 | 12.5 | 12.5 |
| Sheep | 3.1 | 2.7 | 1.9 | 2.1 |
| Total agricultural emissions (kton $NH_3$ per year) | | | | |
| Reference year | 1983 | 1987 | 1988 | 1987 |
| | 53 | 56 | 68 (63)* | 55 |

* Combined with livestock numbers for 1988 (own calculation)

Given the general emission factors per animal, the total $NH_3$ emission in Switzerland as well as its regional distribution can be calculated exactly using the yearly animal census which shows the number of animals per commune (average size per commune 13.7 $km^2$). The distribution of cattle and pigs shown in Figure 1 reveals that high emissions per unit area can be expected mainly in the northeastern and central parts of Switzerland where the density of cattle and in particular of pigs is high.

# 3 Special conditions in Switzerland

## 3.1 Natural conditions

The topography in most parts of Switzerland leads to a great variety of soil types and climatic conditions. These factors which influence ammonia emission (Häni et al., 1990), therefore, can vary strongly within short distances. In

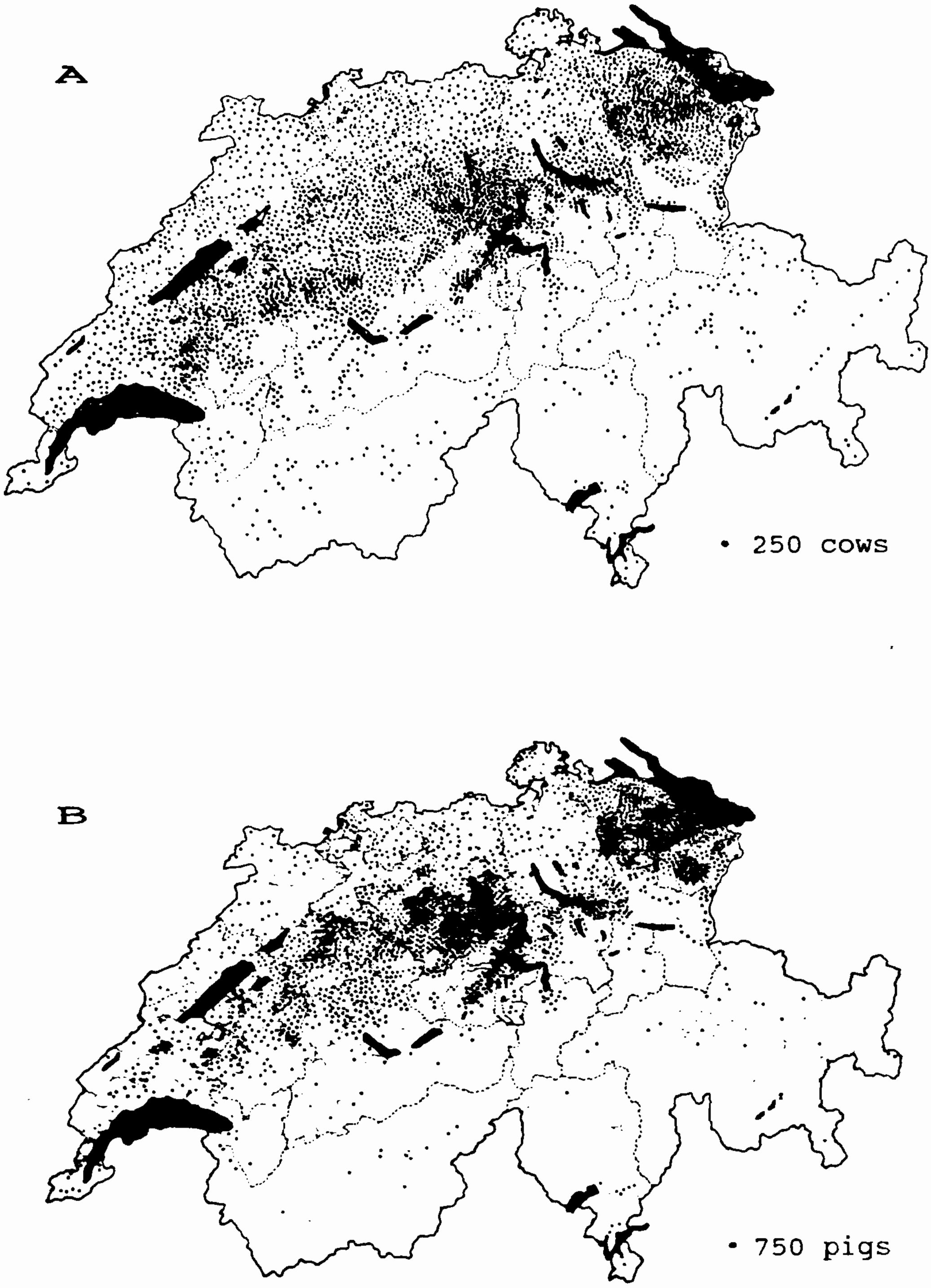


Figure 1. Regional livestock distribution in Switzerland. A: cows; B: pigs.

addition, meteorological conditions are generally unstable. It is therefore very difficult to forecast emissions in the field. These meteorological and topographic conditions will also complicate field measurements of $NH_3$ emissions.

In most areas, slurry can not be injected or incorporated into the soil due to the topography, heavy soils, presence of stones (> 10 cm) and the low proportion of arable land (national average: less than 30 %; mountain areas: 0 %). Most of the slurry and even a high proportion of the solid manure are therefore spread on grassland where N losses are comparatively high since much of the ammonia can volatilize before coming into contact with the soil.

## 3.2 Technical infrastructure

The average size of a Swiss farm is only about 15 hectare which results in a rather high production intensity. As a further consequence possibilities for abating $NH_3$ emissions by technical means or by improved mechanization are very limited because farms and plots are too small to warrant the use of big and expensive machines and agricultural implements.

Most slurry is stored in closed pits. $NH_3$ losses during storage can therefore be assumed to be rather low (De Bode, 1990). Systems with slurry storage below the slatted stable floor are hardly known in Switzerland.

Contrary to systems prevailing in many European countries where all the excrements are collected together, more than 50 % of the cows in Switzerland are kept in stables where solid manure (feces and straw) and urine-rich slurry or liquid manure (urine, part of the feces) are separated. Very little is known about $NH_3$ emission from such stable systems and from these types of manure. With its low ammonium content ($NH_4^+$) of about 20 %, solid manure is expected to have a low emission (Besson et al., 1990). On the other hand, long, open storage and the application of solid manure on grassland could well lead to a rather high emission (Ott, 1990b). As most of the urine is collected in the liquid manure, this slurry has a high $NH_4^+$content (65 % of total N; about 2.5 - 4 kg per $m^3$ of undiluted slurry). $NH_3$ volatilization could, therefore, be high, especially after spreading it on grassland. Treatment during storage of liquid and solid manure may also have a strong effect on ammonia emissions during storage and after application (Besson et al., 1985).

It is very common in Switzerland to keep cattle and pigs on the same farm. The slurry of both stables is usually collected in the same pit. As pig and cattle slurry show considerable differences in the emission of ammonia (Döhler, 1990), the total emission of this mixture may be different than would have been the case had the slurry been treated separately.

## 3.3 Agricultural practices

In Switzerland slurry is generally diluted with water in a ratio of 1 : 0.5 (parts excrements : parts water) to 1 : 3 (Hilfiker, 1989). A dilution of 1 : 1 for pig slurry and 1 : 1.5 or even 1 : 2 for cattle slurry could be an appropriate average. This dilution is much higher than what we find in most Dutch or German studies. A dilution of 1 : 1 can already reduce emission after application by one half (Döhler, 1990), due to the lower concentration and the better infiltration.

Even though the average milk yield is between 5000 and 6000 kg per cow per year, Swiss cows consume nearly 80 % of their energy intake in the form of basic forage (grass, hay, silage etc.; Menzi and Gantner, 1987). This percentage is higher than in other countries with similar production level (except New Zealand). Such a feeding practice might influence the composition of manure and therefore $NH_3$ emission. The great variety of rations used in Swiss farms and considerable regional differences in feeding practices and forage production (Menzi and Thomet, 1985) may also be of importance. Another relevant factor may be the four cattle races common in Switzerland.

Only about two thirds of the dairy cows in Switzerland are grazed during summer. On average these grazed cows spend about 10 hours per day on the pasture for 150 to 220 days per year. This means that only about 15 % of the total nitrogen excreted by cows is on the pastures. This is less than in other countries and than is usually assumed in ammonia emission inventories (Buijsman et al. 1987; Klaassen, 1990).

In Switzerland, an average quantity of about 220 kg of nitrogen is applied per hectare of agricultural land. Only one third of this is in the form of mineral fertilizer. In comparison, in the Netherlands, in 1985, 530 kg of N, 46 % as mineral fertilizer were applied (Van Boheemen, 1987). The relatively low nitrogen level in Switzerland will influence $NH_3$ emission per unit area in the field. A further consequence is a lower nitrogen content in the grass, in spite of the rather high proportion of clover in all Swiss pastures and meadows. It can, therefore, be expected that N consumption and N excretion per cow is lower in Switzerland than in most other European countries. In fact, recent detailed studies have shown an average excretion of about 110 kg of N per cow per year (Flückiger et al., 1989; Menzi, 1991) which is clearly less than the 189 kg for the Netherlands or the 134 kg for Switzerland suggested by Klaassen (1990) on the basis of Dutch data. Consequences of these lower N excretions will be lower $NH_3$ emission factors per cow than in the Netherlands and a lower abatement potential through feeding measures.

So far, we have not evaluated the specific importance of the different kinds of mineral N fertilizers in Switzerland. The most commonly used type is ammonium nitrate.

# 4 Conclusions and future research

The conditions mentioned above will not necessarily lead to significant corrections of the general ammonia emission factors presently applied but they will allow a detailed regional differentiation of the emissions. This is important for the emission inventory and for immission models. A more detailed evaluation of factors affecting ammonia emission will also provide a firm basis for introducing abatement measures on individual farms. These will be of great public interest (ecology; Isermann, 1990; Flückiger, 1988) and very welcome to farmers, enabling them to reduce the costs of fertilization through a more optimal use of manure.

The urgency of the $NH_3$ problem and the retarded beginning of Swiss research on this topic requires a rapid progress to be made with projects in this area. Experiments will therefore be restricted mainly to a consideration of particularly Swiss conditions as mentioned above. Some variables in the project could be solid manure and urine-rich slurry, dilution, importance of feed rations with a high proportion of grass, and emission in Swiss stables. The experiments will be carried out in close collaboration by different agricultural research stations.

The objective of a project being carried out at our institute since 1986 is the development of new differentiated norms for the amount and the composition of manure produced by different domestic animals. As the example in Table 2 (Menzi, 1991) shows, these norms should provide the farmer with a basis for considering specific factors on his farm such as milk yield, ration composition and composition of the grass sward when calculating the annual amount of nutrients (N/P/K) in the manure of his animals. It is the aim of these new norms to provide the basis for the optimal use and distribution of manures. However, this objective will not be achieved fully as long as we have to assume that between 2 and 50 % (Isermann, 1990) of the nitrogen in the manure will be lost as ammonia after the application on the field. Therefore we hope that the results of the $NH_3$ emission study will also allow us to include factors for ammonia losses in the new norms, so that these losses can also be assessed differentially. For example the effects of dilution and climatic conditions during and after the application of manure could be included in these factors. A similar system was proposed by Horlacher and Marschner (1990).

Table 2. N, P and K excreted per cow per month: amount of undiluted slurry (total excrements) produced. Excerpt from: Menzi (1991). All values for a standard cow with 5000 l milk produced per year and 600 kg live weight.
Balanced grass/clover mixture in pastures and meadows.
N - after 15 % reduction for $NH_3$ losses in the stable and during manure storage.

| **a) Standard values** | | | | |
|---|---|---|---|---|
| | N (kg) | P (kg) | K (kg) | $m^3$ |
| | per cow per month | | | |
| Average whole year | 7.5 | 1.5 | 12 | 1.7 |
| Summer rations (grass etc.) | 8.6 | 1.5 | 13.8 | 1.9 |
| Winter rations (hay, silage etc.) | 6.2 | 1.5 | 9.7 | 1.4 |
| **b) Correction factors (%)** | | | | |
| Production level ± 1000 l milk per year | ± 10 | ± 10 | ± 10 | ± 10 |
| Basic forage ration grass + dried maize (summer) | - 11 | - 5 | - 8 | - 6 |
| Grass + silage (winter) | + 3 | + 2 | - 3 | + 5 |
| Hay only (winter) | - 5 | - 3 | - 1 | - 11 |
| Sward composition | | | | |
| Sward with few legumes | - 12 | + 4 | - 8 | -- |
| Sward rich in legumes | + 6 | + 4 | + 15 | + 1 |

Our institute is also involved in working out an interdisciplinary project on N deposition. First contacts between different working groups in Switzerland and with experts in other countries have already been made.

Last but not least it is our hope that the ammonia emission projects will promote an interest in ecological considerations in agriculture, by demonstrating nitrogen fluxes and the effect of agricultural operations on the efficient use of nitrogen.

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# PRELIMINARY ESTIMATE OF AMMONIA EMISSIONS IN ITALY

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## Abstract

A preliminary estimate of ammonia ($NH_3$) emissions in Italy for the years 1985-1988 is presented. Total emissions are estimated at about 500 kt. The main sources, livestock animals, fertilizer application and natural activities from soils account respectively for 60%, 20% and 17%.

# 1 Introduction

A preliminary estimate of ammonia ($NH_3$) emissions in Italy for the years 1985-1988 is presented, based on national statistical data and literature figures on emission factors, mostly proposed by Dutch researchers. The year 1985 has been chosen as reference year, as already done for the CORINAIR emission inventory.

# 2 Livestock animals

$NH_3$ emission factors for animals (Asman and van Jaarsveld, 1991) are listed in Table 1; the number of animals for the different species is given in Table 2; the estimated emissions are shown in Table 3 and Figure 1. Emissions from poultry have not been considered, as there are no official statistics for this species.

Table 1. $NH_3$ emission factors for livestock animals (g/head)

| Animal | Emission factors |
|---|---|
| Cattle | 25118 |
| Pigs | 4824 |
| Horses | 12500 |
| Sheep | 1908 |

Table 2. Number of livestock animals (thousands of heads)

| Livestock animal | 1985 | 1986 | 1987 | 1988 |
|---|---|---|---|---|
| Cattle | 9009 | 8921 | 8898 | 8843 |
| Pigs | 9169 | 9278 | 9383 | 9360 |
| Horses | 398 | 396 | 385 | 384 |
| Sheep | 11293 | 11451 | 11456 | 11623 |

Source: Istat, 1989a; Istat, 1990a

Table 3. $NH_3$ emissions from livestock animals ($10^3$ t)

| Animal | 1985 | 1986 | 1987 | 1988 |
|---|---|---|---|---|
| Cattle | 226.3 | 224.1 | 223.5 | 222.1 |
| Pigs | 44.2 | 44.8 | 45.3 | 45.2 |
| Horses | 5.0 | 5.0 | 4.8 | 4.8 |
| Sheep | 21.5 | 21.8 | 21.9 | 22.2 |
| Total | 297.0 | 295.7 | 295.5 | 293.3 |

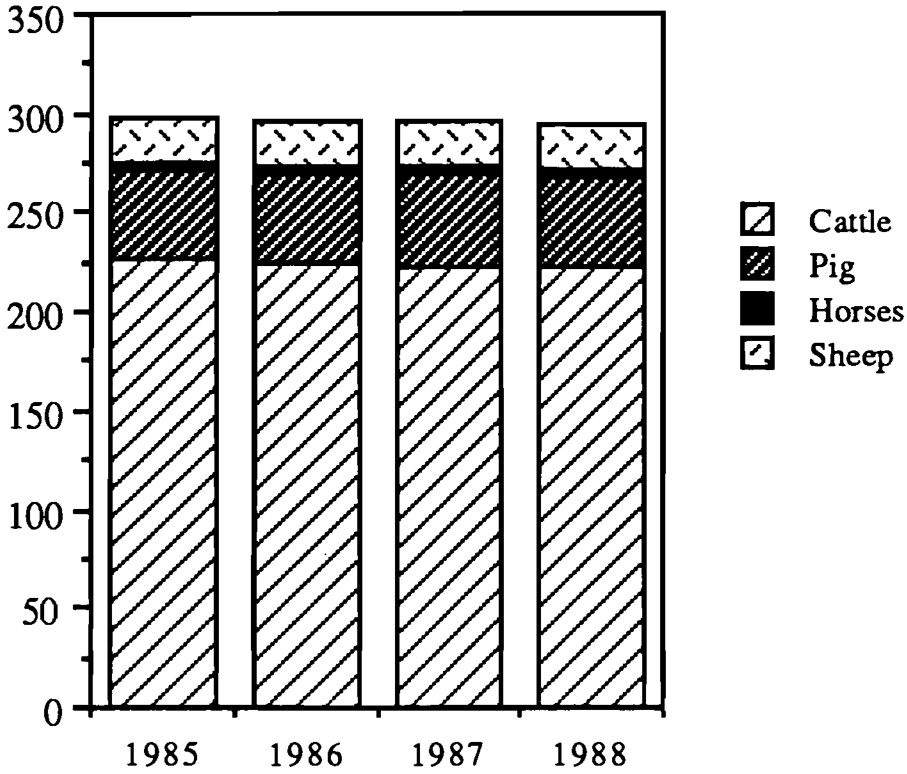


Figure 1. $NH_3$ emissions from livestock animals ($10^3$ t)

# 3 Fertilizer application

$NH_3$ emission factors, expressed as g/t of nitrogen contained in fertilizers, for fertilizer application are listed in Table 4 (Asman and van Jaarsveld, 1991). The amount of nitrogen in the consumption of N-fertilizers is reported in Table 5 (Istat, 1987; Istat, 1988; Istat, 1989a; Istat, 1990a). Emissions of ammonia from nitrogenous fertilizers are shown in Table 6 and Figure 2.

Table 4. $NH_3$ emission factors for N-fertilizer application (g $NH_3$/t of nitrogen)

| Fertilizer application | Emission factors |
|---|---|
| Ammonium sulphate | 180000 |
| Ammonium nitrate | 120000 |
| Urea | 120000 |
| NPK fertilizers | 60000 |

Table 5. Amount of nitrogen in the consumption of N-fertilizers (t)

| Nitrogen | 1985 | 1986 | 1987 | 1988 |
|---|---|---|---|---|
| Ammonium sulphate | 72290 | 77100 | 69260 | 65670 |
| Ammonium nitrate | 223540 | 197880 | 209420 | 217770 |
| Urea | 429440 | 421010 | 418780 | 376470 |
| NPK fertilizers | 262290 | 282260 | 305350 | 304320 |

Table 6. $NH_3$ emissions from N-fertilizers ($10^3$ t)

| N—fertilizer | 1985 | 1986 | 1987 | 1988 |
|---|---|---|---|---|
| Ammonium sulphate | 13.0 | 13.9 | 12.5 | 11.8 |
| Ammonium nitrate | 26.8 | 23.7 | 25.1 | 26.1 |
| Urea | 51.5 | 50.5 | 50.3 | 45.2 |
| NPK fertilizers | 15.7 | 16.9 | 18.3 | 18.3 |
| Total | 107.0 | 105.0 | 106.2 | 101.4 |

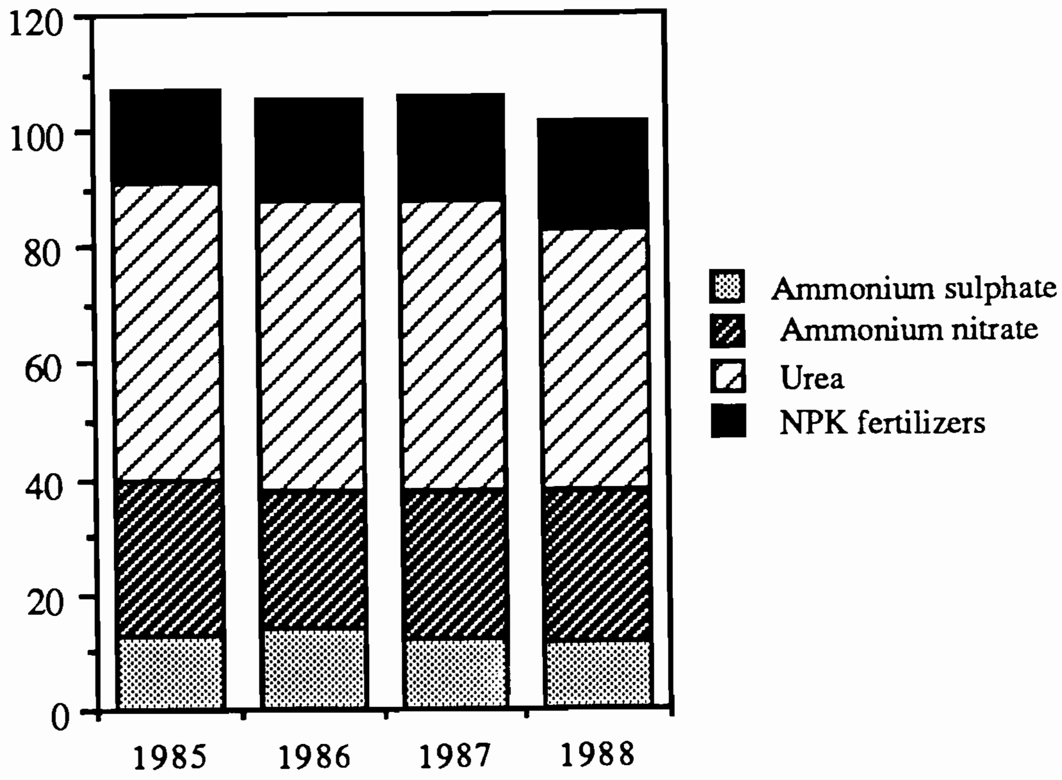


Figure 2. $NH_3$ emissions from N fertilizers ($10^3$ t)

# 4 Industrial processes

Emission factors for the main industrial processes which are relevant for $NH_3$ emissions are listed in Table 7. The source of these values is TNO (TNO, 1984); for coke production the assumed value is drawn from EPA (EPA, 1985). Table 8 shows the basic data for the emission estimate from industrial processes. The relevant emissions are listed in Table 9.

Table 7. Emission factors for industrial processes (kg/t)

| Production | Emission factors |
|---|---|
| Ammonia | 0.8 |
| Nitric acid | 0.05 |
| Urea | 0.350 |
| Ammonium nitrate | 1.45 |
| NPK fertilizers | 2.50 |
| Acrylonitrile | 0.2 |
| Caprolactam | 0.5 |
| Coke | 0.1 |
| Zinc | 0.11 |

Table 8. Industrial productions (t)

| Processes | 1985 | 1986 | 1987 | 1988 |
|---|---|---|---|---|
| Ammonia[a] | 1799272 | 1888586 | 1745111 | 1747675 |
| Nitric acid[b] | 1125159 | 1169257 | 1130059 | 1218769 |
| Urea[a] | 118081 | 1158854 | 1063712 | 1292594 |
| Ammonium nitrate[a] | 932740 | 978405 | 999908 | 1015960 |
| NPK fertilizers[a] | 2249590 | 2102987 | 2347096 | 2323132 |
| Acrylonitrile[c] | 89615 | 126333 | 130000 | 130000 |
| Caprolactam[d] | 210000 | 210000 | 211296 | 180000 |
| Coke[e] | 7412000 | 7193000 | 6853000 | 6723000 |
| Zinc[f] | 210075 | 230400 | 247000 | 242100 |

(a) Istat, 1990b
(b) Factory data
(c) Istat, 1989b; Istat, 1989c (1987, 1988 estimated values)
(d) Istat, 1990b (1985, 1986 estimated values)
(e) Ministero dell'Industria, 1989
(f) Nuova Samim, 1990

Table 9. $NH_3$ emissions from industrial processes ($10^3$ t)

| Production | 1985 | 1986 | 1987 | 1988 |
|---|---|---|---|---|
| Ammonia | 1.4 | 1.5 | 1.4 | 1.4 |
| Nitric acid | — | — | — | — |
| Urea | 0.4 | 0.4 | 0.4 | 0.4 |
| Ammonium nitrate | 1.4 | 1.4 | 1.4 | 1.5 |
| NPK fertilizers | 5.6 | 5.3 | 5.9 | 5.8 |
| Acrylonitrile | — | — | — | — |
| Caprolactam | — | — | — | — |
| Coke | 0.7 | 0.7 | 0.7 | 0.7 |
| Zinc | — | — | — | — |
| Total | 9.5 | 9.3 | 9.8 | 9.8 |

(-) negligible because the emissions are < 250 t

# 5 Transport

Transport emissions (listed in Table 10) have been estimated using an emission factor of 3 mg/km for diesel cars; the emission factor for gasoline cars is negligible (Eggleston, 1991), considering the very small number of them which is fitted with three-way catalysts in Italy.

Table 10. $NH_3$ emissions from transport ($10^3$ t)

| Year | Emissions |
|---|---|
| 1985 | 0.3 |
| 1986 | 0.4 |
| 1987 | 0.4 |
| 1988 | 0.4 (*) |

(*) provisional estimate

# 6 Combustion of lignite and coal

The reported emission factor for lignite is 0.86 g/GJ (Möller et al., 1989) and for coal is 0.48 g/GJ (Allemand, 1991). The resulting emissions are negligible.

# 7 Natural activities

Emissions from natural sources originate from forests soils and other vegetation. An emission factor of 3.6 kg/ha (Möller et al., 1989) has been used (Table 11).

Table 11. $NH_3$ emissions from natural sources ($10^3$ t)

| Year | Deciduous forests | Conifer forests | Other vegetation | Total |
|---|---|---|---|---|
| 1985 | 29.4 | 5.6 | 50.5 | 85.5 |
| 1986 | 29.5 | 5.6 | 50.5 | 85.6 |
| 1987 | 29.6 | 5.9 | 50.0 | 86.0 |
| 1988(*) | 29.6 | 5.9 | 50.0 | 86.0 |

(*) provisional estimate

# 8 Landfilling and sludge spreading

As concerns emissions from landfills, the ratio of nitrogen emissions to methane is about 7.3%; of this, about 10% is ammonia (Eggleston, 1991). For landfills in Italy, we have calculated an emission factor of 6.65 kg/t of $CH_4$ per ton of waste; the resulting emission factor for $NH_3$ is 60 g/t. The amount of wastes treated in landfills is listed in Table 12; the relevant emissions are shown in Table 13. The available emission factor for sludge spreading is 0.5 kg/t (Eggleston, 1991). The resulting emissions are $1.8 \cdot 10^3$ tons.

Table 12. Amount of municipal and industrial solid wastes in landfills and sludges from sewage treatment plants ($10^3$ t)

| | 1985 | 1986 | 1987 | 1988 |
|---|---|---|---|---|
| Landfilling | 119.2 | 122.9 | 126.5 | 130.3 |
| Sludge spreading | 3.5 | 3.5 | 3.5 | 3.5 |

Table 13. $NH_3$ emissions from landfilling and sludge spreading ($10^3$ t)

| | 1985 | 1986 | 1987 | 1988 |
|---|---|---|---|---|
| Landfilling | 7.0 | 7.2 | 7.4 | 7.7 |
| Sludge spreading | 1.8 | 1.8 | 1.8 | 1.8 |
| Total | 8.8 | 9.0 | 9.2 | 9.5 |

# 9 Conclusions

$NH_3$ emissions in Italy, grouped by activities, are listed in Table 14 and shown in Figure 3. The results show that livestock animals and fertilizers application are the main sources; natural emissions from soils are also relevant. Total $NH_3$ emissions are stationary during the years 1985-1988.

Table 14. $NH_3$ emissions in Italy ($10^3$ t)

| Activity | 1985 | 1986 | 1987 | 1988 |
|---|---|---|---|---|
| Livestock animals | 297.0 | 295.7 | 295.5 | 293.3 |
| Fertilizers application | 100.7 | 105.0 | 106.2 | 101.4 |
| Production processes | 9.5 | 9.3 | 9.8 | 9.8 |
| Transport | 0.3 | 0.4 | 0.4 | 0.4 |
| Nature | 85.5 | 85.6 | 86.0 | 86.0 |
| Landfilling and sludge spreading | 8.8 | 9.0 | 9.2 | 9.5 |
| Total | 501.8 | 505.0 | 507.1 | 500.4 |

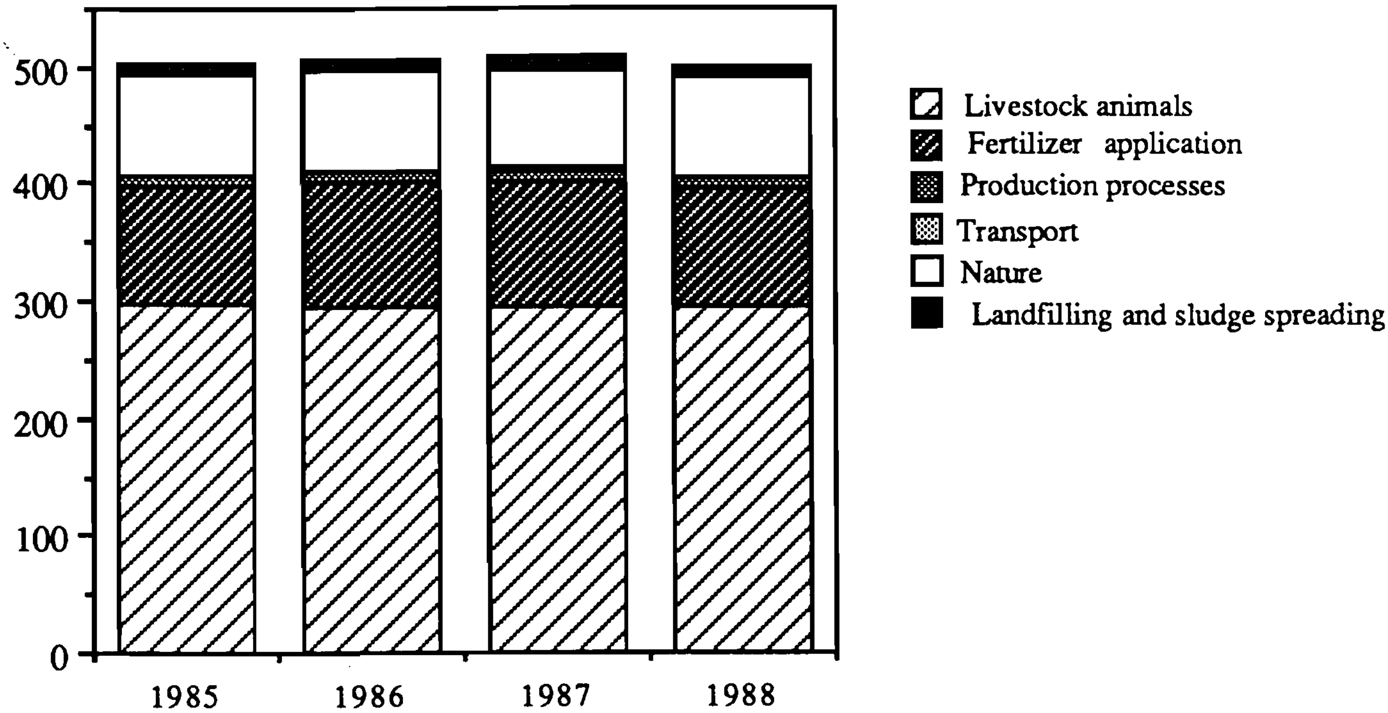


Figure 3. $NH_3$ emissions in Italy ($10^3$ t)

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# A DETAILED AMMONIA EMISSION INVENTORY FOR DENMARK AND SOME DEPOSITION CALCULATIONS

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## Abstract

The total ammonia ($NH_3$) emission is about 149,000 tonne $NH_3$ $a^{-1}$. Model calculations show that dry deposition of $NH_x$ ($NH_3$ + $NH_4^+$) is more important than wet deposition of $NH_x$ in Denmark (land area). About 74% of the total (dry + wet) deposition of $NH_x$ to Denmark originates from Danish sources. But only 37% of the $NH_3$ emitted in Denmark is deposited as $NH_x$ in Denmark.

# 1 Introduction

Nitrogen components are important because they act as fertilizers. In Denmark not only nitrogen deposition to forests is a matter of concern, but also nitrogen deposition to the sea. Nitrogen is a controlling factor for the growth of algae in the Danish coastal waters. High loads of nitrogen in sea areas can therefore lead to algal blooming, and then to an oxygen deficit after degradation of the algae which have sunk to the bottom. This oxygen deficit causes the death of fish and benthic organisms. These effects on the marine ecosystem are the main reason for the extensive Danish research programmes on the nitrogen cycle.

NERI is involved in this research e.g. by developing methods to estimate the deposition of atmospheric nitrogen components to the sea. This research is undertaken in cooperation with Risø National Laboratory and the Danish Meteorological Institute within the framework of the Marine Research Programme '90.

The concentrations and depositions of nitrogen components show a large spatial variability. Moreover, it is difficult and expensive to measure at sea. This is the reason why the concentrations and depositions will be estimated using a limited number of measurements combined with the results of atmospheric transport models. In order to model the depositions, detailed gridded emission inventories of ammonia and nitrogen oxides are needed. These inventories have been made recently (Asman, 1990A; Asman and Runge, 1991).

In this paper the emission inventory for $NH_3$ for Denmark will be presented together with some calculations using the TREND model, which was developed at the National Institute of Public Health and Environmental Protection (RIVM), the Netherlands. The TREND model was used to get a first estimate of the concentrations of ammonia ($NH_3$) and ammonium ($NH_4^+$) in air and the deposition of $NH_x (= NH_3 + NH_4^+)$. Within the Marine Research Programme '90 an atmospheric transport model will be developed which is chemically more sophisticated than the TREND model and which is able to take into account different dry deposition velocities for different surfaces, which the TREND model can not. However, the more sophisticated model is not able to give such spatially detailed information as the TREND model.

# 2 Ammonia emission inventory

The emission factors used to make the inventory are the same as used by Asman (1990B) for Europe (see also: Asman and van Jaarsveld, 1991, this volume). Table 1 gives an overview of the $NH_3$ emissions for Denmark. The

Table 1. Emission of $NH_3$ in Denmark (tonne $NH_3$ $a^{-1}$).
(For computational reasons the data is given more accurately than actually known.)

| Category | Emission |
|---|---|
| Cattle | 65,751 |
| Pigs | 43,846 |
| Poultry | 4,879 |
| Horses | 398 |
| Sheep and goats | 134 |
| Other animals* | 5,341 |
| Fertilizer | 28,579 |
| Total | 148,928 |

* Cats, deer, dogs, foxes, human beings, mink. This category is not taken into account in the gridded survey, as this information was only known for whole Denmark.

emission inventory is mainly based on livestock data for all municipalities in Denmark for the year 1985 and the fertilizer consumption for all provinces for the same year.

The highest detail of the emission inventory is in fact the size of a municipality (about 160 $km^2$). Using an emission inventory with the same emission density everywhere within a municipality in model calculations gives some unrealistic results. By applying this procedure unrealistically high concentrations will be calculated for areas with insignificant emissions, such as forests. It was therefore decided to distribute the emission from livestock evenly over the agricultural area with a municipality. The emission from fertilizers was distributed evenly over the agricultural area in a province. To make this possible, a gridded 1x1 $km^2$ land use map was made for Denmark, which also contains information on the area occupied by each municipality (Runge and Asman, 1989). In model calculations for Denmark a resolution of 5x5 $km^2$ was chosen for the Danish emissions and a resolution of 75x75 $km^2$ for the other European emissions. Figure 1 shows the gridded $NH_3$ emission density for Denmark.

# 3 Model results

The values of the model parameters in the TREND model are the same as used by Asman and van Jaarsveld (1990): a dry deposition velocity for $NH_3$ of $1.2 \times 10^{-1}$ m $s^{-1}$, a dry deposition velocity for the $NH_4^+$ aerosol of $1.4 \times 10^{-3}$ m $s^{-1}$ and a conversion rate of $NH_3$ to $NH_4^+$ aerosol of about 30% $h^{-1}$.

Figures 2-5 show concentrations and depositions of $NH_x$ in Denmark. The concentration pattern for $NH_3$ is clearly determined to a large extent by local sources. Concentrations over sea are therefore very low. The pattern for $NH_4^+$ aerosol and $NH_x$ is to a large extent determined by non-Danish sources (see also Table 2). About 64% of the total deposition in Denmark (land area) is dry deposition of $NH_x$ (mainly $NH_3$). Model calculations show that only 37% of the $NH_3$ emitted in Denmark is deposited as $NH_x$ in Denmark. The largest part is hence transported over long distances in the form of $NH_4^+$ aerosol.

An extensive description of the model results is given in Asman (1990C). The same kind of computations are made for nitrogen oxides ($NO_x$) and reaction products. These calculations show that about 67% of the total nitrogen (from both $NH_3$ and $NO_x$) deposition in Denmark is caused by $NH_x$ whereas this is less than 50% for the Kattegat Sea east of Denmark.

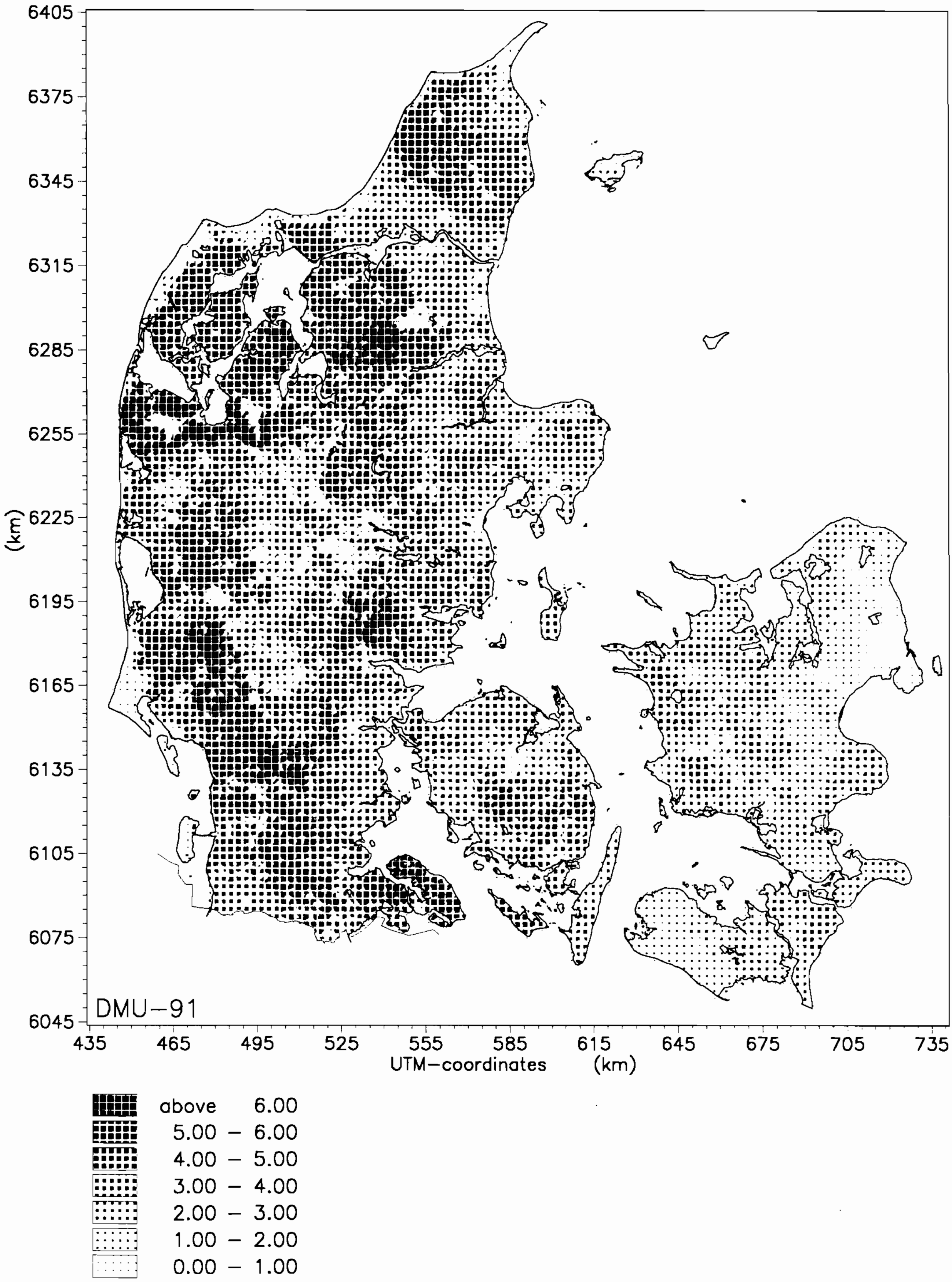


Figure 1. $NH_3$ emission density for Denmark (tonne $NH_3$ $km^{-2}$ $a^{-1}$)

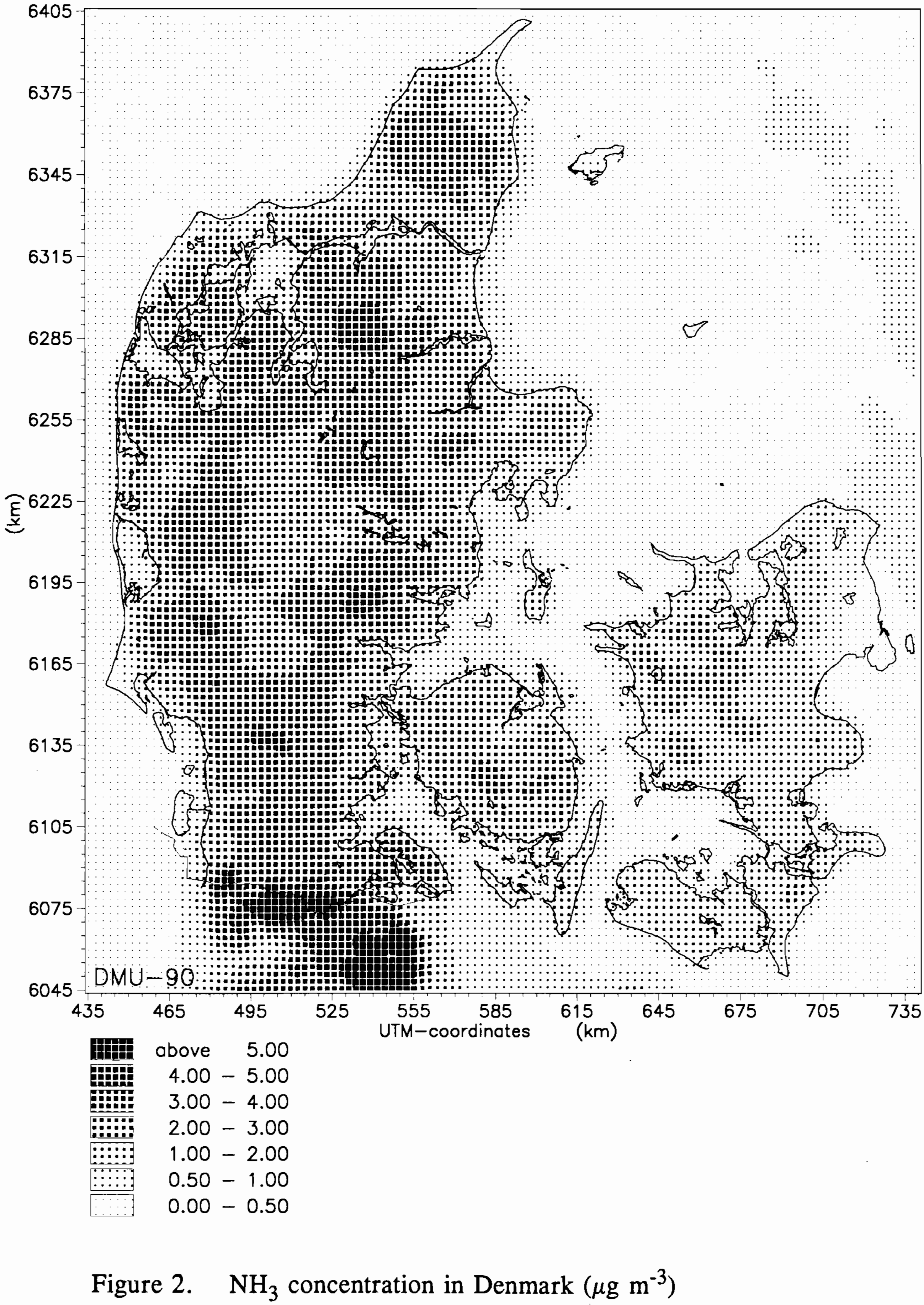


Figure 2. $NH_3$ concentration in Denmark ($\mu g\ m^{-3}$)

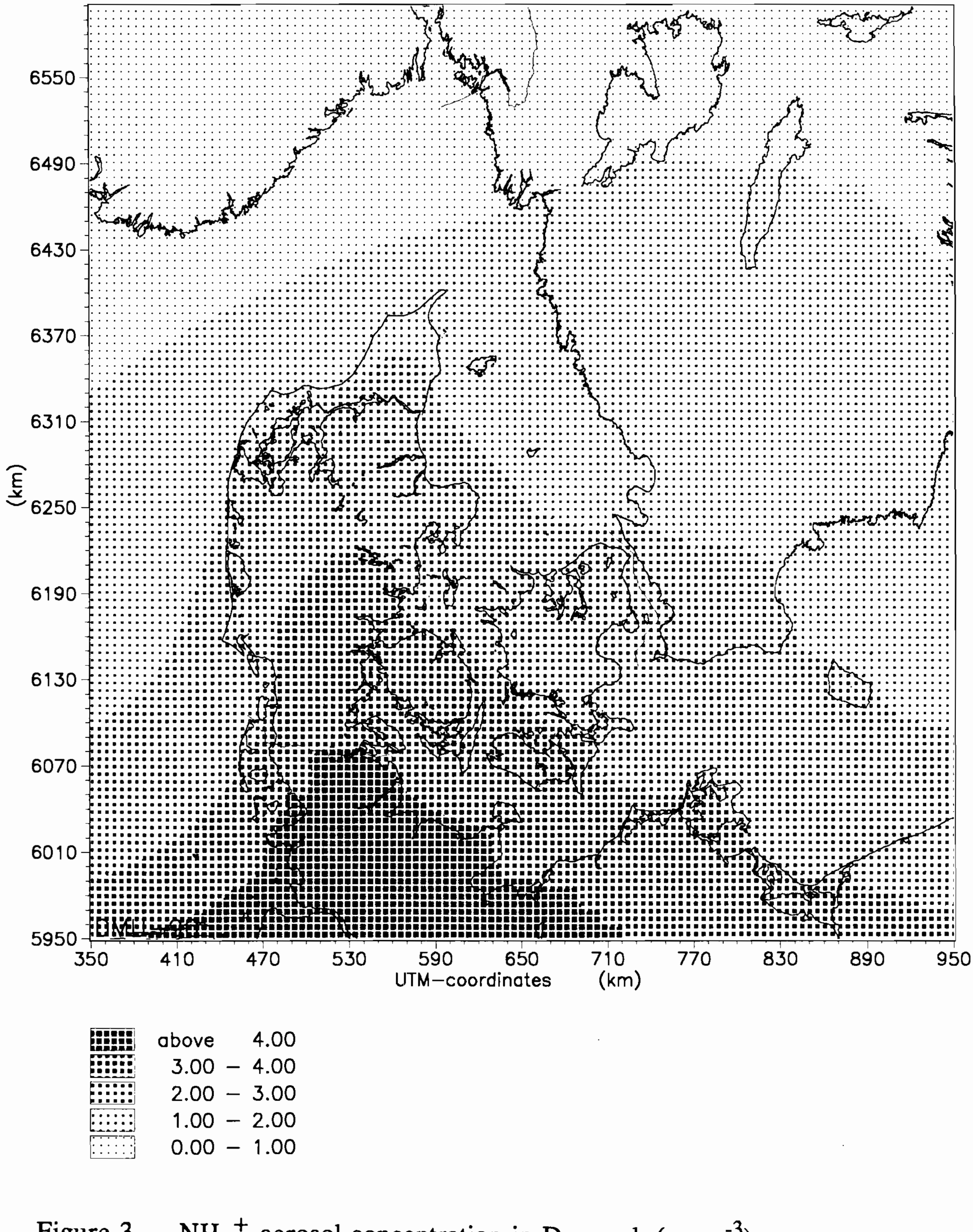


Figure 3. $NH_4^+$ aerosol concentration in Denmark ($\mu g\ m^{-3}$)

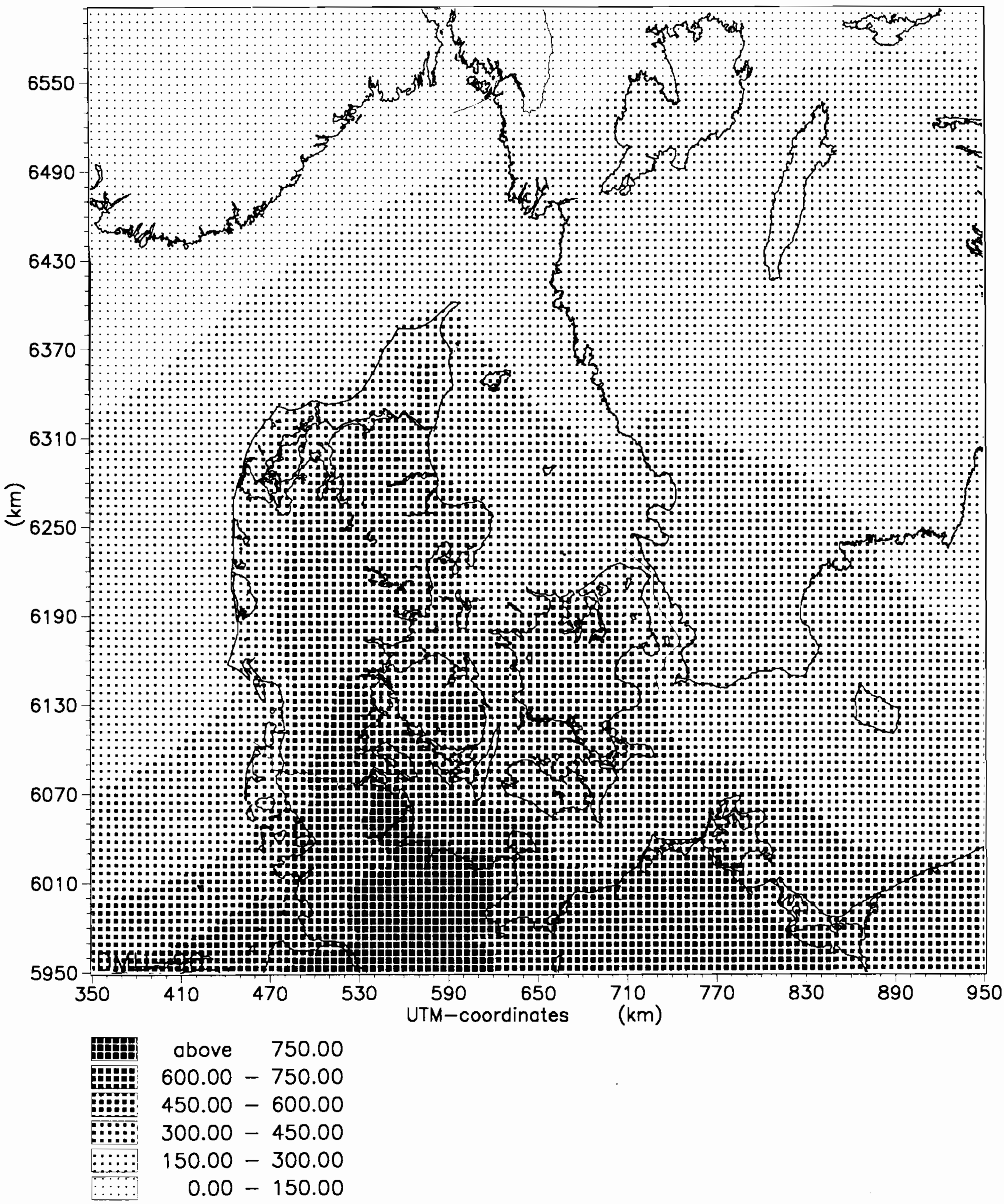


Figure 4. Wet deposition of $NH_x$ in Denmark (kg N $km^{-2}$ $a^{-1}$)

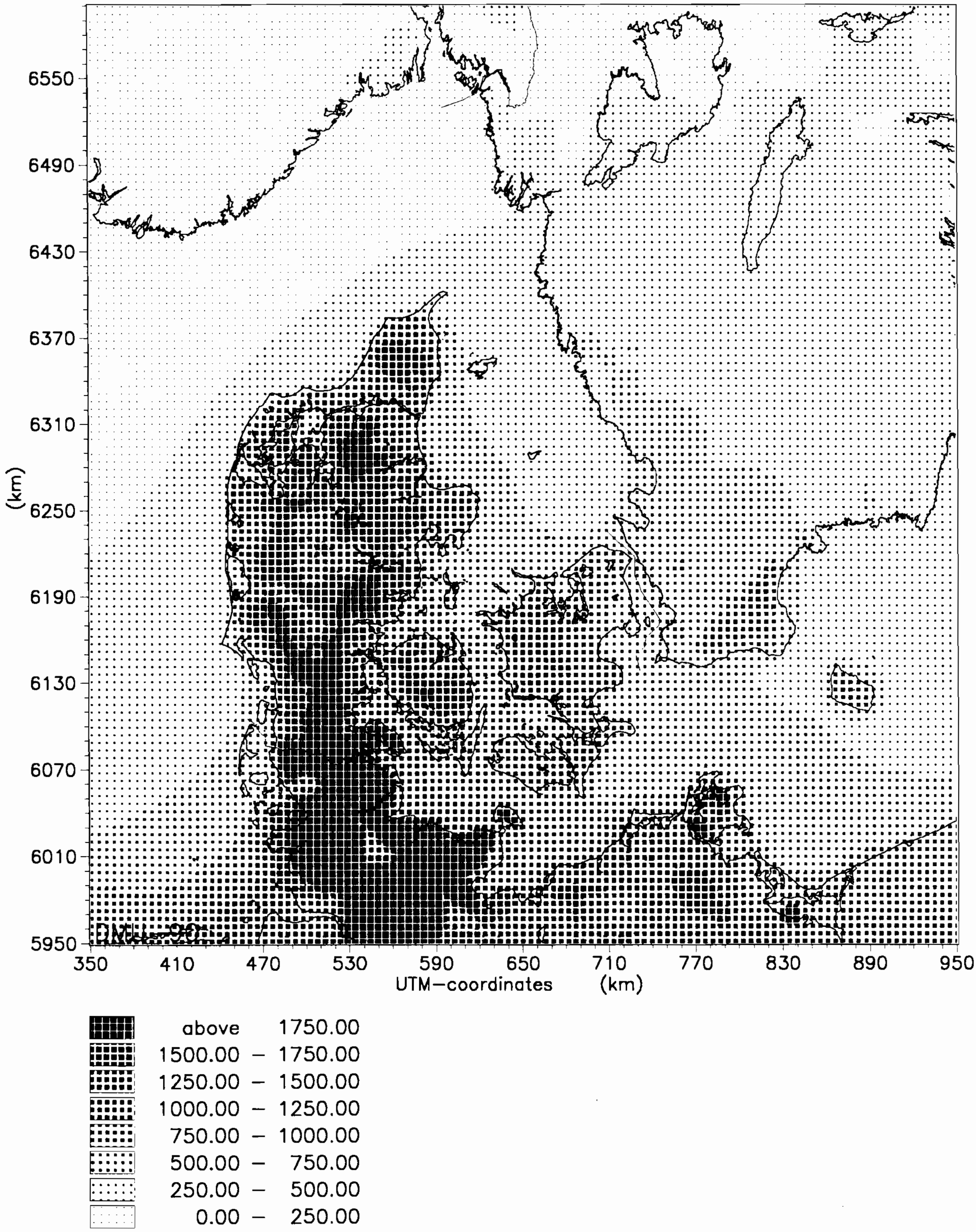


Figure 5. Total $NH_x$ deposition in Denmark (kg N $km^{-2}$ $a^{-1}$)

Table 2. Modelled concentrations and depositions in Denmark (land area) and percentage caused by Danish emissions.
(The uncertainty in the model results is at least 30 %.)

| Component | conc./dep. | % from Denmark |
|---|---|---|
| $NH_3$ concentration ($\mu g\ m^{-3}$) | 2.5 | 97 |
| $NH_4^+$ concentration ($\mu g\ m^{-3}$) | 2.8 | 47 |
| $NH_x$ dry deposition (kg N $km^{-2}\ a^{-1}$) | 883 | 91 |
| $NH_x$ wet deposition (kg N $m^{-2}\ a^{-1}$) | 497 | 46 |
| $NH_x$ total deposition (kg N $km^{-2}\ a^{-1}$) | 1,380 | 74 |


## Acknowledgement

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# DISCUSSION AND CONCLUSIONS ON EMISSION INVENTORIES AND EMISSION COEFFICIENTS FOR AMMONIA

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## 1 Introduction

Discussion during the workshop focused on three questions:

* Which sources are to be included in emission inventories?
* Which emission coefficients should be used for livestock animals and what are the most important factors that tend to cause differences in emission coefficients between countries?
* Which emission factors (N-losses) are to be used for artificial fertilizer?

This contribution summarizes the main conclusions of the discussion on emission inventories and emission coefficients for ammonia.

## 2 Sources of ammonia

The various national and international inventories for ammonia included in this volume show wide differences in the coverage of sources. The international inventories are generally restricted to livestock farming, fertilizer use and industrial sources. Table 1 shows that these sources are usually also covered by national emission inventories. In addition to livestock farming, fertilizer use and industry other anthropogenic sources, such as human respiration , wild animals and sewage sludge, in most national studies are only partially covered. Natural sources (natural soils) are also not always included. This implies that a comparison of national estimates is difficult because of the different coverage of sources.

Two potentially important emission sources, not covered by international inventories, are natural soils and sewage sludge. Regarding the emission from natural sources (i.e soils) the uncertainty in the estimates appears to be fairly large and the question remains whether such a source is to be included in an emission inventory. Since international inventories at present show good

Table 1.    Sources of Ammonia

| | DK | FIN | FRA | FRG | GDR | HUN | ITA | NET | SWI | UK |
|---|---|---|---|---|---|---|---|---|---|---|
| **Livestock** | | | | | | | | | | |
| - Cattle | x | x | x | x | x | x | x | x | x | x |
| - Pigs | x | x | x | x | x | x | x | x | x | x |
| - Poultry | x | x | x | x | x | x | - | x | x | x |
| - Sheep | x | - | x | x | x | x | x | x | x | x |
| - Horses | x | - | x | x | x | x | x | x | x | x |
| **Fertilizer** | x | x | x | x | x | x | x | x | x | x |
| **Industry** | - | x | x | x | x | x | x | x | x | x |
| **Other sources** | | | | | | | | | | |
| - Humans | x | - | - | - | x | - | - | x | x | x |
| - Wild animals | - | - | - | - | - | x | - | x | - | x |
| - Fur animals | x | x | - | - | - | - | - | x | - | - |
| - Pet animals | x | - | - | - | - | - | - | x | x | x |
| - Coal comb. | - | - | x | x | x | - | - | - | - | - |
| - Traffic | x | - | x | - | x | - | x | x | - | x |
| - Sewage sludge | x | - | - | - | - | - | x | x | - | x |
| - Landfill | x | - | - | - | - | - | x | - | - | x |
| **Natural** | x | - | x | - | x | x | x | - | x | x |

correspondence between calculated and measured deposition there is little reason to belief that natural soils are an important source of emissions. The contribution of this source is likely to remain within the general margin of error of 20 per cent of ammonia inventories. The contribution of natural soils might especially be relevant in regions were man-made emissions are fairly low. An accurate inventory should rely on measured emissions rather than general emission coefficients for natural soils taken from the literature.

Some of the national inventories, for example for the United Kingdom, suggest that another potentially relevant source might be sewage sludge. A distinction should be made between emissions from sewage treatment plants and the spreading of sewage sludge on land. In the latter case NH3 emissions are comparable with ammonia emission resulting from cattle slurry spreading. As with the emission factors for natural soils, however, the uncertainty in emission coefficients for sewage sludge is fairly large.

Other anthropogenic sources, such as the additional emission caused by cars equipped with three-way catalysts, are believed to be unimportant for national inventories although they are relevant for local studies. Instead of focusing research on other, minor sources of emissions it might be more important to focus attention on the expected seasonal variation in ammonia emissions. Until now it proved difficult to model this seasonal pattern in a satisfying way.

# 3 Emission coefficients for livestock animals

Remarkably is that only a few of the national studies included in this volume (Finland, United Kingdom and the former German Democratic Republic) rely on national, or country specific estimates for the emission coefficients for livestock animals. All other estimates either go back to data for the Netherlands on N-excretion as included in Buijsman et al. (1987) or make use of recent Dutch data on emission coefficients (Van der Hoek, 1989). The figures for the RAINS (Regional Acidification INformation and Simulation) model rely partly on the recent data for the Netherlands and partly make use of country specific elements such as the length of the meadow period for dairy cows (Klaassen, 1991). It is preferable that countries try to make use of national data as far as possible.

One way of arriving at national emission coefficients for livestock animals is to make use of the recent Dutch insights, generated with the help of nitrogen mass balances, and adapt them to reflect country specific elements. An example of such a nitrogen mass balance is given in Table 2. Following the discussion, two important factors causing differences in the nitrogen content of the excretion of dairy cows might be the milk yield per cow and the N-content of the roughage. Even within countries regional differences are expected. Calculation of N-losses from excretion is fraught with difficulties. Errors of 10 per cent in measured N percentage in the manure are possible. In addition N-losses that occur, and are calculated using a nitrogen mass balance do not necessarily have to be ammonia. In dry, poultry manure nitrogen might also be lost as N2 (de-nitrification). In liquid manure, as from cows, the loss is solely ammonia. Hence, another important element causing differences in ammonia emission is the type of manure: dry or liquid manure. Furthermore, the type of animal is important as well as the age-, weight-, and sex distribution within a specific animal category. In addition, the stall type and the ventilation type is important. Ammonia volatilization is correlated with the amount of ventilation. Results in the Netherlands show that, in especially for poultry but also for pig stables, different stable type cause significant differences in emissions of ammonia. Last but not least the composition of the manure (PH, per cent urea) is important.

Table 2. $NH_3$-N balance for dairy cows

| | Category | Specification | | | |
|---|---|---|---|---|---|
| 1 | INTAKE FEED | Type | Kg | Kg N/kg | Kg N/animal |
| | stall period | silage | 1273 | 0.0286 | 36.41 |
| | (in days): 190 | maize | 532 | 0.0136 | 7.24 |
| | | power fodder | 1090 | 0.0250 | 27.25 |
| | | subtotal | 2895 | 0.0245 | 70.89 |
| | meadow period | grass | 2415 | 0.0377 | 91.05 |
| | (in days): 175 | maize | 183 | 0.0136 | 2.49 |
| | | power fodder | 350 | 0.0250 | 8.75 |
| | | subtotal | 2948 | 0.0347 | 102.28 |
| | total | | 5843 | | 173.18 |
| 2 | RETENTION | Type | Kg | kg N/kg | kg N/animal |
| | stall period | milk | 2850 | 0.0054 | 15.39 |
| | (in days): 190 | calf | 38 | 0.0250 | 0.95 |
| | | subtotal | 2888 | 0.0250 | 16.34 |
| | meadow period | meat | 30 | 0.0250 | 0.75 |
| | (in days): 175 | milk | 3150 | 0.0054 | 17.01 |
| | | subtotal | 3180 | 0.0055 | 17.76 |
| | total | | 6068 | 0.0056 | 34.10 |
| 3 | N IN EXCRETION | | | | kg N/animal |
| | stall period | subtotal 1 - subtotal 2 | | | 54.55 |
| | meadow | subtotal 1 - subtotal 2 | | | 84.52 |
| 4 | $NH_3$ EMISSION STABLE | | | in % | kg N/animal |
| | N excretion x volatilization | | | 13.24 | 7.22 |
| | | | | | kg $NH_3$/animal |
| | $NH_3$ emission; N emission * 17/14 | | | | 8.77 |
| 5 | $NH_3$ EMISSION APPLICATION | | | | kg N/animal |
| | N in manure after stable emission | | | | 47.33 |
| | | | | in % | |
| | Of which mineral N part | | | 49.99 | 23.66 |

Table 2. $NH_3$-balance for dairy cows (contd.)

| | | | | | |
|---|---|---|---|---|---|
| 5 | $NH_3$ EMISSION APPLICATION (continued) | | | | kg N/animal |
| | $NH_3$ emission: | | | in % | |
| | Mineral part x volatilization | | | 50.00 | 11.83 |
| | | | | | kg $NH_3$/animal |
| | $NH_3$ emission; N emission * 17/14 | | | | 14.37 |
| 6 | $NH_3$ EMISSION MEADOW | | | in % | kg N/animal |
| | N excretion meadow x volatilization | | | 12.00 | 10.14 |
| | | | | | kg $NH_3$/animal |
| | $NH_3$ emission; N emission * 17/14 | | | | 12.32 |
| 7 | SUMMARY $NH_3$ EMISSION FACTORS | | | | kg $NH_3$/animal |
| | $NH_3$ stable | | | | 8.77 |
| | $NH_3$ application | | | | 14.37 |
| | $NH_3$ meadow | | | | 12.32 |

# 4 Emission factors for nitrogen fertilizer

Not only do ammonia emission coefficients for livestock animals show, or should show, differences among countries. Table 3 indicates that the ammonia emission for artificial fertilizer, are also less certain as previously believed. The data generally relied upon in national inventories go back to Buijsman et al. (1987). The figures included in Buijsman et al. (1987) should be seen as maximum values. Recent research results show different and more reliable estimates. The results suggests that the difference in N-loss between ammonium nitrate and calcium ammonium nitrate is not as important as previously thought. The N-loss for both types of fertilizer is in the order of magnitude of 5 per cent. Over against that, the losses reported for ammonium phosphate seem to depend on the type of ammonium phosphate: bi- or mono phosphate. For both types losses are smaller than 10 per cent. Research for the United Kingdom (Whitehead, Hurley research station) reports a loss of 3.4 per cent for ammonium mono phosphate. A problem is that international reports on fertilizer use (FAO) do not give the quantities of the different types of ammonium phosphate. The N-loss of urea is believed to be among the highest since it is an ammonium carbonate solution which keeps the PH very high.

Finally, one should not forget that the ammonia losses of fertilizers not only depend on the type of fertilizer but also on temperature and soil type.

Table 3. Emission factors for N-fertilizer (% loss of N-content)

| | RAINS (1990) | Graf (1991) | Buijsman et. al. (1987) | Asman (1990) |
|---|---|---|---|---|
| Ammonium sulphate | 15 | 10 | 15 | 15 |
| Ammonium nitrate | 10 | 0.5 | 10 | 10 |
| Ammonium sulphate nitrate | 12.5 | - | - | 12.5 |
| Calcium ammonium nitrate | 2 | - | 2 | - |
| Urea | 10 | 3.4 | 10 | 10 |
| Ammonium phosphate | 5 | - | 5 | 5 |
| Other nitrogen fert. | 1 | - | 1 | 5 |
| Other complex fert. | 1 | 0.5 | 1 | 5 |
| Other not specified | 1 | - | 1 | 5 |

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# Part II. OPTIONS FOR AND COSTS OF CONTROLLING AMMONIA EMISSIONS

Klaassen, G. (Editor) (1992). *Ammonia emissions in Europe: emission coefficients and abatement costs.* Proceedings of a workshop held 4—6 February 1991, IIASA, A—2361 Laxenburg, Austria.

# COSTS OF CONTROLLING AMMONIA EMISSION IN THE DUTCH LIVESTOCK SECTOR

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## Abstract

The Dutch government aims at reducing the ammonia emission with 50 to 70 per cent in the year 2000 compared with 1986. The abatement possibilities are 1) reducing the nitrogen excretion by adapting the compound feed for pigs and poultry and by lowering the nitrogen doses on grassland; 2) adapting the stables; 3) adapting the application methods for manure and 4) covering the manure storage. The most efficient method is at this moment adapting the application methods for manure. The objective of the government is within reach. A 60 per cent reduction costs about 400 million and an abatement of 70 per cent costs about 1100 million Dutch guilders. The composition of the Dutch livestock and the Dutch soil types differ considerably from Europe. Hence the results are only partly relevant for other countries.

# 1 Introduction

## 1.1 Scope of the paper

Ammonia is one of the components of acid rain. It has been estimated that the livestock sector is responsible for about 90 per cent of the ammonia emission in the Netherlands (Anonymous, 1990a). This percentage is only slightly higher than the estimation of 85 per cent for Europe by Klaassen (1990). It will be obvious that only a reduction of ammonia emission from livestock will decrease the share of ammonia in the acidification.

This paper elaborates the possibilities of reducing the ammonia emission in the Netherlands. The aim of this paper is to analyze the efficiency of various abatement measures in terms of emission reduction and costs. The efficiency will be given at farm level as well as at national level. The possibilities of reducing ammonia emission can be compared with the policy objective of the Dutch government. The objective is to decrease the emission in 2000 with at least 50 per cent and a reduction of 70 per cent is pursued (Anonymous, 1990b).

The Dutch agriculture is very intensive. Some aspects of the livestock sector are mentioned in section 1.2 to enable a comparison with other countries. Section 2 deals with the method and materials. Besides this an overview of the level of ammonia emission subdivided to source and to animal species is given. The results of the analysis will be discussed in section 3. This paper ends with discussion and conclusions.

## 1.2 Dutch livestock in Europe

To interpret the focus and results of this study some background information about the position of the Dutch livestock sector within Europe is necessary. In Table 1 the livestock numbers and the land use in Europe and the Netherlands are given. The Netherlands have a share of about 1 per cent of the acreage arable land and about 1.5 per cent of the permanent grassland in (Western) Europe. The shares of the livestock numbers, especially pigs and poultry, are rather high compared with the acreage of agricultural land. It will be clear that the livestock sector is relatively important in the Netherlands. Furthermore the number of pigs and poultry grew more in the Netherlands than in (Western) Europe. The possibilities of abating ammonia emission depend on the distribution between the different animal species. The Dutch agriculture differs considerably from other countries, but some smaller regions are more or less comparable, for instance the Po valley in Italy and Brittany in France.

# 2 Method and materials

## 2.1 Introduction

Because ammonia volatilizes from manure the reduction methods are strongly related with the possibilities of handling the manure. As shown in Figure 1 emissions of ammonia will therefore occur at the following places:

- in the stable and during storage of manure;
- during the application of manure;
- in the meadow period of grazing animals.

Table 1. Land use and livestock numbers in Europe, Western Europe and the Netherlands

| | Country | | | Netherlands in percentages of | |
|---|---|---|---|---|---|
| | Europe | Western Europe | Netherlands | Europe | Western Europe |
| **Land use (1000 hectare in 1988)** | | | | | |
| arable land | 126,139 | 82,422 | 902[1] | 0.7 | 1.1 |
| perm. pasture | 83,348 | 68,359 | 1,081 | 1.3 | 1.6 |
| **Cattle (1000 head)** | | | | | |
| 1979-81 | 133,377 | 99,720 | 5,071 | 3.8 | 5.1 |
| 1989 | 125,569 | 92,875 | 4,606 | 3.7 | 5.0 |
| index (79-81 = 100) | 94 | 93 | 91 | | |
| **Pigs (1000 head)** | | | | | |
| 1979-81 | 173,389 | 109,971 | 10,058 | 5.8 | 9.1 |
| 1989 | 185,925 | 119,196 | 13,730[2] | 7.4 | 11.5 |
| index (79-81 = 100) | 107 | 108 | 137 | | |
| **Chickens (million head)** | | | | | |
| 1979-81 | 1,223 | 855 | 81 | 6.6 | 9.5 |
| 1989 | 1,291 | 890 | 90[2] | 7.0 | 10.1 |
| index (79-81 = 100) | 106 | 104 | 111 | | |

Source: FAO (1990b)
1 FAO estimate
2 Official figure. Source: Centraal Bureau voor de Statistiek (1990)

These places will be called sources. The abatement possibilities of reducing the emission will be discussed in section 2.5. The livestock numbers are very important multipliers, because they largely determine the quantity of the manure produced. This aspect is not mentioned in Figure 1. The 'autonomous' development of livestock numbers till the year 2000, influenced by market forces, is given in section 2.4. Another possibility of decreasing the total emission of ammonia is reducing the quantity of the manure. Reducing the number of animals is one possibility. This option will be neglected in this paper. A lower level of ammonia emission can also be reached by adapting the nitrogen content of the feeding-stuffs, this gives a smaller nitrogen content in the manure. This influences the emission at the three sources. Section 2.5.1 deals with this possibility. Before discussing the possibilities and their efficiency, the method for calculating the emission is given first with an overview of the emission in 1986.

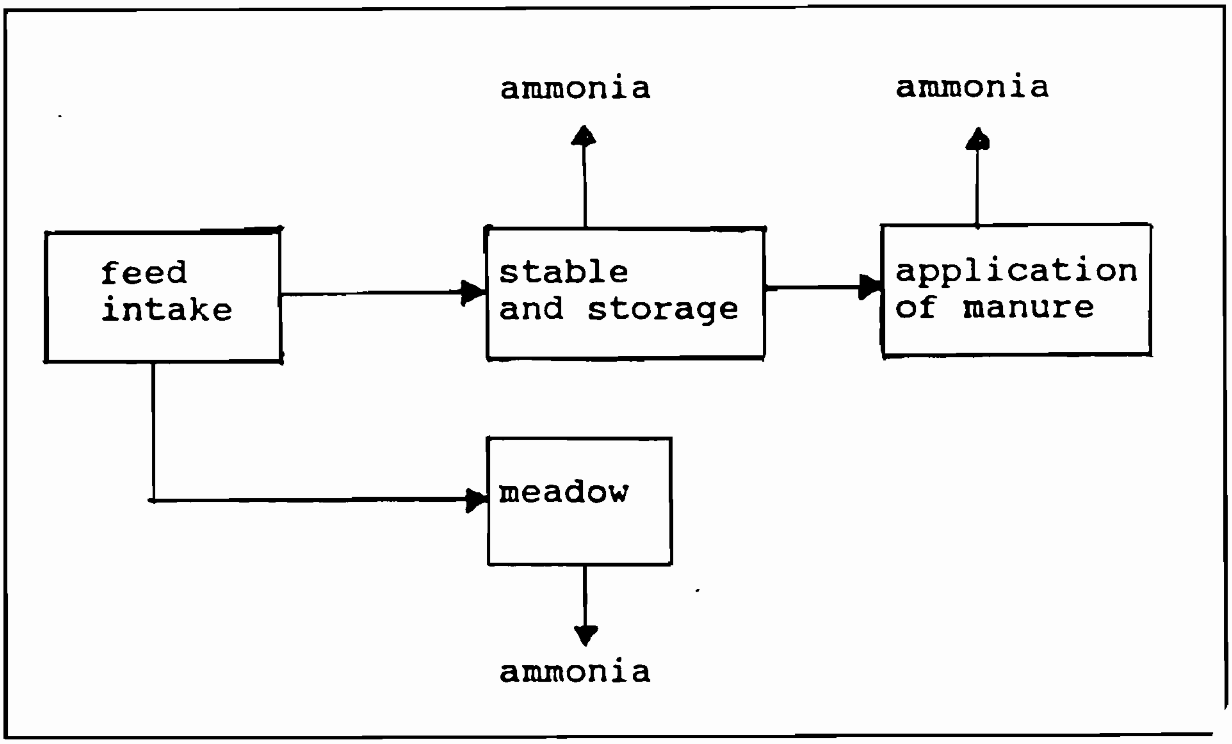


Figure 1. Ammonia losses from livestock farming (source: Klaassen, 1990)

## 2.2 Method

A model has been built to calculate the level of ammonia emission at regional and national level. The different abatement possibilities of reducing the emission of ammonia influence this level (Oudendag en Wijnands, 1989; Oudendag, 1990). This model can be written as the following equation:

$$E_{drl} = \sum_{k=1}^{K} e_{kl} * p_{kl} * dier_{dr}$$

d — animal species; the following species are taken into account: dairy-cows; beef cattle; other grazing animals; fattening calves, fattening pigs; sows; laying hens and broilers;

$dier_{dr}$ — number of animals or manure production for the different animal species (d) and regions (r);

$e_k$ — emission coefficient for each abatement possibility. This coefficient also depends on the animal species.

$E_{drl}$ — emission for livestock species d in region r and for source l;

k — abatement possibilities (see section 2.5), k= 1,K;

l — source of emission. The sources are: 1) stable and storage; 2) the meadow period when animals are grazing and 3) the application period of manure;

$p_k$ — share of abatement possibility k. The sum of all shares is equal to one;

r — regions; the Netherlands is divided into 31 regions. The regions have been classified according to the number of livestock per hectare.

Almost identical formulas have been used to calculate the costs and investments. The emission coefficient $e_k$ and the level of emission $E_{drl}$ have to be changed by a cost or investment coefficient respectively the level of costs or investments. In fact the model is built up with these three formulas. This model will give the level of ammonia emission for a source of a certain animal species in a specific region. By changing an abatement possibility one can analyze the contribution to the reduction of the ammonia emission and its costs of this particular possibility.

By introducing regions in the model the location of the emission source is important. It is clear that the emissions from the stable and also during the meadow period of the grazing animals volatilize at the same locations as the animals are located. The manure however can be applied in another region and therefore the ammonia also volatilizes in that region. Several regions in the Netherlands, especially the sandy soil regions in the south-eastern part of the country, have a rather high number of animals per hectare. This results in manure surpluses, which have to be transported to other regions, for instance a region with less livestock and a high percentage of arable land. The quantity of manure which can be applied at the farm or region where it is produced depends on the application standard for manure and the level of manure production per hectare. The application standard in the Netherlands will be more strict before the end of this decade. The location of the application of the manure is derived from a model which analyzes the optimal transport and treatment possibilities of manure(surpluses) in the Netherlands. This model has been described in Wijnands et al. (1988) and in Luesink and van der Veen (1989).

At farm level comparable formulas have been used for poultry, fattening calves and beef cattle. For dairy and pig farms linear programming models have been developed. The objective of these models is maximization of the farms results, subject to a certain level of ammonia emission and the abatement methods, which are mentioned in section 2.5. For several levels of ammonia emission the farm results will be calculated. The differences in farm results are the costs for the emission abatement (Baltussen et al, 1990a).

## 2.3 Emission of ammonia in 1986 in the Netherlands

As already mentioned in section 1.2 the composition of the Dutch livestock differs considerably from the neighboring countries of the Netherlands. First an overview of the level of emission of ammonia will be given, because knowledge of which sources and animal species are the greatest contributors to the emission of ammonia is important. From this information the relative importance of the abatement possibilities is derived. In 1986 the level of ammonia emission in the Netherlands was about 224 million kg ammonia ($NH_3$). This level is about 2 per cent higher than the level in 1980. The reduction objective of the government has been based on the emission level of 1980. In Table 2 the percentages for the animal species and sources are given.

Application of manure is the major source of ammonia emission, followed by the stable. Possibilities of reducing the emission during the application of manure are very important, because this is the largest source (see Table 2). Despite the high share of the pigs and poultry in the total number of livestock

Table 2. Ammonia emission in percentages (100 = 224 million kg) in 1986 in the Netherlands.

| Animal species | Emission source | | | | |
|---|---|---|---|---|---|
| | Stable | Storage | Meadow | Applicat. | Total |
| Dairy cows | 17.4 | 0.0 | 9.4 | 24.1 | 50.9 |
| Beef cattle | 0.9 | 0.0 | 0.0 | 1.8 | 2.7 |
| Other grazing animals | 0.4 | 0.0 | 0.4 | 0.4 | 1.3 |
| Fattening calves | 0.4 | 0.0 | 0.0 | 1.3 | 1.8 |
| Fattening pigs | 9.8 | 0.0 | 0.0 | 14.3 | 24.1 |
| Sows | 4.9 | 0.0 | 0.0 | 5.8 | 10.7 |
| Laying hens | 1.9 | 0.4 | 0.0 | 3.6 | 5.8 |
| Broilers | 0.9 | 0.4 | 0.0 | 1.3 | 2.7 |
| Total | 36.7 | 0.9 | 9.8 | 52.7 | 100.0 |

Source: Oudendag (1990)

compared with other European countries the dairy cows contribute more than 50 per cent. The pigs and sows have a share of 35 per cent and the chickens of 8.5 per cent. The other animal species are unimportant. The conclusions from Table 2 are:

- emission reduction during the application of manure is very important;
- cattle contributes more than half to the total emission;
- a complete reduction of the emission from the poultry and pig stables means a reduction of 17.5 per cent at maximum.

## 2.4 Development of livestock numbers

The government objective should be attained in the year 2000. As shown in Table 1 the numbers of livestock do not always remain at the same level. The numbers of livestock are very important multipliers. The efforts of reducing ammonia emission can greatly be undone by an increase of the number of livestock. The opposite to this - a decreasing number of animals - simplifies attaining the objective. To make an estimation of the level of ammonia emission in the year 2000 assumptions about the numbers of livestock have to be made. In a survey made by Douw et al. (1987) the situation of the agriculture in the year 2000 has been estimated. Firstly, because of an increasing productivity the same amount of products can be produced with fewer animals or fewer inputs (for instance feeding-stuffs). The milk production per dairy cow, for instance, will increase annually with about 1.5 per cent. Secondly the food consumption per head of the population will increase more slowly than in the past because it is reaching its saturation point. Thirdly the population of the Western countries stabilizes. In the past the production of agricultural products grew faster than the consumption. Well known are the surpluses of agricultural commodities in the European Community, which for instance resulted in the introduction of the milk quota system. In Table 3 an estimation of the numbers of livestock in 2000 compared with 1986 has been made. With this size of livestock the Netherlands can meet the (domestic and foreign) demand for Dutch livestock products.

Table 3. Estimated numbers of livestock in 2000 in percentages of the number in 1986.

| Animal species | Percentages |
|---|---|
| Dairy cows | 75 |
| Other cattle | 128 |
| Fattening calves | 80 |
| Fattening pigs | 110 |
| Sows | 105 |
| Laying hens | 75 |
| Broilers | 107 |

## 2.5 Abatement possibilities

### 2.5.1 Feeding-stuffs

First a decrease of the excretion of nitrogen by animals will be discussed because it influences the overall emission. The excretion depends largely on the input of nitrogen in the feed.

Pigs and poultry get almost only compound feed which comes from outside the farm. Moreover, these feeding-stuffs are mainly based on imported commodities. The nitrogen contents of the animals menu can be adapted. The first possibility is a better tuning of the compound feed to the nutrients needs of the animals. The nutrient need of a fattening pig, for instance, depends on the fattening stage. At the start of the fattening period the need for protein is higher than the need in the last weeks before it will be slaughtered. This means that the animals get several types of compound feed during their life. This method is called stage feeding. The second method is that no overdoses of nutrients are given. The price of the compound will rise because relatively cheap commodities will be substituted by more expensive ones. Because the supply of some specific commodities is limited the reduction of the protein content in the compound feed will be less than can be calculated for each separate animal species. This restriction has taken into account. In Table 4 the possibilities are summarized. The decrease in N-excretion differs between 5 for sows and 22 per cent for fattening pigs. A decrease with 5 per cent for sows for instance will also result in a decrease of ammonia emission in the stable and during the application with the same percentages. The increase of costs is relatively small.

Table 4. The N-excretion (kg N) per animal on year base and costs (HFL = Dutch guilders) per 100 kg compound feed.

| | Sows | Pigs | Hens | Broilers |
|---|---|---|---|---|
| N-excretion | | | | |
| - Basis (2000)[1] | 36.26 | 13.44 | 0.65 | 0.37 |
| - Stage feeding | 36.26 | 12.97 | [2] | 0.33 |
| - Stage feeding + decrease protein contents | | | | |
| in kg | 34.45 | 10.48 | 0.60 | 0.30 |
| in percentage basis | 95 | 78 | 92 | 81 |
| Costs | | | | |
| - Basis (2000)[1] | 48.95 | 47.10 | 55.00 | 68.00 |
| - Stage feeding | 48.95 | 47.14 | [2] | 68.00 |
| - Stage feeding + decrease protein contents | | | | |
| in guilders | 50.20 | 47.39 | 55.55 | 69.02 |
| in percentage basis | 103 | 101 | 101 | 102 |

Source: Baltussen et al. 1990a

1) The objective of the government should be reached in the year 2000. The results will presented for that year. Therefore the estimated needs of the animals in 2000 are used.

2) This possibility does not apply here.

The ammonia emission from cattle greatly depends on the use of organic and inorganic nitrogen fertilizer. The Dutch agriculture uses twice or three times as much nitrogen per hectare agricultural land compared with other countries. Even if it is compared with countries like Germany, Denmark or the United Kingdom, which have also a highly developed agriculture (FAO, 1990a). Along with the major part of the nitrogen in the manure the average Dutch dairy farm uses about 328 kg nitrogen per hectare grassland from chemical fertilizer (van Dijk en van Vliet, 1990). A decrease of this use will result in a decrease of the nitrogen (and therefore protein) content of the roughage from grass. Baltussen et al. (1990c, page 103-105) estimated that a decrease of the nitrogen application from 400 to 200 kg N per hectare grassland results in 11 to 30 per cent smaller ammonia emission for the cattle. The costs are very moderate (less than 1 per cent of the average production costs) or even negative, which means that the income will increase. The results are strongly related with the farming system (for instance the grazing system, the number of cows per hectare etc.). The ammonia emission from cattle is about 55 per cent of the total. The maximum reduction of the total emission will therefore be between 5 and 15 per cent.

Whether the results of adapting the compound feed or decreasing the nitrogen use per hectare are valid for other countries depends on the farming systems. If compound feed form the major part of the feed input for pigs and poultry, comparable results should be possible. Because of the high level of nitrogen use in the Netherlands a decrease in the use of nitrogen fertilizer in other countries has only a theoretical value. Even a level of 200 kg N per hectare is rather high for several countries.

### 2.5.2 Application of manure

Improved application methods for manure are the most promising measures for reaching a reduction of the ammonia emission in the Netherlands. In Table 5 the possibilities are summarized. Major reductions up to 90 per cent can be reached by ploughing down the manure directly after spreading or by injecting the manure. The costs of the mentioned methods are rather moderate. Also the quantity of nutrients available for the crops and the efficiency of the nitrogen in manure will increase. The savings on inorganic fertilizer can be 1 to 1.5 guilder per $m^3$ pig and cattle slurry. Injection is possible on sandy soils. Sod manuring has been developed for peat-soils. Sprinkling and spreading diluted manure can be applied in the areas where enough water is available for the agriculture and in flat areas because of running off of manure. It will be obvious that injection and sod manuring are only possible on soils without stones or other obstacles. In other countries than the Netherlands these methods can therefore only be applied on rather small areas. Table 2 and 5 show that a major reduction of the emission can be realized. Because more than 50 percent of the emission occurs from application, the total emission can decrease with at least 40 per cent.

Table 5. Ammonia emission for improved application methods and additional costs (HFL).

| Method | Variable costs/m$^3$ | Fixed cost/farm | Emission per cent |
|---|---|---|---|
| Surface spreading | 0 | 0 | 50 |
| Grassland | | | |
| - injection | 4.00 | 0 | 5 |
| - sod manuring | 4.00 | 0 | 10 |
| - sprinkling | 3.00 | 4,410 | 15 |
| - applying diluted manure | 0 | 7,540 | 15 |
| Arable land | | | |
| - ploughing down within 1 day | 0.65 | 0 | 35 |
| - ploughing down at once | 3.25 | 0 | 5 |

Source: Baltussen et al. (1990a).

### 2.5.3 Housing and storage

To attain the government's objective of a reduction of 70 per cent in the year 2000 a smaller emission from the stable will be necessary. At this moment a lot of research is going on to develop methods to reduce emission from the stables. No figures from practical experience are yet available. Therefore the figures and methods presented in Table 6 are tentative.

Air filtration, which is aiming at removing the ammonia from the ventilation air, is very expensive. Mostly the costs are higher than the labor incomes of the farmers. Despite the high percentages of emission reduction the effect on the total emission will be rather low; only 18 per cent of the total emission comes from housing pigs and poultry.

Table 6. A tentative survey of reduction methods of housing animals.

| Animal species | Method | Emission reduction per cent | Additional costs HFL/animal/year | Investment HFL/animal place |
|---|---|---|---|---|
| Dairy cows | flushing | 50 | 100 - 150 | 500 - 750 |
| Fatt. calves | air filtration | 80 | 100 - 140 | 291 - 407 |
| Fatt. pigs | small adaptation[1] | 25 | 2 | 10 |
| | big adaptation[2] | 50 | 15 | 75 |
| | air filtration | 70 - 90 | 56 - 73 | 163 - 375 |
| Sows | small adaptation[1] | 25 | 10 | 50 |
| | big adaptation[2] | 50 | 50 | 250 |
| | air filtration | 40 - 90 | 84 - 210 | 1244 - 1089 |
| Laying hens | air filtration | 80 | 3.58 | 10.60 |
| Broilers | floor heating | 23 | 0.13 - 0.20 | 2.50 |
| | air filtration | 80 | 3.58 | 10.60 |

Source: Baltussen, et al., 1990a
1) for instance the use of a siphon-trap.
2) for instance flushing manure.

The leaching of nitrate to the groundwater is another problem, besides ammonia emission and the surplus problem, which is related to animal manure. Leaching can be prevented by spreading manure at the start of the growing period of the crops. The storage period and storage capacity have to be enlarged. The additional storage will be built next to the stables and only a minor part of the manure will be stored in it. Hence the emission of ammonia from storage will increase. In the year 2000 the emission from storage will therefore be higher than in 1986 as mentioned in Table 2. By covering this additional storage the ammonia emission will decrease with 90 per cent. The annual costs depend on the size of storage. They are between 2 and 6 guilders per $m^3$ and the investment costs are between 23 and 66 guilders.

### 2.5.4 Grazing period of cattle

During the grazing period cattle contributes about 10 per cent to the total ammonia emission. A lower use of nitrogen will reduce this emission. This option is already described in section 2.5.1. By keeping the cattle in the stable during the whole year, the only other possibility, ammonia emission by cattle in the meadow will be zero. It also will be obvious that the emission in the stable and during the application of manure will increase. Oudendag and

Wijnands (1989) estimated an increase of the overall emission with 15 per cent. We can conclude that housing the cattle the whole year is not recommendable.

# 3 Results of abatement of emission and costs

## 3.1 Results at national and regional level

The government objective is a reduction of ammonia emission of 50 to 70 per cent compared with the level of 1980 (the level of 1986 is slightly higher than 1980). This applies for the emission of all animals together. So if one animal species can not reach this objective, the other species have to reach a higher level. In Table 7 the results at national level are given for separate abatement methods. Adaptation of the application method, which reduces the emission with 46 per cent, is relatively cheap. Adaptation of the feeding-stuff (and nitrogen dose per hectare) for all animal species is second best. For grazing cattle the nitrogen use per hectare has to be 200 kg instead of 400. A maximum adaptation of the stable has a relatively small effect. An emission reduction in the stable means a larger concentration of ammonia in the manure. Half the reduction will volatilize during the application. So it is recommendable to use advanced application methods before adapting the stables. Covering the storage has very little influence because the share in the total emission is also very low. Air filtration is very expensive and is therefore not recommendable. The total costs are about 1,200 million guilders.

Table 7. Reduction of emission (in percentage of the level of 1986), additional costs in million HFL and additional cost per per cent emission reduction in million HFL/per cent reduction.

| Method | Reduction | Costs | Cost per % |
|---|---|---|---|
| Emission low application | 46 | 84 | 2 |
| Adaptation feeding | 28 | 300 | 11 |
| Adaptation stables | 18 | 611 | 34 |
| Covering storage | 1 | 87 | 87 |
| Air filtration | 16 | 1,174 | 73 |

Source: Oudendag, 1990.

It is not allowed to add the separate effects, which are presented in Table 7. An adaptation of the feeding results in a lower nitrogen excretion and therefore in a lower emission in the stable, during the application of manure and in the meadow for grazing animals. Two alternatives have been composed in which these interactions were taken into account. Because of the autonomous development (see section 2.4) of the numbers of livestock and of some minor adaptations in the field of the feeding the ammonia emission reduces with about 6 per cent. In the first alternative the autonomous development has been combined with emission low application methods and the adaptation of the feeding-stuffs. The ammonia emission decreases with another 51 per cent, compared with 1986 in total with 57 per cent. This is a smaller figure than the adding up of the two figures in Table 7. In the second alternative the first alternative has been combined with the maximum adaptation of the stables and the manure storage will be covered. Then the total reduction of emission will be 69 per cent. The costs and the investments are summarized in Table 8.

Table 8. Annual costs and investments in million HFL for two alternatives

| Method | Alternative 1 | | Alternative 2 | |
|---|---|---|---|---|
| | Costs | Investment | Costs | Investment |
| Application | 84 | 0 | 84 | 0 |
| Adaptation feeding | 300 | 135 | 300 | 135 |
| Adaptation stables | 0 | 0 | 600 | 2,500 |
| Covering storage | 0 | 0 | 90 | 920 |
| Total | 384 | 135 | 1,074 | 3,555 |

Source: Oudendag (1990)

The emission of ammonia for some regions is shown in Figure 2. The Peel region has high livestock numbers per hectare and the Flevopolders have low numbers. Despite a decrease of about 70 percent the absolute level in the Peel region is almost as high as the average in the Netherlands in 1986. In the Flevopolders the decrease is rather low, because a larger quantity of manure from the surplus region, which will be transported to these Polders, has to be applied.

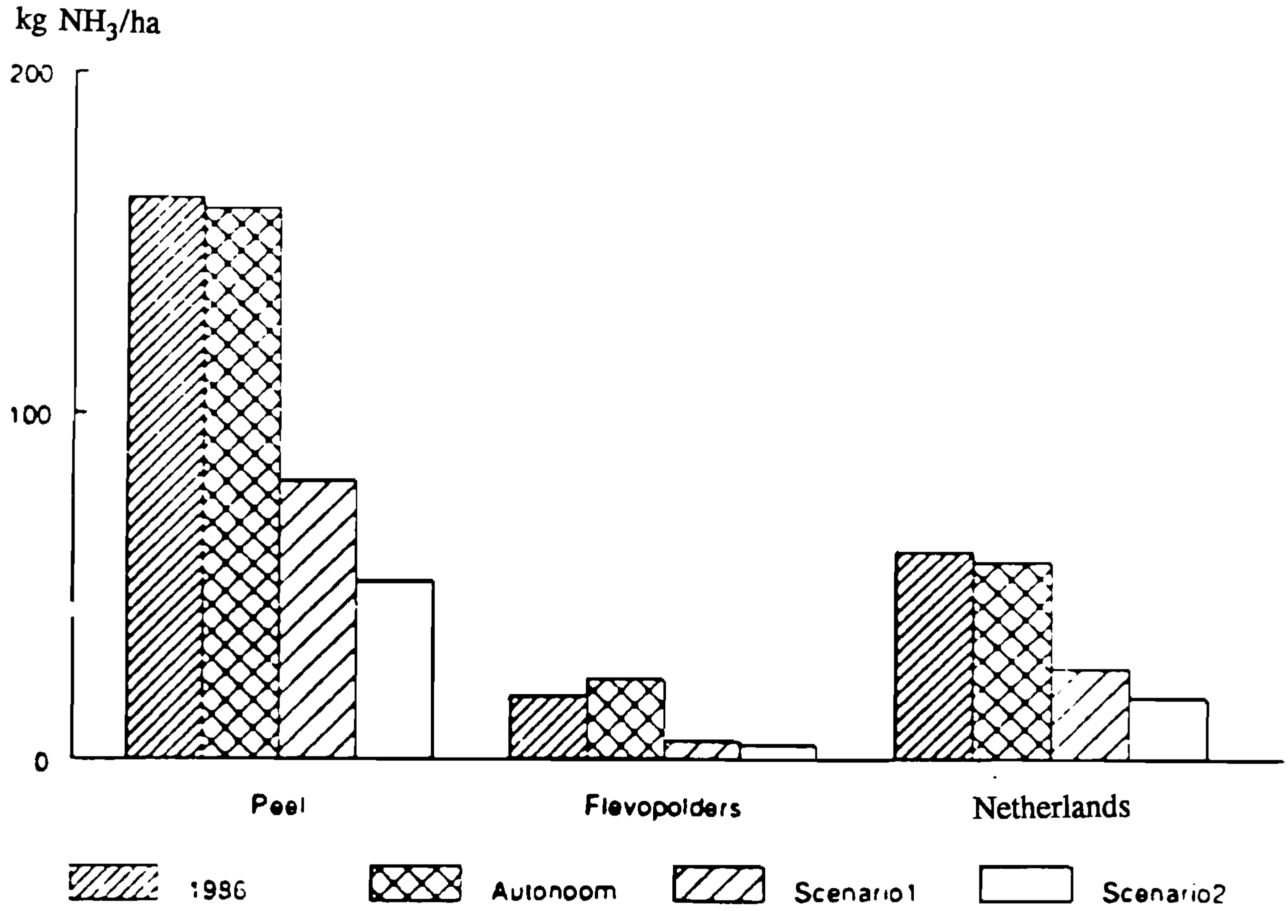


Figure 2. Ammonia emission in kg per hectare in 4 regions and in the Netherlands. Source: Oudendag (1990)

## 3.2 Results at farm level

In this section the results of applying the abatement possibilities (see section 2.5) for dairy farms and for fattening pig farms will be presented. All dairy cows together are the largest contributors to the emission of ammonia, followed by pigs (see Table 2). In Table 9 the results for a dairy farm with 2.4 dairy cow per hectare on sandy soils are summarized. Emission low application is the first abatement method because of its high efficiency and farmers are already using it at farm level. It is shown that a reduction of more than 50 per cent is within reach. To obtain this reduction the nitrogen level per hectare has to be lowered with 200 kg (method 3) or a stable adaptation - which decreases the stable emission with 50 per cent - will be necessary (method 4). The additional costs can be compared with the production costs (costs of labor excluded) of about 2400 guilders or with the labor income of the farmer which amounts to about 600 guilders per dairy cow (replacements included). A reduction of 50 per cent increases the production costs with 7.5 per cent or decreases the labor income of the farmer with about 30 per cent. It is also shown that the additional costs are much lower for the first percentages of reduction than for the last percentages. Compare, for instance, the results of method 1 and the difference between method 3 and 5 or 4 and 5.

Table 9. Reduction of emission (in percentage of 1986), ammonia emission and additional annual costs for a dairy farm

| Method | Reduction (%) | Emission (kg) | Costs (HFL/cow) |
|---|---|---|---|
| 1986 (basis) | 0 | 38.3 | 0 |
| 2000 (autonomous development) | - 2 | 38.9 | 0 |
| 1. Emission low application | 39 | 23.2 | 43 |
| 2. Method 1, 300 kg N per hectare + covering manure storage | 45 | 20.9 | 74 |
| 3. Method 2 but 200 kg N per hectare | 50 | 19.3 | 180 |
| 4. Method 2 + stable adaptation | 61 | 15.0 | 184 |
| 5. Method 4 + 200 kg N per hectare | 64 | 13.7 | 290 |

Source: Baltussen et al 1990a

The possibilities of reducing emission at the dairy farm give a very good impression for all types of cattle.

The second farm type - fattening pig farm - gives an indication of the possibilities for pigs and poultry sector. The results are presented in Table 10.

Table 10. Reduction of emission (in percentage of 1986), ammonia emission and additional annual costs per fattening pig place

| Method | Reduction (%) | Emission (kg) | Costs (HFL/ fattening pig place) |
|---|---|---|---|
| 1986 (basis) | 0 | 5.40 | 0 |
| 2000 (autonomous development) | 2 | 5.32 | 0 |
| 1. Emission low application | 51 | 2.67 | 2.49 |
| 2. Method 1 + adaptation feeding[1] | 61 | 2.09 | 4.75 |
| 3. Method 2, small stable adaptation + covering storage | 70 | 1.61 | 7.42 |
| 4. Method 3, big stable adaptation + 50 % central manure processing[2] | 82 | 0.97 | 25.40 |
| 5. Method 3, air filtration + 50 % central manure processing[2] | 91 | 0.51 | 68.40 |

Source: Baltussen et al, 1990a.

1) stage feeding and a decrease of the protein contents (see Table 4).

2) 50 % central manure processing means that half of the manure production will be processed in a factory where no ammonia volatilizes. No emission will occur during the application of the processed manure.

Again it is shown that emission low application has a great impact - a reduction of 50 percent - and the costs are rather low. The second method is adaptation of compound feed; another 10 per cent can be realized and the total costs are almost twice as high. The costs of the abatement can be compared with the production costs (labor excluded) of 860 guilders or the labor income of the farmer of 39 guilder per fattening pig place. This last figure makes it clear that a reduction of 90 per cent has only a technical meaning. If the farmer has to pay these costs, he will be forced to stop farming and no ammonia will volatilize.

Results for other animal species are more or less comparable with those mentioned in the Tables 9 and 10. For farms with sows in Figure 3 a brief overview is shown of the reduction and the costs belonging to the abatement possibilities. This Figure gives a good impression of the considerable increase of costs beyond the point of a fifty or sixty per cent decrease of ammonia emission.

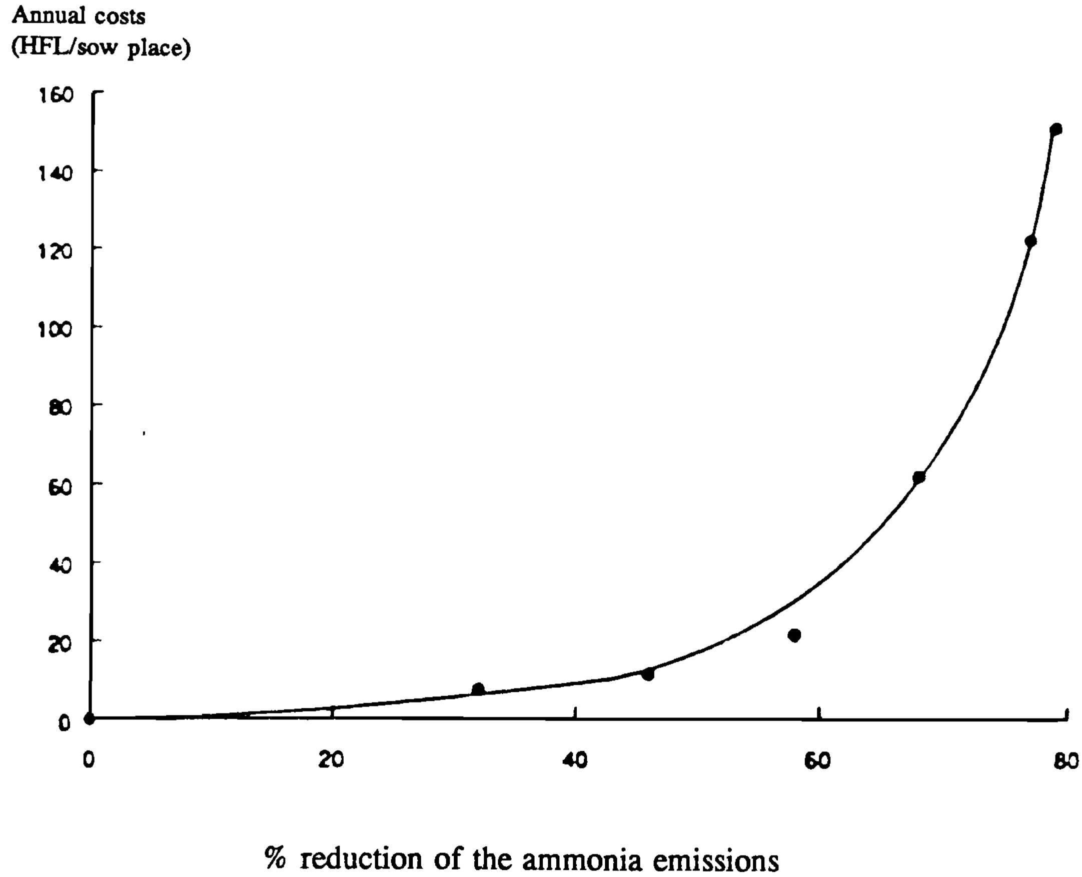


Figure 3. The annual additional costs in HFL per sow place and the level of ammonia emission. (Source: Baltussen et al., 1990b)

# 4 Conclusions and discussion

The Dutch livestock sector differs considerably from other European countries. The livestock numbers per hectare are relatively high and pigs and poultry have a relatively high share.

In the Netherlands dairy cows contribute some 50 per cent to the total ammonia emission; fattening pigs and sows 35 percent. Some 55 per cent of the total emission of ammonia volatilizes during the application of manure and some 37 per cent volatilizes in the stable.

The most efficient abatement methods for abating emission of ammonia are more advanced application methods, with a low emission. Adaptation of the feeding-stuffs is the second best alternative. This influences the nitrogen excretion (nitrogen concentration in the manure) and therefore decreases the ammonia emission in the stable, in the meadow, during the storage and during the application of manure.

A reduction of the emission of ammonia with 57 to 69 per cent can be achieved. The annual additional costs are about 400 respectively 1100 million guilders. An additional reduction of 12 per cent costs almost twice as much as the first 57 per cent reduction. The government objective for the year 2000 is within reach. In several Dutch regions the level of ammonia emission remains high.

Because of the specific situation in the Netherlands - composition of the livestock numbers and soil type - the abatement methods are only partly relevant for other countries. Adaptation of stables and adaptation of the composition of compound feed for pigs and poultry are most relevant for other countries. Application methods for grassland need soil without stones and other obstacles and flat areas. Direct ploughing down of manure on arable land is of course applicable in all countries.

The adaptation of buildings is still in research and therefore tentative. Air filtration is only possible in buildings with a mechanical air conditioner (fans) and is very expensive.

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# POSSIBILITIES AND COSTS OF NITROGEN REDUCTION THROUGH ADAPTED FEEDING

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## Abstract

Ammonia emission caused by livestock can be reduced by minimizing the N-excretion per unit produced by specifically matching nutrient requirements. Possibilities to reduce N-excretion are determination of requirements, realisation of nutrient recommendations, phase feeding, increasing N-utilization and supplementation of growth promoters. Phase feeding and supplementation of synthetic amino acids for fattening pigs can reduce N-excretion up to 40%. In feeding ruminants, however, there are limitations to adapted feeding because of rumen microbial turnover. For dairy cows feeding of special concentrates for balancing crude protein of the basic ration may be successful. Several possibilities of adapted feed regimes are discussed referring to feed and management costs, such as input of labor, required farm management, and advisory service. Reducing N-excretion to 1/3 through supplementation of synthetic amino acids raises costs by about 10 DM per pig.

# 1 Introduction

The most important source of ammonia emission in animal production is nitrogen excreted via faeces and urine. Nitrogen of urine (urea) is degraded mainly to ammonia. Quantity and pathways of N-excretion depend on intensity of feeding and composition of diet. Adapted feeding is the fundamental approach to reduce N-excretion and therefore ammonia emission.

N-excretion can be expressed per unit of product or per hectare. To reduce global ammonia emission N-excretion should be expressed per unit of product.

A minimal N-excretion per unit produced can be realized by feeding according to the exact nutrient requirements. This should be the intention of an ecological feeding regime. Review 1 shows several aspects to reduce N-excretion.

**Review 1**: Approaches to reduce N-excretion per unit produced through adapted feeding

1) Determination of requirements
   - permanent task of science

2) Realization of current recommendations
   - avoiding excess nutrient supply
   - avoiding deficiency in nutrient contents

3) Phase feeding
   - adapting nutrient contents according to live weight and performance

4) Increasing nitrogen utilization of diet
   - selecting adequate feedstuffs
   - supplementation of synthetic amino acids
   - supplementation of "protected" protein

5) Supplementation of growth promoters

Feeding according to recommendations can only be successful with a good knowledge of nutrient requirements. Examples for actual research work on nutrient requirements are: requirement of essential amino acids in feeding pigs and poultry and nitrogen turnover of ruminants. Ecologically orientated research means more basic research in animal production. These nutrient recommendations need to be transferred into practical feeding. In the remainder some considerations regarding adapted feeding of pigs and cattle are presented.

# 2 Possibilities of adapted feeding

## 2.1 Pig feeding

In West Germany 65 % of feed for fattening pigs is farm mixed. There is little information about the nutrient contents in these farm mixed diets. Spiekers and Bunge (1990) investigated pig feeding in Westphalia. In this inquiry 2/3 of the examined feed mixtures did not meet recommended values. Table 1 shows some results for lysine and crude protein in starter and finisher diets .

Table 1. Evaluation of diets for fattening pigs in farms of Westfalen-Lippe (results of an inquiry)

| Diet | Starter | Finisher |
|---|---|---|
| Number | 61 | 22 |
| Initial live weight (kg) | 23 | 51 |
| Lysine content: | | |
| excessive | 11 % | 41 % |
| short | 39 % | 14 % |
| Crude protein content: | | |
| excessive | 21 % | 64 % |

Consequences of deficient contents are lower performance with corresponding increase in feed and nitrogen inputs. The diets with excessive crude protein contents showed a mean value of 189 g crude protein per kg in starter and 181 g in finisher diets. The recommended values are 175 g crude protein per kg in starter and 150 g in finisher diets. In the finishing period an adaptation of crude protein contents could decrease N-excretion up to 25 %. Further studies confirmed these results. Comparable problems exist in feeding of sows (Bunge and Spiekers, 1989). A consequent application of existing nutrient recommendations could result in a considerable decrease of $NH_3$-emissions.

The second approach to reduce N-excretion in pig production is phase feeding. Figure 1 shows the wide range of lysine recommendations during fattening. From piglet to the end of the fattening period the recommended content of lysine per MJ metabolizable energy halves. Adjusting the amino acid and crude protein supply several times during fattening leads to a considerable decrease in ammonia emission.

At our institute, several trials were conducted dealing with phase feeding and supplementation of synthetic amino acids (Beste, 1988; Kleine Klausing, 1990; Spiekers et al., 1990). Fattening periods, daily gains and feed consumption are presented in Table 2. Table 3 shows the crude protein contents realized and the resulting N-excretion. These trials demonstrate the possibilities for reducing N-excretion by using phase feeding and supplementation with synthetic amino acids.

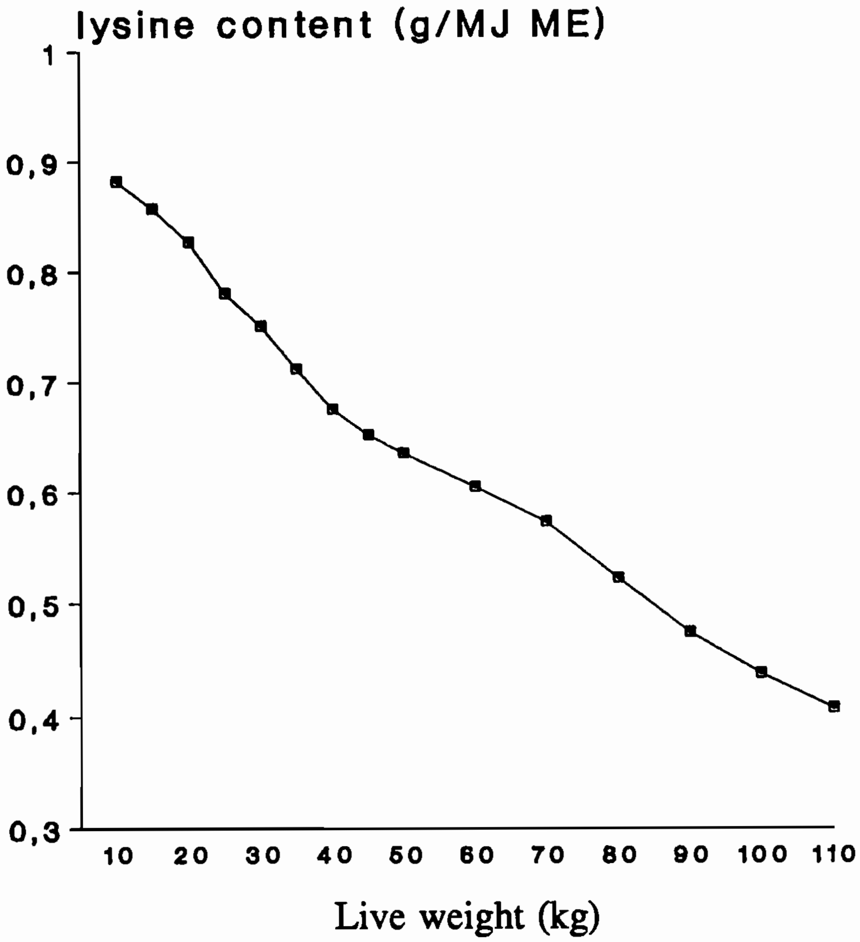


Figure 1. Lysine recommendations in fattening pigs during the fattening period (mean daily gain 700 g) (GfE, 1987)

Table 2. Mean performance of 10 feeding trials in fattening pigs (diet: wheat, soybean meal, soybean oil and minerals)

| Period | Live weight (kg) | Gain (g/day) | Feed consumption (kg) |
|---|---|---|---|
| Starter | 30 - 50 | 610 | 51 |
| Grower | 50 - 75 | 700 | 75 |
| Finisher | 75 - 100 | 780 | 83 |

Table 3. Comparison of different feeding systems in fattening pigs (ration: wheat, soybean meal, soybean oil and minerals)

| Feeding system | single-phase | two-phase | three-phase | three-phase + lys, met, thr, trp |
|---|---|---|---|---|
| **Starter:** | | | | |
| crude protein (g/kg) | 197 | 197 | 197 | 161 |
| **Grower:** | | | | |
| crude protein (g/kg) | | 182 | 182 | 142 |
| **Finisher:** | | | | |
| crude protein (g/kg) | | | 164 | 118 |
| N - fed (kg per pig) | 6.6 | 6.2 | 6.0 | 4.6 |
| N - deposit (kg per pig) | - 1.8 - | | | |
| N - excretion (kg per pig) | 4.8 | 4.4 | 4.2 | 2.8 |
| **Relative (%) N** | 109 | 100 | 95 | 64 |

Another approach to reduce the N-excretion is feed formulation aiming at minimizing crude protein content. Thus the same supply of essential amino acids can be realized with less crude protein by optimizing protein and energy sources. Table 4 shows a review of trials at our institute which demonstrate possible reductions of N-excretion. Further trials dealing with reduction of N-excretion have been published by Lenis (1987), Neupert and Essen (1988), Lenis (1989), Schwarz (1989), Essen (1989), Broecke and Aumüller (1990) and Jongbloed and Coppoolse (1990).

Table 4. Possibilities to reduce N-excretion in fattening pigs (30 to 100 kg live weight) in comparison to a wheat/soybean meal control diet (results of trials at the Institut für Tierernährung, Bonn)

| Measure | Reduction (%) |
|---|---|
| Exchange of wheat against oats | 20 |
| Exchange of soybean meal against peas and methionine | 20 |
| Exchange of soybean meal against field beans and methionine | 12 |
| Soybean meal portion reduced by supplementation of lysine and methionine | 27 |
| Soybean meal portion reduced by supplementation of lysine, methionine, threonine and tryptophan | 34 |

An adapted feed regime for sows leads to an appreciable reduction of N-excretion. Implementation of a two-phase feeding regime with different diets for pregnant and suckling sows is important. In feeding piglets supplementation of synthetic amino acids seems to be recommendable. Supplementation with the commercially available synthetic amino acids lysine, methionine, threonine and tryptophan is of special interest. N-excretion can be reduced by up to 20 % when such adapted feeding regimes are applied for piglets and sows (Spiekers and Pfeffer, 1990). In the Netherlands, two-phase feeding of sows could decrease N-excretion up to 24 % (Lenis, 1989).

At our institute we fed lysine, methionine, threonine, tryptophan and isoleucine to piglets. N-excretion was reduced by up to 1/3 (Spiekers et al., 1990). Further progress in reduction N-excretion through pig nutrition is expected by changing the amino acid recommendations from gross to praecaecal digestible amino acids (Lenis, 1989). As this state of knowledge, however, there is insufficient information to introduce this system in practice. Growth promotors can influence performance and feed consumption and therefore reduce N-excretion per unit produced.

## 2.2 Cattle feeding

Cattle, especially dairy cows, cause the main part of ammonia emission (Asman and Jaarsveld, 1990; Broecke and Aumüller, 1990; Möller and Schieferdecker, 1990). The amount of concentrate supplied per dairy cow varies widely. The effect of the level of concentrate supply on N-excretion was calculated according to data of Pfeffer and Potthast (1988). They fed different amounts of concentrates, while roughage was fed ad libitum. The calculations are based on a milk quota of 100.000 kg (see Table 5 ).

Table 5. N-excretion per 100,000 kg milk quota at different supply of concentrates (calculation according to Pfeffer and Potthast, 1988)

| Concentrates dt/cow per year | 9 | 14 | 19 | 24 |
|---|---|---|---|---|
| Milk yield kg/cow per year | 5,600 | 6,200 | 6,300 | 6,400 |
| No. of cows per 100,000 kg milk | 17.9 | 16.1 | 15.9 | 15.7 |
| N-intake, kg roughage[1] | 1,840 | 1,644 | 1,521 | 1,386 |
| Concentrates[2] | 464 | 649 | 870 | 1,085 |
| Σ | 2,304 | 2,293 | 2,391 | 2,471 |
| N-export, kg milk | 540 | 544 | 560 | 565 |
| N-balance, kg | 1,764 | 1,749 | 1,831 | 1,906 |
| per forage area - kg/hectare[3] | 215 | 238 | 270 | 308 |

1) 14 % crude protein in dry matter
2) 18 % crude protein
3) 100 dt dry matter per hectare

Concentrate levels of more than 14 dt per cow per year result in increasing N-surplus per 100.000 kg milk quota. Accordingly the N-surplus per ha forage area rises with the concentrate level from 215 to 308 kg N per ha per year.

The next subject to be pointed out is the crude protein balance in dairy cows diets. Excessive protein supply is a common problem to German dairy farmers. A report by Mohrenstecher-Strie (1989) showed that 20 out of 25 investigated diets had daily crude protein surpluses between 90 and 620 g per cow. This is due to the fact that grass and its preserves contain high amounts of crude protein per unit of energy. This problem is intensified by the tendency towards higher protein contents in grass which are caused by fertilization, number of cuttings and selection of high yielding varieties.

Usually a protein surplus in single feedstuffs is compensated by mixing them with low protein components. Model calculations by Spiekers and Pfeffer (1990) showed that maize silage is very suitable for this purpose. But in those areas where gras is the only available roughage there are considerable problems in feeding protein balanced diets.

Table 6 shows that the classical feeds for compensating crude protein, dried sugar beet pulp and cereals, contain about 15 g crude protein per MJ NEL and therefore have only a limited capacity for crude protein compensation. In this regard the use of a special industrial manufactured balance concentrate is desirable.

Table 6. Energy and crude protein contents of feedstuffs for crude protein balanced rations in feeding dairy cows

| Feedstuff | Energy | Crude protein | |
|---|---|---|---|
| | MJ NEL/kg | g/kg | g/MJ NEL |
| Wheat | 8.1 | 120 | 14.8 |
| Dried sugar beet pulp | 6.7 | 101 | 15.1 |
| Barley | 7.0 | 105 | 14.6 |
| Balance concentrate* | 7.0 | 50 | 7.1 |

* special mixture

This feed should contain 50 g crude protein per kg at an energy level of 7 MJ NEL per kg. The composition of such a feed could be 27 % manioc, 70% citrus pulp and 3 % minerals. It contains 7.1 g crude protein per MJ NEL and could compensate considerable amounts of crude protein of the basic ration.

From the figures given in Table 7 it is evident that the use of a special balance concentrate reduces N-excess most effectively regardless whether the diet is based on grass silage or pasture. On a dairy farm with permanent grassland only and a production level of 6.000 kg milk per cow per year the use of a special balance concentrate can reduce N-excretion in the range of 10 to 15 kg per cow per year. A further decrease of emissions can be accomplished in the feeding of low yielding cows provided that part of the daily intake from pasture or grass silage can be replaced by maize silage or other feedstuffs of low crude protein contents.

Table 7. N-excess (g per cow per day) after crude protein balancing in rations based on grass silage or pasture

| Ration | Concentrates[1] off ... kg milk | Milk yield (kg per cow per day) | | |
|---|---|---|---|---|
| | | 30 | 20 | 10 |
| Grass silage ad libitum | 8 | 69 | 74 | 77 |
| + 6.2 kg dried sugar beet pulp | 21 | - | 6 | 66 |
| + 5.1 kg wheat | 21 | - | 6 | 67 |
| + 3.5 kg balance concentrate[2] | 16 | - | - | 58 |
| Pasture ad libitum | 16 | 85 | 90 | 174 |
| + 10 kg maize silage | 16 | 43 | 45 | 131 |
| + 10 kg maize silage + 3 kg balance concentrate[2] | 21 | - | 10 | 131 |
| + 7.3 dried sugar beet pulp | 31 | 8 | 68 | 174 |
| + 6 kg wheat | 31 | 5 | 67 | 174 |
| + 4 kg balance concentrate[2] | 25 | - | 53 | 174 |

1) 18 % crude protein; 6.9 MJ NEL per kg
2) 5 % crude protein; 7.0 MJ NEL per kg

Another clue to solving the problem of N-pollution is protein quality. As milk yield increases, decreasing degradability is required (GEH, 1986) to provide for sufficient undegraded protein in the small intestine. If easily degraded protein is fed to high producing dairy cows the daily protein intake has to be increased to meet the total amino acid requirement in the small intestine. Spiekers and Pfeffer (1990) demonstrated these relationships with some calculations.

In this scenario, feedstuffs with low crude protein degradabilities are needed. However, the evaluation of degradability is controversial since no practicable method has yet been established. Altogether the positive effect of such feed on N-pollution is only small because it can only be expected at high production levels and at the same time high degradabilities of protein (grass silage, pasture) as shown above.

The quality of undegraded feed protein is of minor importance. Microbial protein which accounts for the greater portion of N in the intestines sufficiently matches the requirements for milk production with regard to its amino acid pattern (ARC, 1984). Accordingly, trials with "protected" amino acids in most cases showed no effect under German feeding conditions (Spiekers, 1988). Of greater importance in practice is the improvement of microbial protein synthesis by supplying sufficient amounts of energy.

In fattening bulls, however, the chances to reduce ammonia emissions are small if the nutrients recommended are met. Spiekers and Pfeffer (1990) showed that there is only a small decrease in the recommended amount of crude protein per unit of energy during fattening. N-demand of rumen microbes caused the difference between fattening bulls and fattening pigs. In fattening bulls crude protein degradability is of importance only at the beginning of the fattening period. In this period consideration of crude protein degradability in feed formulation could reduce N-excretion.

Supplementation of growth promotors in bulls had a small effect on N-excretion. Feed conversion could be improved.

# 3 Costs of adapted feeding

For the shown possibilities realization of adapted feeding in practice caused different costs and increased expenses for advice and training. Differences exist in feed costs and technical costs. Table 8 gives a review of costs, demand of advisory service and farm management and efficacy of adapted feeding to reduce $NH_x$-emission.

In the following text, the feeding regimes for pig and cattle production are discussed.

## 3.1 Pig production

An efficient transfer of feed recommendations in practice requires an increase in feedstuff analysis and systematic advisory service. An assumption must be sufficient farm management. This is given to most of the young well educated farmers.

In fattening pigs and piglet production phase feeding causes lower feed costs according to prices of protein and energy feeds (Spiekers et al, 1989). On the other side, costs of feed technology and input of labor increase. Often additional investments are necessary in feed silos and feed technology (Schwarz, 1989). The standard of farm management had to be increased.

Table 8. Possibilities and costs to reduce $NH_x$-emission through feeding

| Livestock | Activity | Costs | | | Requirement of | | Efficacy in reducing $NH_x$ emission |
|---|---|---|---|---|---|---|---|
| | | feed | general | labor | farm management | advisory service | |
| **1) Transfer of nutrient recommendations in practice** | | | | | | | |
| Pig and cattle production | systematic advice (feedstuff analysis) | | + | + | +++ | +++ | ++ |
| Dairy cows | special CP balance concentrates | + | ++ | ++ | + | + | +++ |
| **2) Phase feeding** | | | | | | | |
| Fattening pigs | starter, grower and finisher | | ++ | ++ | + | + | ++ |
| Sows | lactation and pregnancy | | ++ | ++ | + | + | ++ |
| **3) Selected protein and energy feeds** | | | | | | | |
| Pig and poultry | minimize CP | ++ | + | | ++ | ++ | ++ |
| Dairy cows | less CP degradability | + | | | ++ | ++ | + |
| **4) Supplementation of synthetic amino acids** | | | | | | | |
| Piglets | lys, met, thr and trp | ++ | | | + | + | +++ |
| Fattening pigs | lys, met, thr and trp | +++ | | | + | + | +++ |
| Sows (lactation) | lys, met, thr and trp | +++ | | | + | + | ++ |
| Poultry | met and lys | ++ | | | + | + | ++ |
| **5) Supplementation of "protected" protein** | | | | | | | |
| Dairy cow | decrease CP | ++ | + | + | ++ | ++ | + |
| Fattening bulls | decrease CP | ++ | + | + | ++ | ++ | + |
| **6) Supplementation of growth promoters** | | | | | | | |
| Fattening pigs | reduction in feed consumption | + | | | + | + | + |
| Fattening bulls | reduction in feed consumption | + | | | + | + | + |

+ low; ++ medium; +++ high
CP = crude protein

Ecologically orientated feed formulation, particularly supplementation with synthetic amino acids, increases feed costs. Additional feed costs rose more than proportional with increasing reduction of N-excretion (Broecke and Aumüller, 1990). This effect is caused by the increasing costs of supplementation of synthetic amino acids.

Table 9 shows, for two feeding trials, changes in feed costs with and without supplementation of lysine, methionine, threonine and tryptophan. A 1/3 decrease in N-excretion caused an increase in feed costs of 10 DM per pig. Hopp et al. (1990) showed an increase in feed costs of about 9 DM when N-excretion decreased from 4.1 to 2.5 kg N per pig. Comparable costs resulted in the Netherlands (Broecke and Aumüller, 1990).

Table 9. N-excretion and feed costs in fattening pigs (30 - 100 kg live weight) with supplementation of lysine, methionine, threonine and tryptophan

| Protein feed | N-excretion | Feed costs (DM per pig) | |
|---|---|---|---|
| | kg per pig | Amino acids | Total |
| **Trial based on wheat (Spiekers et al., 1990)** | | | |
| Soybean meal* | 4.4 | - | 92.0 |
| Soybean meal | 3.0 | 12.8 | 100.8 |
| **Trial based on barley/wheat (Grünewald, 1992)** | | | |
| Soybean meal | 2.8 | 12.9 | 102.3 |
| Field beans | 2.9 | 14.4 | 105.5 |
| Canola meal | 3.1 | 10.8 | 103.6 |
| Peas | 2.7 | 13.3 | 100.5 |

* without amino acid supplementation

Extra costs of adapted feeding may ultimately ruin profitability in fattening pigs. Ecological aims therefore may necessitate changes in economic frameworks.

## 3.2 Cattle production

Feed analysis of roughage is supposed to realize feed recommendations in cattle production. Feed analysis is economical, but has a limited acceptance in practice. Some advisory service and ability in feed formulation is necessary to make use of roughage analysis. A useful method to examine protein provision of dairy cows is measuring milk urea contents (Mohrenstecher-Strie, 1989).

Calculations of Pfeffer and Potthast (1988), referring to economic and ecological optimum concentrate supply to dairy cows, showed good agreement between these two aims. Feeding special crude protein balance concentrates to dairy cows has only little influence on feed costs. Of importance are higher input of labor and possibly additional investments in feed technology for a second concentrate.

A small increase in feed costs causes feed formulation which consider crude protein degradability. Of greater importance are the mentioned problems in measuring crude protein degradability. Supplementation of "protected" protein increases feed costs, too.

# 4 Recommendations to introduce adapted feeding

As shown, there are considerable possibilities to reduce ammonia emissions through adapted feeding. In feeding poultry, horses, sheep, goats and pet animals there are possibilities in adapted feeding, too.

Review 2 shows recommendations in feed industry, advisory service, administration and science to introduce adapted feeding into practice.

**Review 2**: Recommendations to reduce ammonia emissions through adapted feeding

<u>feed industry:</u>

- declaration of higher nutrient contents
- offering special feeds: - balance feeds
  - phase feeds
- consideration of crude protein degradability in feed formulation
- economic valuation of ecology in feed formulation

<u>advisory service:</u>

- escort of animal production in feeding
- N-balance for ranking of farms
- consideration of crude protein degradability in feed advice
- special training programs for farmer

<u>administration:</u>

- evaluation of animal production according to real N-excretion
- benefit supplementation of synthetic amino acids
- benefit special feed advice
- benefit investments for phase feeding

science:
- more research about nutrient requirements
- creating extend feed evaluation systems
- accelerate introduction of given knowledge

These recommendations should be basic for further discussions. Differences in feeding, economies and administration between countries have to be taken into account.

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# REDUCTION OF AMMONIA VOLATILIZATION FROM ANIMAL HOUSES

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## Abstract

The paper deals with the ammonia volatilization from animal husbandry in the Netherlands, especially with the volatilization from animal houses and storage tanks. Examples are given of changes in housing systems to reduce the ammonia emission. Attention is paid to slurry removal systems (pigs), drying systems (poultry), floor cleaning systems (cattle) and deep litter systems (pigs). From these technical solutions the effect on ammonia emission is given as far as available. The same holds for the costs of the new systems. Air cleaning seems not to be an economic solution, mainly because of energy use, ventilation techniques and the removal of the diluted water.

# 1 Introduction

The ammonia volatilization from animal husbandry contributes to the acidification of the environment. To reduce this acidification in the future a substantial reduction of the ammonia volatilization must be reached. For several years a research program was executed in the Netherlands to reduce the ammonia emission from animal industry. There are three main tracks in this program: animal feeding, slurry application and housing systems.

This paper mainly deals with animal housing systems. There are principal differences between cattle, pigs and poultry houses. The solutions tested in research and in practice are based on the basic knowledge of ammonia production and volatilization. A reliable and accurate measuring of the ammonia volatilization from animal houses is difficult. For naturally ventilated houses these measurements are hardly possible.

Calculations of extra investment and running costs are difficult to make for housing systems with lower ammonia emission.

# 2 Basic knowledge

Faeces and urine contain, among other products, undigested proteins, N-compounds and urea (cattle and pigs) and uric acid (poultry). These products are converted to ammonia. Urea is converted within a very short time after the urine is produced. The converting time for uric acid is significantly longer. The equilibrium between $NH_4^+$ and $NH_3$ can be written as:

$$NH_3 + H_2O \leftrightarrow NH_4OH \leftrightarrow NH_4^+ + OH^-$$

The ammonia volatilization depends on the concentration of $NH_3$ in the slurry, manure or litter. Verdoes and Aarnink (1991) describe the most important factors which influence the concentration of $NH_3$ and the volatilization:

- pH: below pH 6 there is nearly no $NH_3$ left and the volatilization is nearly zero;
- temperature: the volatilization is greater when the temperature rises;
- concentration: the volatilization increases with the concentration;
- movement of the liquid and the top air layer: in a quiet situation the volatilization of $NH_3$ is lower;
- aeration: air injection will nitrificate the ammonia to $NO_3$. But if the concentration is too high at starting aeration there will still be volatilization of ammonia.

This basic knowledge is used to reduce the ammonia emission from animal houses.

# 3 Changes in housing systems

## 3.1 Pig houses

The first condition is storing the slurry outside the pig house in a sealed tank. The next important item is the transportation of the pig slurry to that tank. The slurry should not stay longer than necessary in the pig house. Therefore the pen floor is also important related to the ammonia volatilization. In the dunging area there must be a slatted floor with a high throughput for the faeces. The Research Institute for Pig Husbandry, mostly in cooperation with the Research Institute for Agriculture Engineering, is testing the following options:

a. scrapers on a sloping floor under the slats;
b. sewerage pipes under the floor of the slurry canal;

c. flushing systems in several designs.
Besides this strategy there is another approach:
d. deep litter system, based on woodchips.

ad a. Scrapers
The pig house has partly slatted floors and the canal under the slats has a sloping floor. This slope is two-sided: in the length of the canal (2%) and a transverse slope (3%). In the middle there is a small gutter to guide the scraper and to drain away the urine continuously. The scraper is used twice a day. In this system straw can be used as a bedding because the scraper ensures a total removal of the mixture of faeces with straw. In the future, the use of a bedding material can be prescribed in the Netherlands to improve animal welfare. By using a bedding the energy use for heating will be lower.

In this system it is very important that:
- the solid floor does not become dirty,
- the floor of the canal is very smooth,
- the urine cannot penetrate in the floor and
- the slats have a good throughput for the faeces.

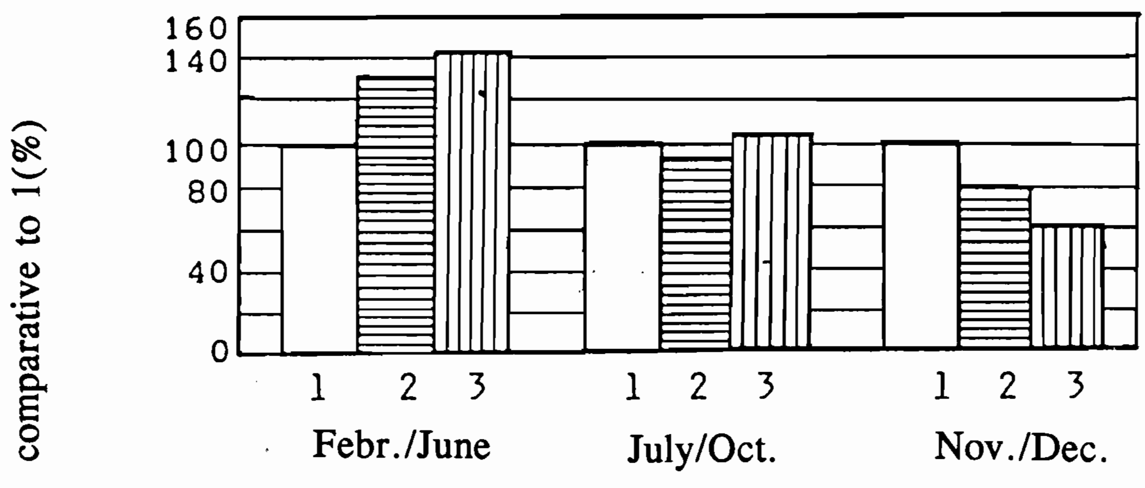


Figure 1: Seasonal effect on the ammonia emission from storage systems in a pig house

1. storage under the whole house (4 months)
2. only storage under the slats (1 week)
3. scraper system as described

Up to now there was always either more or less pollution of the solid floor by the finishing pigs. So it is not clear what the exact effect of this system is on the reduction of ammonia volatilization. The best results show a reduction of 40%. Experiments also showed that during and after separate application of the solids and the urine on the land the $NH_3$ emission is lower than by applying the unseparated slurry. Calculations of this system for a pig house with 6 compartments, with 80 heads each, show an extra investment of HFL (Dutch Guilders) 111,000.--. The annual cost, consisting of extra investment and the running costs, are about HFL 15.-- per delivered pig. By using straw as a bedding the annual cost are HFL 18.--.

ad b. Sewerage pipes
A pipe (diameter 200 mm) is mounted under the flat floor of the canals. These canals are under the slatted part only and are about 1.6 m wide. The pipe is connected to the floor by a T-pipe every 2 m. The end of this sewerage pipe is blocked by a so-called Apollo ball, which can be pulled up for a nearly complete discharge to the storage pit. Mostly the slurry is pumped from this pit to the storage tank. This discharge is done weekly. Up to now there are no data available about the reduction of the ammonia volatilization.

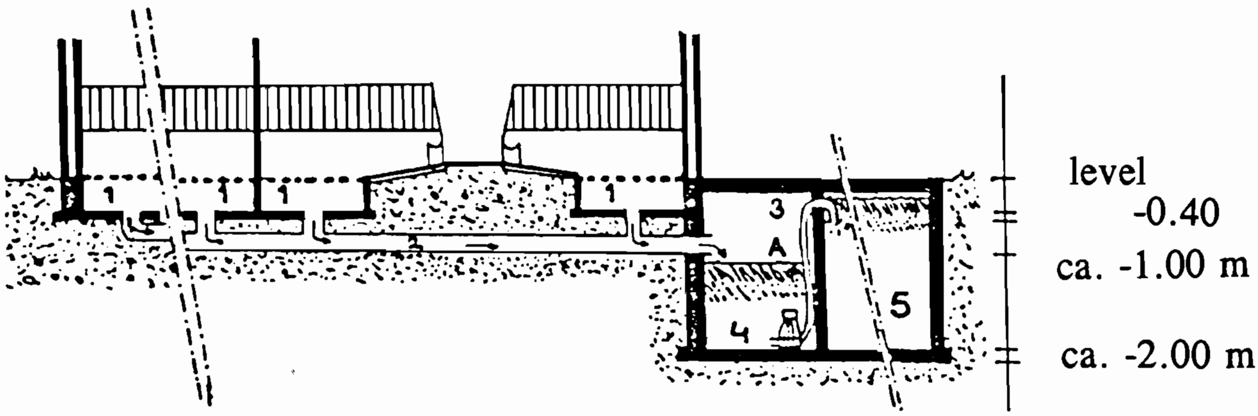


Figure 2: The sewage pipe system

1. 40 cm deep canal
2. sewage pipe
3. small tank
4. slurry pump
5. large tank

The slurry pans in a farrowing unit are a modification of the sewerage pipe system. These pans are situated under the slats and connected to a sewerage pipe. At a weekly discharge the ammonia volatilization is reduced by about 40%.

ad c. Flushing systems
In case of flushing systems a liquid is used to remove the slurry out of the canals. This can be done in several ways.

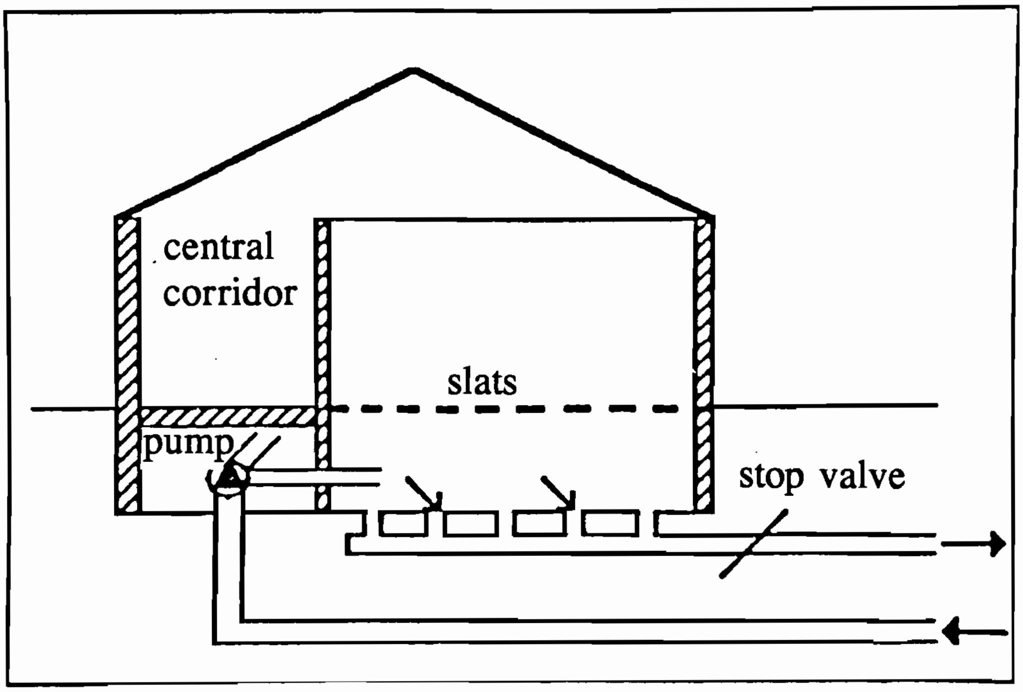


Figure 3: Recharge

Combination of flushing with a sewage pipe system

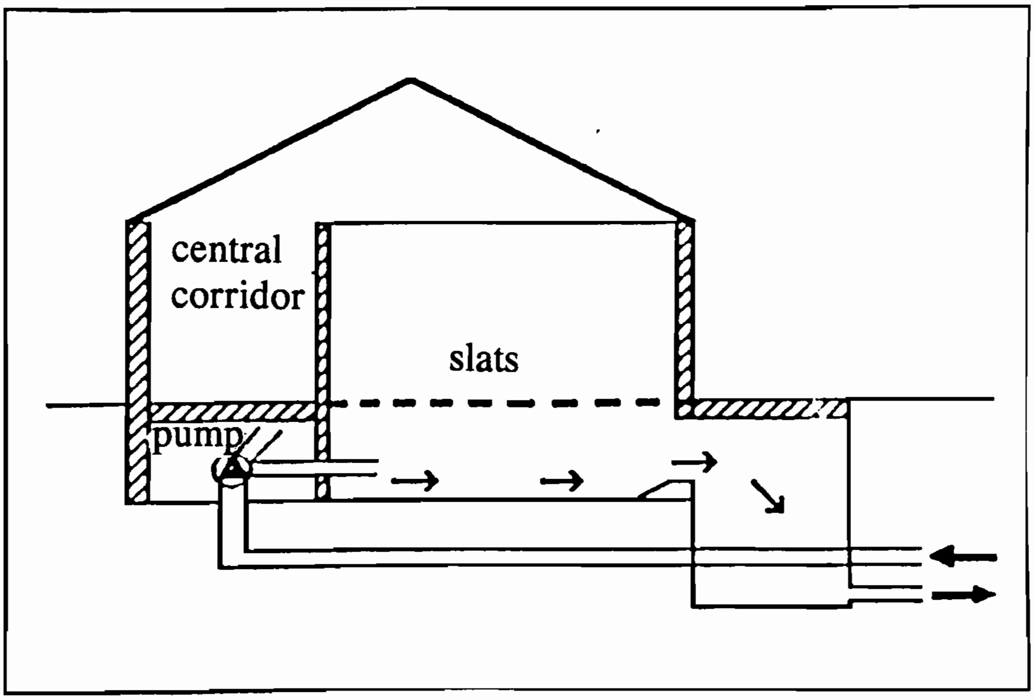


Figure 4: Diluting
Flushing system with a threshold in the canal

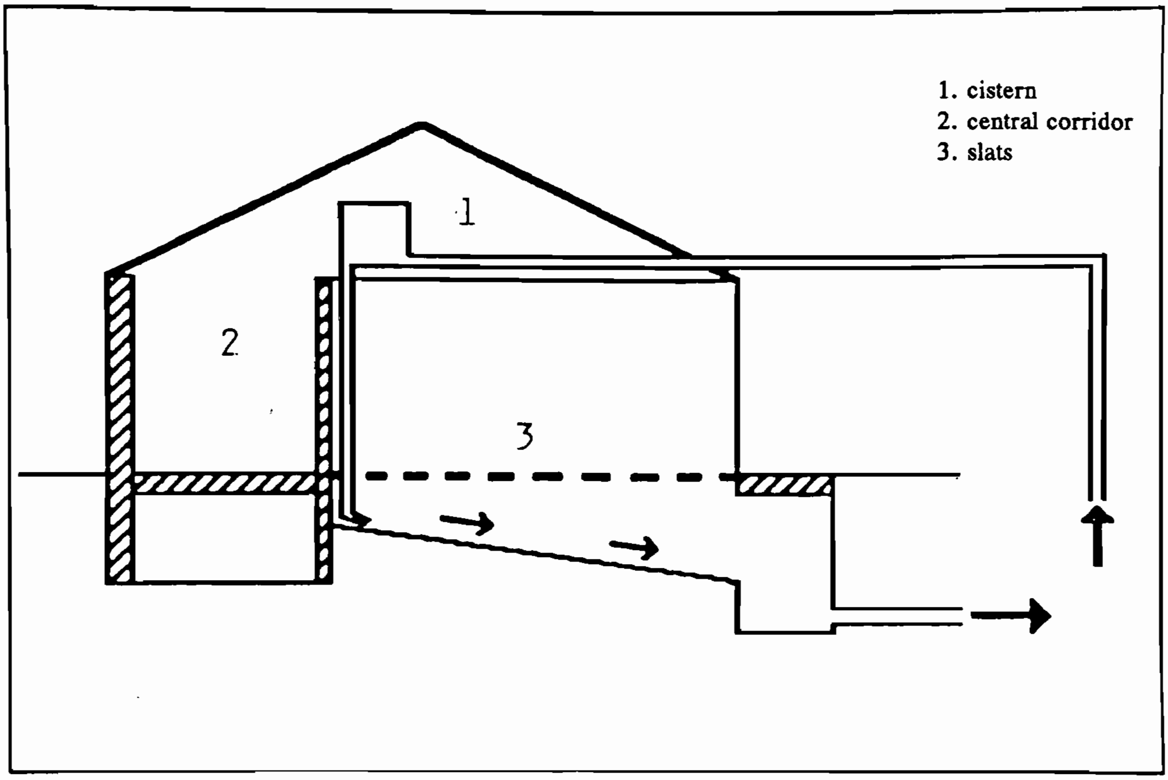


Figure 5: Flushing system on a sloping floor

The differences concern the lay-out of the canals and the techniques of production of the flushing liquids. Experimental data are only available from three flushing systems and with one system for the production of the flushing liquid. The results are nearly similar with a light favor for the re-charge system. The flushing liquid is produced by aerating the liquid fraction after separating the flushed manure. As a result the available $NH_3$ is converted to $NO_3$. Through denitrification $N_2$ is produced, which will volatilize. Due to this treatment there is no ammonia left in the flushing liquid.

Compared with fully slatted floors and storage of the slurry underneath the pighouse a reduction of the $NH_3$ volatilization of nearly 70 % has been reached by this system. It is plausible that the remaining 30 % volatilizes from the floors. For that reason extra attention has to be paid to the slats now. A better throughput of the faeces is very important. Concrete slats with 100 mm beams and 18 mm slits have a very poor throughput. Metal slats have a better throughput, but are substantially more expensive.

For flushing systems additional investments are needed. On a practical scale no data are available to make reliable calculations.

ad d. Deep litter system

Deep litter systems are based on the decomposition of the manure inside the pig house. In the pens there is a layer of at least 70 cm small woodchips, which stay there for at least two years. The manure is mixed with these chips weekly and an additive is added to control the processes. The temperature within the deep litter bed is between 40 and 50° C. Because of this relatively high temperature the moisture evaporates and the dry matter content of the bed remains constant (50 - 60%). There is rather an accumulation of minerals in the bed, except nitrogen. There are several possibilities for the N losses: $NH_3$, $N_2$ and $N_2O$. At this time no data are available to give a reliable impression of the N losses.

This system does not have higher investment costs. Up to now the costs of the additives used and of the mechanisation to do the mixing are high. This seems to make the system more expensive. Real calculations can only be made after data on animal performances are available.

## 3.2 Poultry houses

Most work in this field is done by Kroodsma (IMAG) in cooperation with the Poultry Research Institute (COVP). It is necessary to make a distinction between laying hens and broilers. Broilers are housed on a litter system, whereas laying hens are in cages. In both situations it is important to keep the manure as dry as possible.

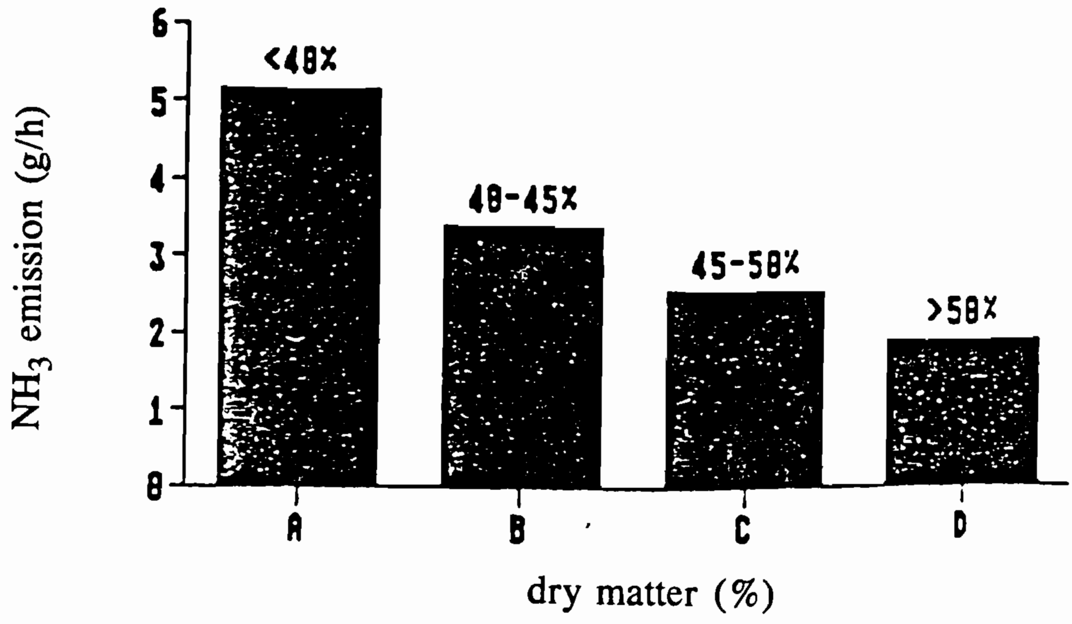


Figure 6: Relation between the ammonia emission and the dry matter content of the poultry manure (Kroodsma et al., 1989)

### 3.2.1 Laying hens

Most houses for laying hens are based on cages with storage of wet manure. To reduce the ammonia emission it is necessary to store the manure in a covered tank. In such a system the annual ammonia emission can be limited to 30 g $NH_3$ per hen. New developments are based on the drying of the poultry manure. This can be done on belts under the cages by blowing air over the belts. With such a system a dry matter content of 45 - 60% can be reached. Storage of this manure in a covered place means that the manure gets heated and that there still will be an emission of moisture and also of ammonia. The ammonia emission can increase to 230 g per hen per year.

Therefore this manure should be dried to at least 70% dry matter. There are several possibilities for drying using the ventilation air of the hen house. In periods with low temperature and high moisture extra heating of the air is necessary. With such systems it is possible to reduce the ammonia emission to 20 g/hen per year. Some of these systems are now available on the Dutch market.

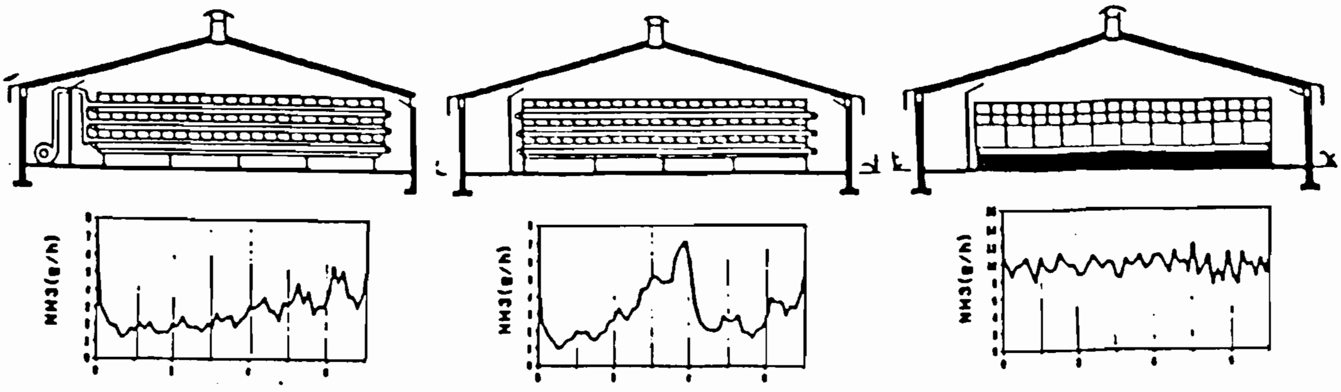


Figure 7: The average emission from three different cage systems for laying hens (Kroodsma, 1989)

### 3.2.2 Broilers

All broilers in the Netherlands are housed on litter systems. For the litter, mostly wood shavings are used. To reduce the ammonia emission it is important to keep the litter as dry as possible. Attention must be paid to the drinking system to prevent spoiling of water. A good ventilation system stimulates the evaporation and the removal of moisture. So there are different management rules which the farmer can use to prevent a wet litter. New systems have a (partly) net floor through which the droppings fall on a belt. Ventilation air is sucked over this belt to dry the manure. Also experiments are running with a curtain on which the litter is placed. The ventilation air is sucked through this curtain for keeping the litter dry. There are no data available about the ammonia emission. Also the exact costs are unknown.

### 3.2.3 Houses for dairy cattle

In cooperation with the Research Institute for Cattle Husbandry the Institute for Agricultural Engineering in the Netherlands did most research on ammonia emission from houses for dairy cattle. Oosthoek (1989) concluded that at least 50% of the total emission is from the slatted floor in cubicle houses. To reduce the emission it is also important to store the slurry in a covered tank. The crust on dairy slurry can also reduce the emission, but less than a good cover. Reduction of the emission from the floors can only be reached by a combination of cleaning and spraying. Hence research develops systems with a combination of scrapers and water nozzles. The aim is to minimize the use of water, mainly because of the storage capacity. Systems with a solid sloping floor are also new. The urine drains to the middle where a small gutter is situated.

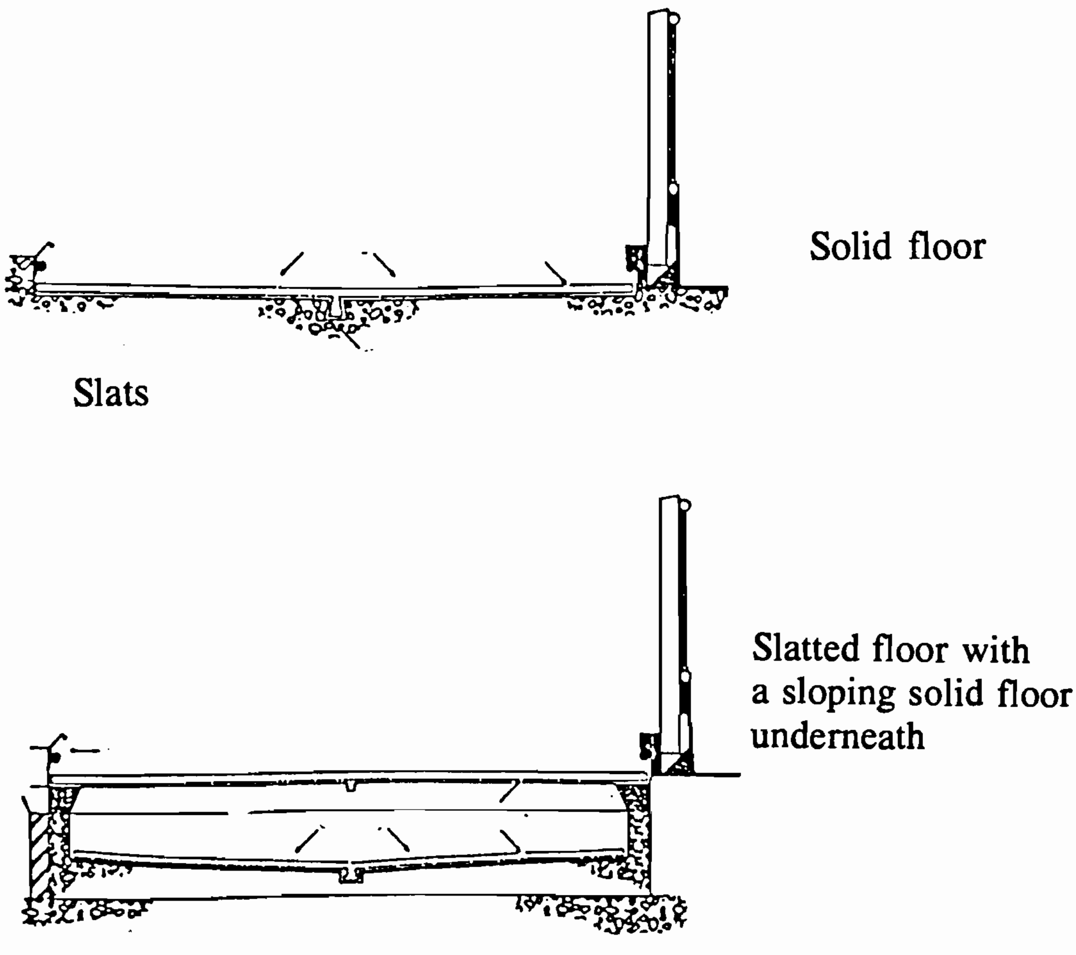


Figure 8: Experimental floors in cubicle house for dairy cows (Donker, 1989)

The faeces are scraped several times a day. There is also a combination of both systems, where the sloping floor is built under the slats. The first measurements show a considerable decrease of the ammonia emission. A completely different system is the acidification of the slurry in the storage pit under the slatted floor. With nitric acid the pH is lowered to 4.5. Such a low pH prevents any production of $NH_3$. In this system the slurry must be mixed completely every day and nitric acid is added automatically. This system increases the N-content of the slurry, but this is mostly useable on grassland. Up to now problems with denitrification have to be solved to prevent N-losses and rising pH levels.

There are no data available about the costs for these new systems, because the total effects of the systems are unknown.

# 4 Applicability of new systems to reduce $NH_3$ emission

The most important condition for a broader use of these new systems will be the economic and technical reliability. As soon as systems give good technical results calculations must be made on the economic effects. There can be other conditions to introduce the new techniques. In the Netherlands regulations are expected concerning the animal welfare and the animal health. There can be a contradiction between environmental aspects and animal welfare. Solid floors for pigs are desirable for the welfare, but when they get dirty the emission will increase rapidly. In principle there are no restrictions for use in other climatological conditions than the circumstances in the Netherlands. Sufficient information about the environmental effects of ammonia emission and some financial support to introduce the new systems will be important in the future.

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# COSTS OF EMISSION-POOR MANURE APPLICATION IN THE NETHERLANDS

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## Abstract

The increasing attention for environmental application of manure has led to new technical developments. The main object is: to reduce the ammonia emission at field application of slurry. Currently there are three systems developed which give a high reduction (85 - 100 %) of the ammonia leakage in comparison to the surface spreading of slurry: slurry injection, turf injection and turf impregnation. In the Netherlands the new methods of manure application can be done by the farmer or by the agricultural contractor, although deep injection is almost always done by contractors. An agricultural contractor normally buys a machine with a great working width. This requires an investment of HFL 150,000 excluding tractor. The contractor also invests in self propelled machines. Total investment HFL 420,000. The yearly costs depend on the investments and the use of the machine annually. The annual hourly costs amount to between HFL 199 and HFL 247. With a capacity of 30 to 40 $m^3$/hour, the price/$m^3$ amount to between HFL 5 and HFL 8. The distance between the plot and the storage of slurry is about 500 m. The expenditure by exploration of the machine by the farmer himself costs between HFL 8 and HFL 11/$m^3$ by a yearly slurry production of 1000 - 1500 $m^3$. In comparison to surface spreading, the fertilizing value of the new methods increases by HFL 1.50/$m^3$. When the new systems cost about HFL 3.50/$m^3$ more than surface spreading, the additional charges of environmentally applying the slurry, amounts to HFL 2/$m^3$. This means for a dairy farm about HFL 30/cow/year.

# 1 Introduction

In the Netherlands the environment is still an area of growing concern of which people become increasingly aware. This is evident from the number of legal measures being taken by the government to tackle environmental problems. One of these problems is manure emission. The government aims at reducing manure emissions by 30% in 1994 and by 70% in the year 2000 as compared to 1980. Manure application is to contribute substantially to this reduction. Efforts are made to come to an 80% emission reduction in above-ground spreading per 1 January 1995 (Ministerie LNV, 1990).

But even before the government had announced that it intended to enforce legal measures the agricultural sector was well aware that it had to do its bit. Thus during the latter half of the 1980s farmers began to turn their attention to the development and application of new methods for manure application that would be less detrimental to the environment.

In the Netherlands manure is applied in different ways that largely come under the following categories:

1. slurry injection (deep-injection)
2. turf injection (shallow-injection)
3. turf impregnation (sod-manuring)
4. the use of manure spreading harrows
5. irrigation
6. fertigation.

# 2 A short description of the different methods (Krebbers et al., 1991; DLV, 1991)

## 2.1 Slurry injection

Slurry injection means that manure is injected into the soil at a depth of 12 to 18 cm by a slurry injector. The injection tines are 50 cm apart. The broad duckfoot shares of the machine (16 to 18 cm) make for a proper distribution of the slurry into the soil. The ground is levelled afterwards by a pressure roller. The advantage of this machine is that the manure can be applied in large amounts at one go. The application of 40 $m^3$ per ha is no problem which makes it easy to apply the whole amount in spring. The disadvantage is that turf and roots are cut which increase the risk of desiccation. This makes the use of this method limited as regards the time of year and is less suitable for heavy clay and peaty soil. The nutrients reach the crops slightly later.

## 2.2 Turf injection

With this method, manure is injected into the soil (15-35 $m^3$ per ha) through vertical slits, spaced at widths of 20 to 30 cm. The soil is then levelled by two pressure rollers.

## 2.3 Turf impregnation

The turf impregnator makes slits into the soil 5 to 7 cm in depth into which the manure is placed. The slits are spaced at widths of 20 to 30 cm. This method allows for less manure to be applied than with the method of turf injection, nor is the soil levelled out afterwards. The two methods however are suitable for a wide range of soil conditions and can be carried out at all times. The poor carrying capacity of peaty soil may be a problem. Drying out is less of a problem with turf injection and virtually non-existent with turf impregnation. The nutrients are readily available.

Figure 1 shows a diagram of the methods as described above with an indication of how the manure is distributed in the soil.

## 2.4 Use of manure spreading harrows

The manure is spread over the soil by means of a system of hoses fitted with harrows. In this way the manure can be distributed in between the grass. The harrows are spaced 20 cm apart and applications of 10 to 15 $m^3$ per hectare can be made. The advantage of this system is that the turf itself is not damaged and the grass does not get soiled. The difference in density of the turf however makes the system not universally applicable as yet.

## 2.5 Irrigation

Direct irrigation of manure after or during spreading is done with an irrigation system consisting of a (moving) hose reel and sprinklers. The bigger moving hose reels are less suitable as the manure will dry on the grass. The disadvantage of this system is that it takes a lot of work and operates at low capacity. Another problem is the availability of surface water needed to operate the system now and in the future.

## 2.6 Fertigation

Via an automatic hose reel or a pendulum system the diluted manure is spread over the land. The desired dilution is one part manure three parts water. The maximum application is approximately 20 $m^3$ per ha.

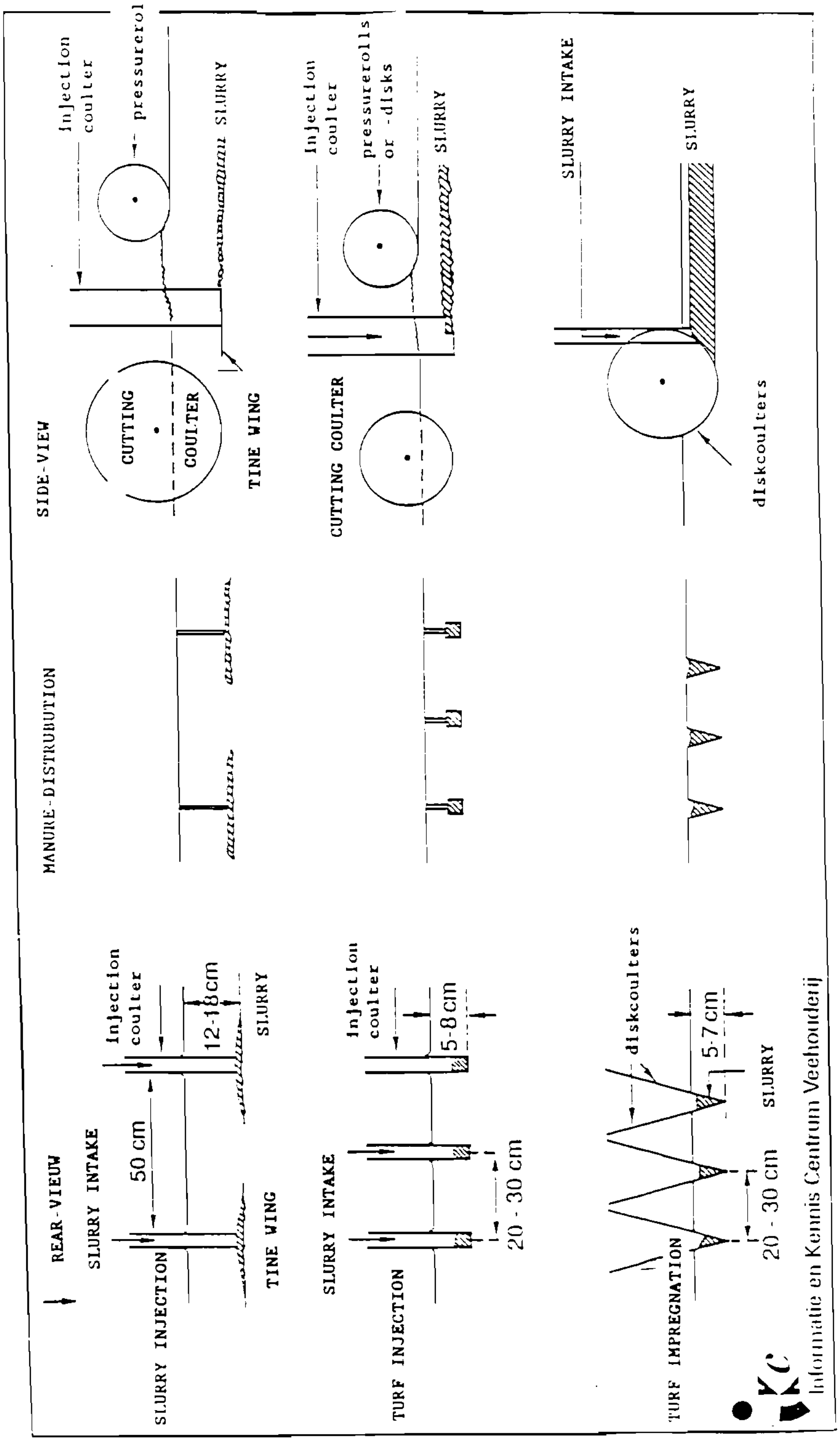


**Figure 1** Diagram of distribution of fertilizer in the soil for slurry injection, turf injection and turf impregnation.

The advantages of the systems mentioned under 2.5 and 2.6 are that they can be used on soils with poor carrying capacity and under wet conditions. Nutrients are readily available to the crops. Disadvantages, notably for the method mentioned under 6 (fertigation), are low capacity and the amount of work involved. Side-winds may hamper even distribution when the method of fertigation is used and the availability of water may also cause problems.

# 3 Ammonia emission

The Institute of Agricultural Engineering (IMAG) has carried out measurements of ammonia emissions for the different methods as described above (Klarenbeek et al, 1989). The results of these measurements can be found in Table 1. Less work is done for the method using manure spreading harrows so that the figures shown are still tentative.

**Table 1**. Ammonia emission of the different manure application methods

| Method used | Individual application per ha ($m^3$) | % ammonia emission | % emission reduction |
|---|---|---|---|
| Above ground | 15 | 20—100% (average 60%) | ----- |
| Slurry injection | 20 | 0—5% | 95—100% |
| Turf injection | 20 | 0—5% | 95—100% |
| Turf impregnation | 20 | 2—15% | 85—95% |
| Manure spreading harrows | 12 | 13—27% | 55—75% |
| Irrigation | 15 | 5—25% | 55—90% |
| Fertigation | 15 | 10—35% | 45—83% |

When the emission reduction percentages are put beside the objective formulated by the government on this point we find that only slurry injection, turf injection and turf impregnation meet the 80% reduction requirement. As these methods can be applied on a wide range by cattle farmers in the Netherlands and as these methods are designated in Dutch manure legislation as emission-poor, they will be developed further. However, this does not mean that other methods and systems will be left out of consideration. Since turf injection and turf impregnation are quite similar in terms of working methods, investments and annual costs, the two methods will be discussed under the same heading of turf impregnation.

# 4 Investments and annual costs

The first question we come across is who is to invest in the new equipment. So far it has been the agricultural contractors themselves who have put their money into these new devices. 425 turf impregnators had been bought by contractors up to September 1990. This figure will be 1170 by May this year.

At the nationally organised show of manure processing equipment held in August 1990 it was very clear that manufacturers have turned toward the production of smaller farming equipment for the individual stock farmer beside the manufacture of bigger farming machinery for the agricultural contractor. Of the 29 machines on display ten were also suitable for purchase by the smaller farmers. That is why we have looked at both categories.

First we shall give a description of the methods and the equipment used in table 2.

**Table 2.** Description of methods and equipment

| **Method** | **Equipment** |
|---|---|
| 1. Above ground | vacuum tank 10 $m^3$ with controlled tandem support and broad low-pressure tyres, tractor 65 kW |
| 2. Slurry injection | slurry injector, 6 rows of tines, covers 3m, tractor 100 kW |
| 3. Turf injection | turf injector, covers 3m, worm gear pump tanker 6 $m^3$, tractor 65 kW |
| 4. Turf impregnation | turf impregnator, covers 5.6 m, worm gear pump tanker 10 $m^3$, tractor 100 kW |
| 5. Turf impregnation | turf impregnator, covers 5.6 m, self-propelled three or four wheel vehicle, tank capacity 10 $m^3$, 180 kW |

Investments necessary for the purchase of the above farming machinery are based on their listed value (Krebbers et al., 1990). Due to rapid developments in this field, there is a wide range of types and prices. Therefore, the figures given should not be seen in absolute terms but rather serve to make comparisons.

Table 3 gives figures for investment and costs per hour for the different methods used based on the work done on a contract basis. The annual costs of the equipment covers interest, depreciation for wear and tear, maintenance, insurance and fuel. For the calculation of wages per hour costs of equipment have been augmented with labor costs (HFL 37.50 per person per hour) and costs of storage (1.5%), general costs (3.5%), management (10.25%), business risk (9%). These figures are used nationally to establish recommended rates.

**Table 3.** Survey of investments, usage, calculated costs and rates on the basis of contract work.

| Method | Replacement value | Hours of usage per year | Costs of equipment per hour | Rates of method per hour |
|---|---|---|---|---|
| 1. Above ground | 129,000 | 700 | 35 | 108 |
| 2. Slurry injection | 245,000 | 700/400[1] | 101 | 182 |
| 3. Turf impregnation small | 160,000 | 700 | 58 | 126 |
| 4. Turf impregnation large | 285,000 | 700 | 110 | 199 |
| 5. Turf impregnation self-propelled | 420,000 | 900 | 146 | 247 |

1. 700 hours for the tank and 400 hours for the injector.

Purchase by cattle farmer

When an individual cattle farmer decides to buy a turf impregnator he will have to take the above amount of HFL 160,000 (turf impregnation small) into account. However, the calculations are based on the premise that the tractor is already present on the farm which reduces investment costs to HFL 75,000. Annual costs amount to over HFL 11,600. For the tractor only the variable costs (extra maintenance and fuel) necessary for the spreading of manure are to be taken into account. These come to HFL 12 per hour (Havinga, 1990).

The number of hours of usage depends on the manure production on the farm. On a farm with 70 dairy cows spreading takes about 70 hours, 100 hours for a farm with 100 dairy cows. This amounts to roughly 1 hour per dairy cow. For this situation the total costs amount to HFL 11,600 plus HFL 12 per cow.

On a farm with 70 dairy cows this comes to HFL 12,440 (= HFL 178 per hour). On a farm with 100 dairy cows this is HFL 12,800 (HFL 128 per hour). Labour costs for the cattle farmer are left out of these calculations. Generally there will be sufficient labour capacity on the farm.

Capacity calculations

On the basis of time studies carried out by IMAG and empirical research done by IKC (Information and Knowledge Centre) standards have been set to arrive at capacity calculations (Krebbers et al. 1990). On the basis of such things as acreage, speed of transport and speed of work the capacity of the different methods can be calculated. These are shown in table 4.

**Table 4.** Calculated capacities in $m^3$ per hour for the different methods with different distances of transport (= distance between storage and plot).

| Method | Distance of transport in km | | | | | |
|---|---|---|---|---|---|---|
| | 0.0 | 0.5 | 1.0 | 2.0 | 3.0 | 4.0 |
| 1. Above ground | 31 | 26 | 24 | 20 | 17 | 14 |
| 2. Slurry injection | 33 | 27 | 25 | 21 | 17 | 15 |
| 3. Turf impregnation small | 19 | 16 | 14 | 12 | 10 | 9 |
| 4. Turf impregnation large | 32 | 28 | 25 | 21 | 18 | 15 |
| 5. Turf impregnation self-propelled | 38 | 32 | 28 | 23 | 20 | 17 |

The wider the coverage the higher the capacity. Long distances imply lower capacity. The higher capacity for method number 5 (turf impregnation, self-propelled) in comparison with method no 4 is due to the higher speed of a self-propelled vehicle. Once capacities and costs per hour are known it is quite easy to calculate the costs per $m^3$. They can be found in Table 5. Table 5 has an additional category 6 showing the costs per $m^3$ for emission-poor manure application by the individual farmer.

**Table 5.** Comparable costs in guilders per $m^3$ of emission-poor manure application methods for the different distances of transport

| Method | | Distance of transport in km | | | | | |
|---|---|---|---|---|---|---|---|
| | | 0.0 | 0.5 | 1.0 | 2.0 | 3.0 | 4.0 |
| 1. | Above ground | 3.46 | 4.15 | 4.50 | 5.33 | 6.26 | 7.49 |
| 2. | Slurry injection | 5.60 | 6.74 | 7.23 | 8.83 | 10.42 | 12.01 |
| 3. | Turf impregnation small | 6.63 | 7.88 | 9.00 | 10.40 | 12.29 | 14.18 |
| 4. | Turf impregnation large | 6.22 | 7.11 | 7.86 | 9.55 | 11.34 | 13.13 |
| 5. | Turf impregnation self-propelled | 6.50 | 7.22 | 8.82 | 10.74 | 12.35 | 14.53 |
| 6. | Turf impregnation by the individual farmer | | | | | | |
| | with 70 dairy cows | 11.79 | | | | | |
| | with 100 dairy cows | 8.26 | | | | | |

Table 5 shows that with an increase in distance of transport the price per $m^3$ rises. The more capital intensive a method is, the bigger the increase. Distance of transport 0.0 km occurs when there is in-between storage on plot and in-between transport. This involves a mobile container and a tractor with transport tank. A regular supply of manure is thus ensured. The advantages are mainly the higher capacity of the machine and the lower costs for greater distances. The farmer can also take care of the in-between transport himself should the occasion arise, in which case he keeps down his own expenses. The costs of in-between storage and in-between transport are HFL 2.61 per $m^3$. It can be deducted from Table 5 when it pays to make use of it. This is the case with distances of more than 1 km.

In comparison with the conventional above-ground application method the costs of emission-poor application rise from HFL 2.60 to HFL 4.70 per $m^3$, depending on the method used and for distances of 0.5 km. This is a considerable increase.

The costs of emission-poor application are higher for the individual farmer than for agricultural contractors, despite the fact that the costs for the individual farmer do not include wages. If wages were included, the costs for the individual farmer would increase by about HFL 2 to HFL 3 for distances of 0.5 and 3.0 km respectively.

# 5 Aspects of fertilization

As a result of lower emissions more nutrients will be available to the crops. The cattle farmer will therefore be able to save on chemical fertilizer costs. Research by Meer et al. (1990) and Boxem and Zonderland (1990) (Research station for cattle, sheep and horse husbandry, PR) has shown that between 1 and 1.5 kg of pure nitrogen extra is released from organic manure. At a price of HFL 1.20 per kg of N savings will approximate to HFL 1.50 per $m^3$ of manure. This amount does not cover the extra costs of emission-poor manure application.

# 6 Consequences for Dutch dairy farming

Application of emission-poor methods will result in higher costs for the Dutch dairy farmers. The average increase in costs will be HFL 3.50. On the other hand, emission-poor manure application will lead to chemical fertilizer savings of HFL 1.50. On balance emission-poor methods will cost HFL 2 per $m^3$ more than conventional methods.

Depending on pasture utilization and young stock densities, manure production per cow including young stock ranges from 14 to 20 $m^3$ per year. This implies extra costs of HFL 28 to HFL 40 per cow.

From the manure produced by the total of 1.9 million dairy cows in the Netherlands an estimated 85% is stored as slurry. Given a manure production of 16 $m^3$ per cow including young stock per year, total slurry output from dairy farming comes to 26 million $m^3$. This means a financial effort of HFL 52 million for Dutch dairy farming.

# 7 Conclusions

- In the future the bulk of emission-poor manure application in the Netherlands will be through turf injection and turf impregnation in particular. About 70 % of the area is suitable for these methods.

- Slurry injection will be used on light-land soils especially in spring.

- Other methods will first have to undergo technical improvements to satisfy the 80% reduction requirement.

- The equipment is known to be heavy. This prohibits application to low-lying peat soils.

- Investments in these methods are considerable, also for contractors.

- Emission-poor methods will mainly be applied by contractors because of high capital requirements.

- Despite extra costs, individual cattle farmers will also apply these methods.

- As to distances of more than 1 km, it pays to make use of in-between storage and in-between transport.

- The average costs of emission-poor manure application will increase by HFL 3.50 per $m^3$. The extra costs are partly compensated for by chemical fertilizer savings of HFL 1.50 per $m^3$. On balance the cattle farmer's income will fall by HFL 2 per $m^3$.

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# AMMONIA ABATEMENT POLICY IN THE NETHERLANDS

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## Abstract

Ammonia is a major source of acidification in the Netherlands. Agricultural ammonia emissions in the Netherlands amounts to about 224 kT per year. Dutch government has announced a policy program to abate agricultural ammonia emissions, including legal measurements. The aim is to reach in the year 2000 a total acid deposition ($SO_x$, $NO_x$ and $NH_x$) of 2400 acid equivalents per hectare per year. In 2000 the agricultural ammonia emissions will be reduced at a national level by 70%. In addition, special measures are necessary at a regional and sometimes at a local level, where high ammonia deposition peaks occur, to reach the acid deposition objective in these areas. Technical measures at all fields are necessary to reduce the ammonia emissions adequately. Total investment costs for the Dutch agricultural business run up from HFL 160 million in 1990 to over HFL 400 million in 1994. The government costs are HFL 55 million in 1990 up to 133 million in 1994 respectively.

# 1 Introduction

Ammonia is a major source of acidification in the Netherlands. Almost all ammonia emissions originate from agricultural activities. As a consequence a special program for ammonia emission reduction had to be made according to the National Environmental Policy Plans (Ministry of Housing, Physical Planning and Environment, 1989b, 1990) and as an integral part of the acidification abatement policy (Ministry of Housing, Physical Planning and Environment, 1989a). This ammonia abatement program has been formulated in the government document "Plan of Approach of Limiting Agricultural Ammonia Emissions" (Ministry of Agriculture, Nature Management and Fisheries and Ministry of Housing, Physical Planning and Environment, 1990b)

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which both the Ministers for Agriculture and Environment submitted to Parliament in October 1990.

This plan presents all the measures and actions which will be undertaken during the next years to cope with the aim of reducing the ammonia emissions in the Netherlands by an average of 70 percent in the year 2000 in relation to the emission level of the year 1980.

Together with the emission reduction objectives of the other acidification gases ($SO_x$: 80%, $NO_x$: 50% and VOC: 60%), as well as the expected emission reduction from abroad, this will lead to an average acid deposition of 2400 acid equivalents per hectare per year, which is the deposition objective for the year 2000.

# 2 Ammonia emission and deposition

For a good understanding of the ammonia problem in the Netherlands it is important to look at the spatial distribution of livestock over the country (Figure 1).

Cattle (farms) are more or less equally distributed over the Netherlands (Figure 1a). Although there are large operational differences between farms, this branch of the livestock industry is characterized by its farmland-dependent nature. In the northern and western parts of the country this is the main type of agricultural business.

The situation is different in the southern and eastern parts of the country, where, in addition to the farmland-dependent cattle industry, there is also a flourishing intensive livestock industry which is not farmland-dependent (Figure 1b).

This industry consists chiefly of swine and poultry-raising farms. However there are also firms which specialize in other species, such as mink and fox, rabbits, ducks, etc.

Currently there are about 5 million cows, 15 million pigs and 92 million chickens in the Netherlands. The total amount of ammonia emission in 1980 was about 224 million kg ammonia ($NH_3$); in 1986 it was slightly higher, about 228 kT. The contribution of the various sources to the emission in 1980 has been given in Table 1.

Fig. 1a

Fig. 1b

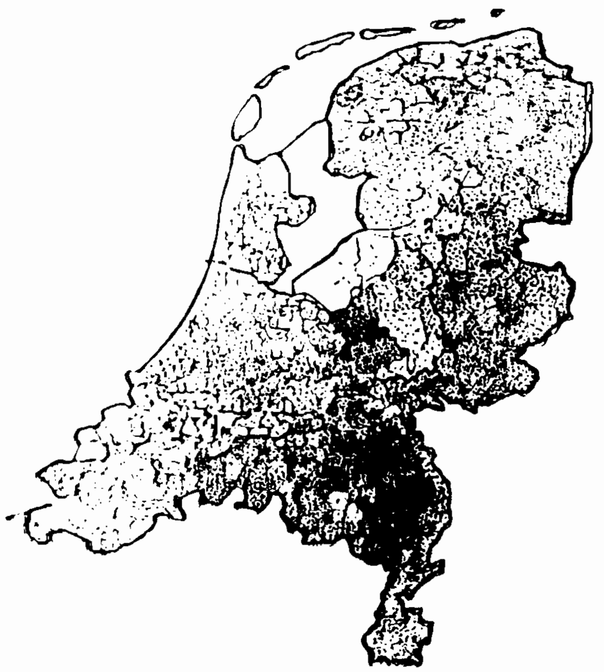

Figure 1. Spatial distribution of livestock in the Netherlands;
Fig. 1a: Dairy cows and calf cows
Fig. 1b: Intensive livestock industry (pigs, laying hens and broilers)
Source: Min. of Agriculture, Nature Management and Fisheries, Landbouwtelling, May 1982

Table 1. Ammonia emissions (tons/$NH_3$/year) in the Netherlands in 1980 for different types of animals (Ministry of Housing, Physical Planning and the Environment, 1987)

| Animal | Stable + storage | Application | Meadow | Total | % |
|---|---|---|---|---|---|
| Cattle | 43,000 | 71,000 | 27,000 | 141,000 | 63 |
| Pigs | 20,000 | 31,000 | -- | 51,000 | 23 |
| Poultry | 19,000 | 9,000 | -- | 28,000 | 12 |
| Sheep | -- | -- | 4,000 | 4,000 | 2 |
| Total | 82,000 | 111,000 | 31,000 | 224,000 | 100 |
| Percentage | 36 % | 50 % | 14 % | 100 % | |

In absolute terms cattle industry has the highest ammonia emission, accounting for about 63 percent of the total. The sources of emissions are distributed relatively equal over the country. The second large source (35%) is the intensive livestock industry (pigs and poultry). These sources, however, are highly concentrated in relatively small areas.

As a consequence of the unequal spatial distribution of the livestock farms, the regional differences in ammonia deposition are very large. The resulting ammonia deposition (mol $NH_3$/hectare/year) for the year 1986 is given in Figure 2.

On the receptor side, not all soil types are equally sensitive to acidification. In Figure 3 an overview has been given of the parts of the country which are sensitive for acidification (Ministry of Housing, Physical Planning and Environment, 1989a).

Problems arise primarily in the southern and eastern parts of the Netherlands. In these areas both cattle industry and intensive livestock industry are responsible for a concentrated network of ammonia emitting sources. This situation, coupled with a soil type which is extremely sensitive to acidification, leads to serious acidification damage to forests and natural areas.

Recent research (RIVM, 1991) has indicated that due to a higher deposition speed of ammonia, the regional differences are even larger than originally assumed. The heavily polluted areas seem to be more polluted by ammonia, and the relatively less polluted areas have a lower deposition of ammonia. The consequence of all this is that an average 70 % emission reduction over the country as a whole is more than adequate to prevent damage in the northern and western parts, but in the south and mid-east regions it falls far short.

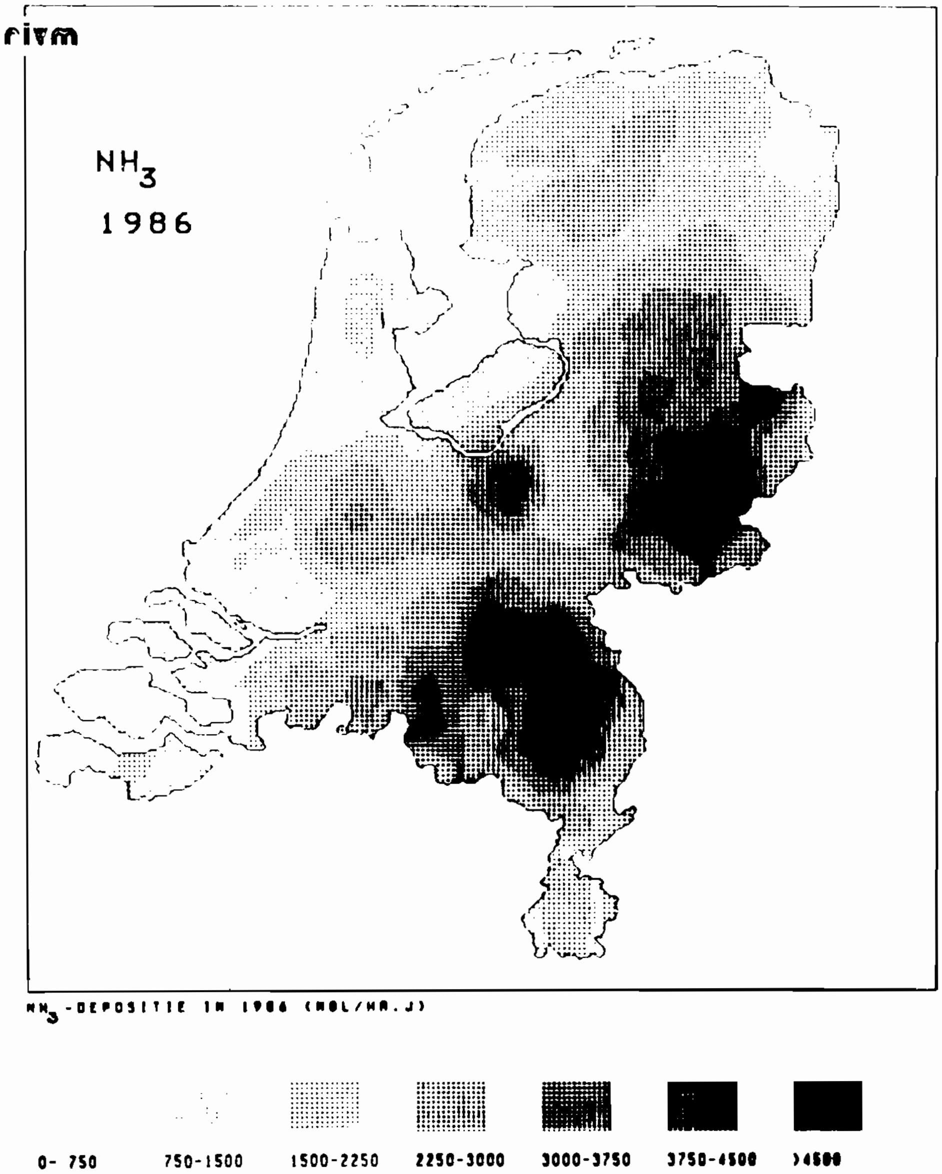


Figure 2. Ammonia deposition in the Netherlands in 1986 (mol $NH_3$/hectare/year)

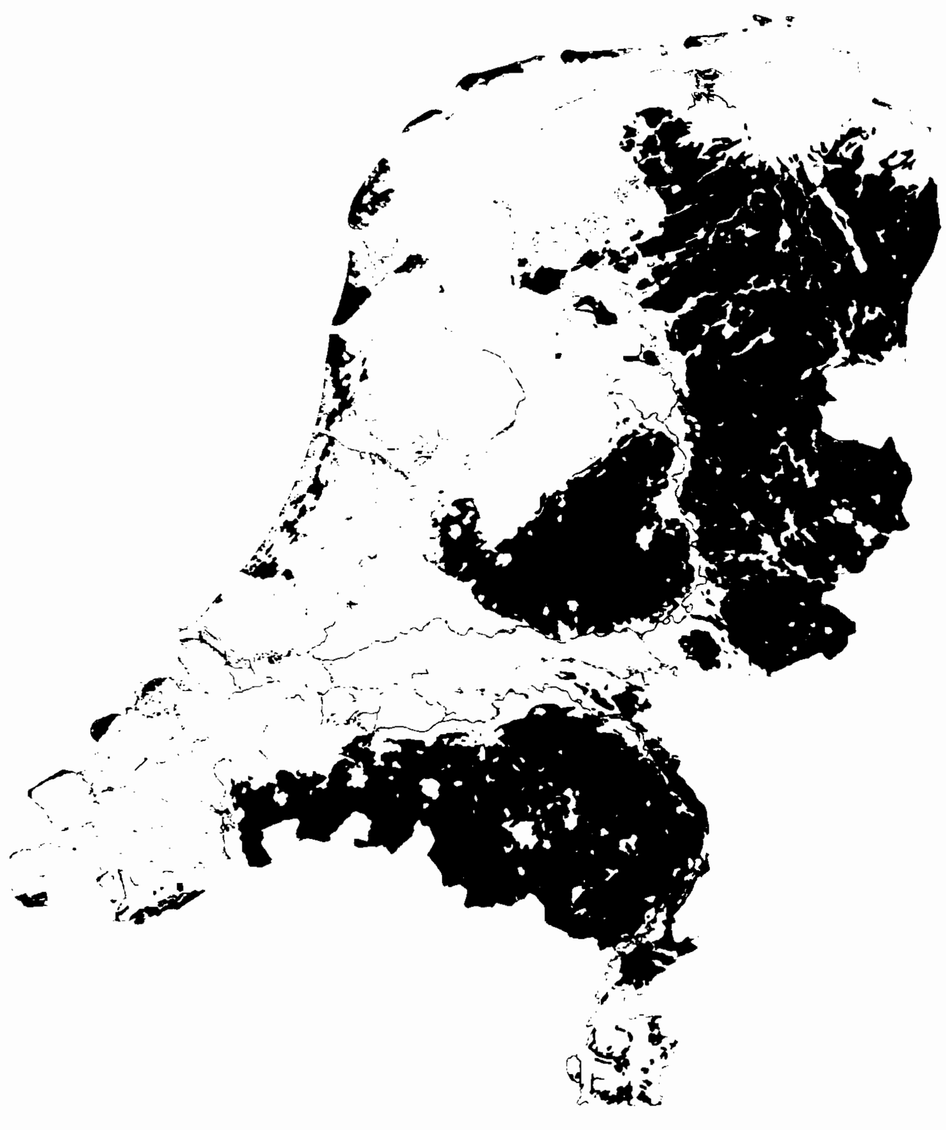

Figure 3. Overview of regions (dark areas) in The Netherlands which are sensitive to acidification
Source: Ministry of Housing, Physical Planning and the Environment (1989a)

In Figure 4 the regions have been indicated where in the year 2000, after execution of the generic ammonia abatement measures, the acid deposition objective of 2400 acid equivalents per hectare per year will still be exceeded.

So the ammonia problem can only be partially tackled with the formulation of an solely general emission reduction objective (i.c. the average 70% emission reduction) for the country as a whole. Unfortunately, this was not realized at first. In addition to the generic actions for controlling ammonia emissions, which must be taken by all livestock companies, a regional approach to the problem is absolutely necessary.

# 3 The Dutch ammonia abatement policy

## 3.1 Introduction

The current Dutch ammonia policy (October 1990) can be described as a sort of three step approach:

a. **generic policy**, aiming at an average emission reduction of 70 percent for The Netherlands;
b. **regional policy**, designed to abate high levels of regional deposition;
c. **object-oriented policy**, with the purpose of abolishing local peaks in deposition.

## 3.2 The generic policy

The ammonia produced by the livestock industry is not only emitted by stables, manure storage silos and cattle pastures but it is also formed by the spreading of manure and fertilizer as well as the processing of manure. In order to reduce the ammonia emission measures must be taken in all of these areas as well as in the area of animal feeding.

Measures relating to the spreading of manure on the land are very important. In the first place because it is the major source (50% of the total emission). In the second place because the measures taken in other areas will be less significant, since the nitrogen components (ammonia) will still evaporate. So measures must always be taken in combination to each other.

It is also important to guard against shifting the problem from one environmental compartment to the other. This may occur especially when manure with a higher nitrogen content than before, due to emission prevention measures, has been added to the soil in quantities exceeding the need of the plant, after which it can wash away as nitrate.

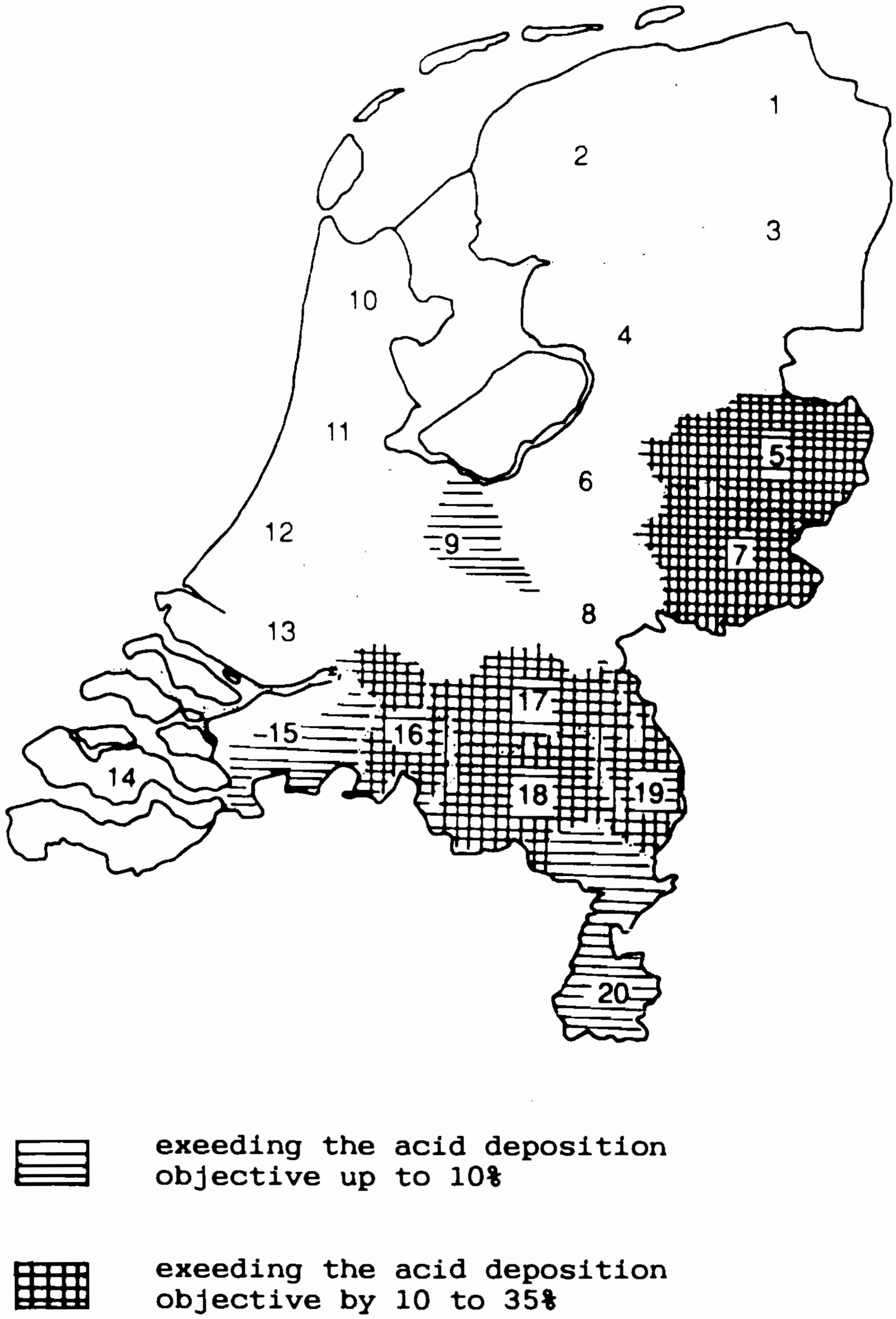


Figure 4. Regions in The Netherlands which are sensitive to acidification where in the year 2000 after execution of the generic ammonia policy (70 % emission reduction) the acid deposition objective (2400 mol acid equivalents/ha/year) will be exceeded.

Within the generic policy measures may be taken at the following fields:

a. Feed
b. Housing and storage
c. Pasturage
d. Use of manure

In the following paragraphs the technical possibilities and consequently the Dutch policy measures within these fields will be described briefly.

### 3.2.1 Feed

The excretion of nitrogen (and emission of $NH_3$) via manure and urine can be limited considerably by tuning the feed composition more closely to the animals' needs. This can be accomplished by reducing the nitrogen content of the feed or by what is called "multi-phase feeding".

Multi-phase feeding, where two or more different kinds of feed are provided, has already been used rather widely in the Dutch intensive livestock industry. Reduction of the nitrogen content of feed is still in the development phase, however. How successful this will be depends largely on the possibilities for replacing certain components with synthetic amino acids. But this is a very expensive option, and for the moment a stumbling block in the way of more general application.

Cattle receive roughage as well as feed concentrates. Nitrogen utilization can be optimized by means of supplementary feeding with high energy products containing low protein (e.g. sugar beet pulp, potato fibers) and by decreasing the fertilizer dosage to grassland, thereby also reducing the nitrogen content of the grass subsequently eaten by cattle. This limits $NH_3$ emissions from the manure during both the stall and pasture periods. It currently appears that it may be possible to limit the nitrogen input by something like 10 to 30 percent, depending on the type of livestock farm in question.

By the government Plan of Approach the application of multi-phase feeding will be stimulated both by advising and subsidizing the extra required feedstores and feed dose apparatus.

### 3.2.2 Housing and storage

Reducing ammonia emissions from farm buildings is not easy, because of the diversity of animals and housing systems.

A great deal of research is being carried out in the Netherlands nowadays to develop housing systems with a low ammonia emission.

In September 1990 the results of an extensive study on the mechanism of the ammonia production in stables were presented (Ministry of Agriculture, Nature Management and Fisheries, 1990a). For cattle, swine and poultry stalls the different existing housing systems were analysed and the possible solutions and techniques were evaluated by four aspects: environment, health, animal welfare and agricultural aspects.

Most present-day housing systems in the Netherlands have a manure storage under a slatted floor. It is evident that this storage cannot be covered. It is believed however that the $NH_3$ emission from stalls can be reduced considerably (>50%) by controlling the factors that cause the formation of ammonia. In general these factors are: limiting the amount of time that the manure remains in the stall, keeping the floors as dry and manure-free as possible, drying the manure quickly, minimizing the length of time during which the manure is in contact with the air, and cooling or adding acid to manure slurry rapidly.
For dairy cows stalls the following options seems to be promising:

- reducing the length of time during which the manure is in contact with the air;
- removing the manure regularly from the stall to a (closed) storage tank, e.g. by stable washing and scraping systems;
- using additives such as saltpeter or sulfuric acid which suppress the formation of $NH_3$.

Systems comparable to the ones just mentioned above can also be considered in the swine industry. However, an important difference is that stalls in this sector are often equipped with a forced air ventilating system, which makes it possible to make use of either biological or chemical air cleaning systems. If they are well designed and maintained, these systems can reduce $NH_3$ emissions considerably (by more than 80 percent).

Another system which looks rather promising for the pig industry are the so called deep litter systems. These systems are based on the decomposition of the manure within a thick layer of very fine woodchips.

The development of low-emission stall systems in the poultry industry is already quite advanced due to the manure and stall climate problem. Ammonia emissions can be reduced by:

- rapid drying of the manure produced in the stall, on the manure belt if possible;
- rapid removal of the manure from the stall;
- keeping the straw on the stall floor (the presence of which depends on the stall system) as dry as possible;
- ventilation air cleaning (chemical or biological).

For the broiler industry the first low emission housing system with an estimated ammonia reduction of about 80-90% has already been commercially available.

For laying hens a new technique has been developed which can be used in existing stall systems, the so-called ventilation tunnel drying system. In this system wet poultry manure is removed from the stall every day. Stall ventilation air (which may or may not be preheated) is used to dry the manure to a dry substance percentage of about 60%-70% within a 24-hour period. In theory this system can reduce $NH_3$ emissions from stalls by at least 90 percent.

The straw in poultry stalls with a so-called ground housing system must be kept as dry as possible in order to limit $NH_3$ emissions. Floor insulation and floor heating also appear to offer potential.

In the government Plan of Approach legal measures have been announced by July 1, 1994. On the basis of the Nuisance Act a General Administrative Order is being prepared in which maximum ammonia emission factors per animal place will be ordered. This legal instrument may fail to come in force if the policy evaluation in 1992 makes evident that alternative policy instruments are available.

At the latest in the year 2005 all housing systems have to meet the requirements from the General Administrative Order. The extra capital cost for low emission systems will be subsidized.

### 3.2.3 Manure storage

$NH_3$ emissions from manure storage tanks may be limited by covering the stores. Research has shown that a right kind of covering system can reduce emissions from manure storage by at least 80 percent. Accumulation of hazardous or potentially explosive gas mixtures under the cover, such as hydrogen sulfide and methane ($H_2S$ and $CH_4$) must be prevented by maintaining a small gap in the cover. Covering manure storage facilities also has the benefit of keeping rainwater out of the manure. This reduces the necessary volume of the facility, which is an additional advantage.

On the basis of the Nuisance Act, recently a General Administrative Order has been enacted in which it is ordered that all liquid manure stores which were built after July 1, 1987 must be covered by January 1, 1992.
The government Plan of Approach announced that by the year 2000 all liquid manure storage tanks built before July 1, 1987 must be covered.

The covering of existing and new manure stores will be subsidized until January 1995.

### 3.2.4 Pasturage

Theoretically pasture emissions can be limited by reducing the period of time during which cattle is in the pasture. However, given current Dutch practices with regard to feeding, housing of livestock, and storage and use of manure, reducing the pasture period would actually increase $NH_3$ emissions unless abatement measures were taken in other areas. This is because the composition of feed is different in the pasture. Because pasture emissions account for only a relatively small share (14%) in the total $NH_3$ emissions in the Netherlands, measures in this area are not being contemplated. Nor do pasture-specific measures appear to have much relevance in other countries.

### 3.2.5 Use of manure

The $NH_3$ emissions which are generated during the application of manure onto the land can be abated effectively by keeping the period of time during which the manure is in contact with the air as short as possible. This can be done by:

- spreading manure only when the weather is cool and dark and when showers are approaching;
- putting the manure directly into the soil by injection, sod fertilization, sod injection or direct ploughing down;
- water spraying after spreading the manure;
- sprinkling or drenching (diluted) manure on grassland.

Shifting of the nitrogen problem from the air to the soil and/or surface waters (nitrate washing) must be guarded against. However, from an agricultural perspective there is an advantage in that crop and grass can utilize the nitrogen better, provided it is applied at the right time. This can lead to savings in the use of fertilizer.

The $NH_3$ emissions generated by the spreading of manure can be reduced by about 90 percent with low-emission application techniques.

More than 50% of the total amount of $NH_3$ emission in the Netherlands come from manure application, so low-emission application is one of the most effective abatement measures. In addition, low-emission application can prevent the evaporation of ammonia captured in the manure as a result of abatement measures in other areas.

The Plan of Approach comprises a phased introduction of low-emission techniques for grass and farmland. The periods during which the application of manure is prohibited differ according to the nature of the soil.

The ultimate target is to apply the manure in the growing period with a low-emission technique.

The prohibition period has been formulated in a General Administrative Order on the basis of the Soil Protection Act. In this Act also maximum allowable gifts of manure (in the form of phosphate) are prescribed. These requirements are in force since May 1, 1987 and will become more stringent in three phases. The second phase is the period 1991-1995.

The prohibition period for the application of manure in this period has been given in Figure 5. The investment in low-emission apparatus for manure application has been subsidized.

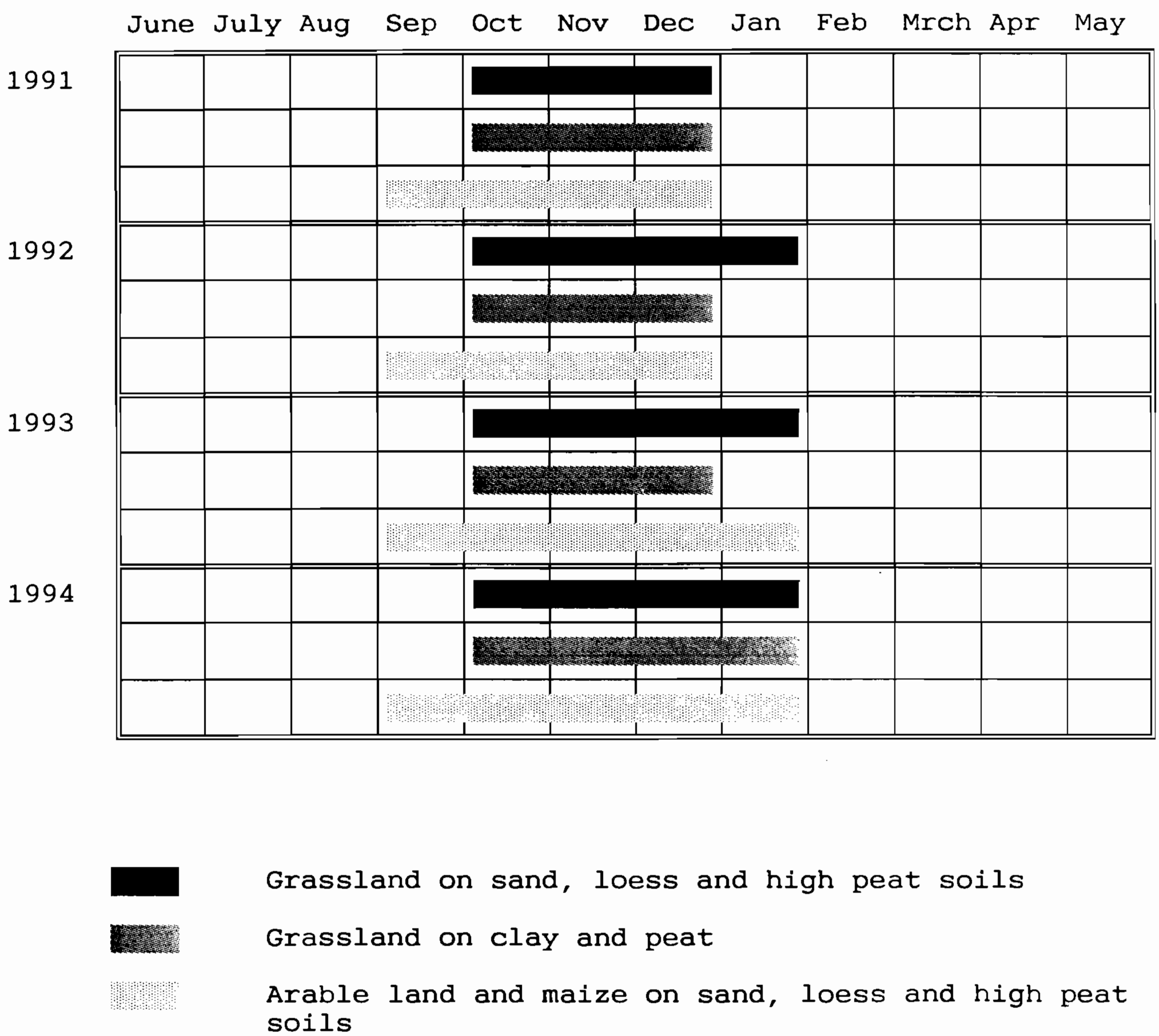


Figure 5. Manure application prohibition period for the years 1991-1995 (Amending Act Decree on Manure Use).

## 3.3 The regional policy

One of the difficulties in developing the ammonia policy is that execution of the generic policy more or less exhausts the technical possibilities for abating emissions at the source. This means that there must be a search for measures in the fields of land use planning, land development, socio-economic policy and future regulations concerning the relocation of manure production rights. Also a study takes place for more structural measures which might be introduced on a regional level. It is already clear that relocation of firms will be the answer in many cases. The government is thus faced with the question of how this can be accomplished without incurring enormous costs. In short: how can the agricultural firms be stimulated to undertake action themselves, that is, to find another site for their operations? This relates mainly to the non-land-dependent livestock industry. Should this policy prove inadequate, then volume measures in the livestock sector might be unavoidable.

## 3.4 The object-oriented policy

Despite execution of the generic policy - and if possible the regional ammonia policy - a significant percentage of Dutch forests and natural areas are still not being protected adequately. Extremely high deposition of ammonia occurs especially in situations where livestock industries are located in the middle of forest and/or natural areas. In many cases this deposition amounts to more than 10,000 mol $NH_3$ per hectare per year. This extremely high deposition can be abated by drastic technical measures in some cases. However in most situations relocation or termination of activities will be the only answer. It is estimated that some hundreds of locations in the Netherlands are eligible for this policy.

The Plan of Approach announced for the short term is the development of an enforcement program, including the legal instrument. It is expected that the execution of object-oriented policy will start in 1992.

# 4 Costs of ammonia abatement

## 4.1 Total costs for the agricultural business

The costs associated with the abatement of ammonia are substantial. The generic policy alone requires an investment for the agricultural business of 160 million guilders in 1990 running up to 400 million in 1994.

Little can yet be said about the costs of the regional policy. However it is expected that it will be substantial, especially if firms have to be relocated or volume measures taken.

In Table 2 the estimated investment costs for the livestock industry have been specified according to the different fields of application.

Table 2. Investment cost for livestock industry

| | Initial investments in million Guilders | | | | |
|---|---|---|---|---|---|
| | 1990 | 1991 | 1992 | 1993 | 1994 |
| Multi-phase feeding | 20 | 30 | 40 | 50 | 60 |
| Low emission stalls | 5 | 10 | 20 | 40 | 75 |
| Manure application | 50 | 90 | 100 | 80 | 80 |
| Manure storage | 85 | 66 | 112 | 178 | 185 |
| Other | p.m. | p.m. | p.m. | p.m. | p.m. |
| Total | 160 | 196 | 272 | 348 | 400 |

## 4.2 Costs for the government

For the stimulation of environmental investments at farm level the Dutch government placed the following amounts of subsidies at the disposal of the Dutch agricultural business:

| Year | Million guilders |
|---|---|
| 1990 | 40 |
| 1991 | 50 |
| 1992 | 72 |
| 1993 | 104 |
| 1994 | 118 |

In the Agricultural Structure Memorandum (Ministry of Agriculture, Nature Management and Fisheries, 1990c) the following amounts have been mentioned on behalf of the stimulation of industrial manure processing:

| Year | Million Guilders |
|---|---|
| 1990 | 45 |
| 1991 | 61 |
| 1992 | 77 |
| 1993 | 93 |
| 1994 | 115 |

For the execution of the object-oriented policy a total amount of 70 million guilders for the period 1990-1994 is available. Annually this is an amount of about 16 million.

## Acknowledgement

I thank my colleague Dipl. Ing. Henk Hannessen for his support and especially for supplying me with some basic information originating from the report he drafted on behalf of the Economic Commission for Europe of the United Nations (ECE), titled "Reducing Ammonia Emissions through the Implementation of Measures in the Livestock Industry" (Hannessen, 1990).

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# POSSIBILITIES AND COSTS OF CONTROLLING AMMONIA EMISSIONS IN EUROPE

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## Abstract

This paper presents a submodule which computes the costs of controlling ammonia emissions in 27 European countries. The submodule will be incorporated into the RAINS (Regional Acidification INformation and Simulation) model. Abatement options are low nitrogen feed, stable adaptations, covering manure storage, biofiltration and low ammonia applications of manure. Cost estimates are based on country-, animal-, and technology specific data such as stable size, fertilizer price, manure production per animal and the investments per animal place. Some example results are shown for the year 2000 for Finland and the Netherlands. They suggest that ammonia emissions in Finland could be reduced by 30% over the 1980 without costs. A similar reduction in the Netherlands would cost 76 million DM per year. The maximum feasible reduction of the unabated emissions in 2000 are 60% in Finland and 65% in the Netherlands.

# 1 Introduction

Acidification of the environment caused by atmospheric pollution is one of the major environmental problems in Europe. Not only sulphur compounds but also nitrogen compounds contribute to acidification in the form of nitrogen oxides ($NO_x$) and ammonia ($NH_3$). The Regional Acidification Information and Simulation (RAINS) model developed at IIASA combines information on several stages of the acidification processes in the environment: the sources of emissions and the potential for their abatement, the atmospheric transport and the environmental effects of acid deposition (Alcamo et.al., 1990). Since the RAINS model is designed as a tool for the assessment of the efficiency of different pollution control strategies, the analysis of removal potential and the associated control costs forms an essential part of the model. At present cost functions for controlling $SO_2$ emissions as well as for $NO_x$ emissions are incorporated in the model. Potential and costs of control of $NH_3$ emissions, however, have not yet been incorporated.

The aim of this paper is to describe the costs of control model for $NH_3$ emissions and to give some examples of results. A detailed description can be found in Klaassen (1991a). In contrast to the cost estimates available for controlling sulphur and nitrogen emissions, the cost estimates for the abatement of ammonia emissions are more uncertain, at least for specific control options such as stable adaptations, due to a lack of practical experience. The requirement to assess the abatement costs for all 27 countries of Europe necessarily limits the level of detail which can be maintained. Although cost estimates are based on recent information, data and computational constraints require simplifications, which might appear to be too crude for studies focusing on one country. Therefore the results of the costs sub module should be seen as comparative rather than absolute cost estimates: the emphasis is put on international consistency and comparability.

Major sources of ammonia emissions are livestock farming, consumption of nitrogen fertilizer and industry (Klaassen, 1991b). Options are available to control ammonia emissions from livestock farming and industry. Ammonia from livestock farming is released during three basic processes (Figure 1):

- in the stable and during storage of manure,
- during the application of manure,
- in the meadow period.

For each of these processes techniques are available to control ammonia emissions. In addition, changes in the nitrogen content of the feed influence emissions of all three processes. The following options can be distinguished to control the ammonia emissions from livestock farming (e.g. Baltussen et al. 1990a):

* changes in the nitrogen content of the fodder (such as multiple stage foddering)
* adaptations during stable and storage of manure:
  - stable adaptations (such as manure flushing)
  - covering manure storage
  - cleaning of stable air (biofiltration or scrubbing)
* low ammonia application (e.g. direct ploughing down of manure).

Changing the nitrogen content of the fodder affects the ammonia emissions of all three processes: stable and storage, application and in the meadow (Figure 1). Adaptations of stable and storage affect both stable plus storage as well as emissions during application since the nitrogen content of the excretion after the stable emission may increase. Table 1 presents the options distinguished in RAINS. Including the combinations of the various abatement techniques 48 different options are available. All combinations which are possible, as well as the reductions in emissions of these techniques, are presented in Klaassen (1991a).

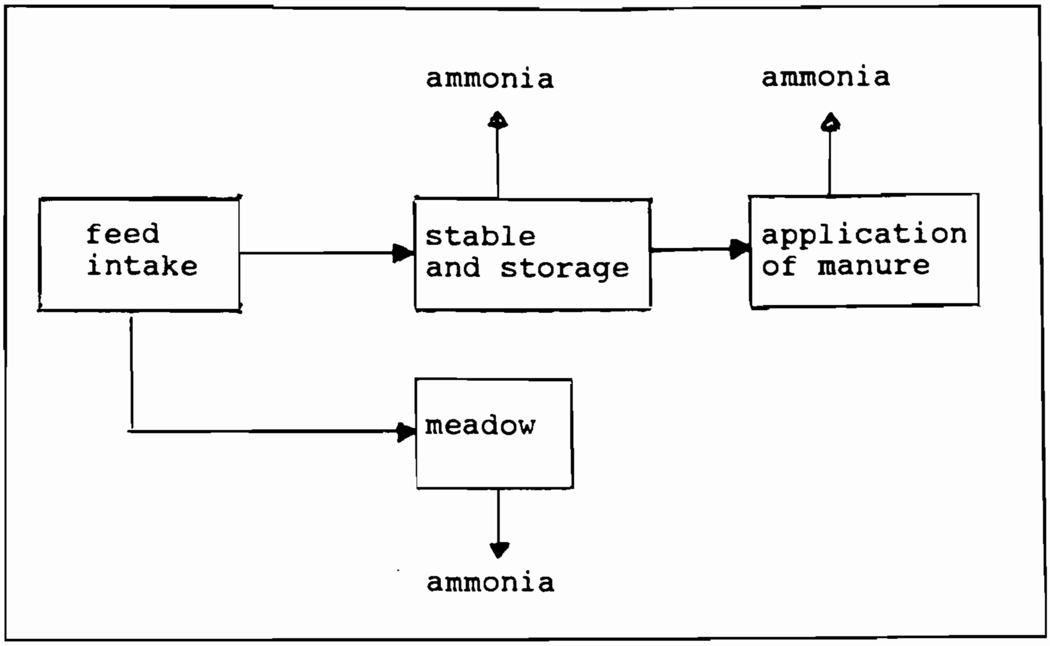


**Figure 1.** Ammonia losses from livestock farming

Table 1. RAINS abatement options for ammonia emissions

| LIVESTOCK FARMING: | | | | | | |
|---|---|---|---|---|---|---|
| OPTIONS PER PROCESS | FODDER | STABLE AND STORAGE | | | APPLICATION | TOTAL NUMBER OF OPTIONS |
| | low N-fodder | stable adaptation | closed storage | biofil-tration | low $NH_3$ application | |
| | (LNF) | (SA) | (CS) | (BF) | (LNA) | (number) |
| ANIMAL TYPE | | | | | | |
| dairy cows | x | x | x | | x | 11 |
| other cattle | | | x | | x | 3 |
| pigs | x | x | | x | x | 11 |
| laying hens | x | x | | x | x | 11 |
| broilers | x | x | | x | x | 11 |
| sheep | | | | | | |
| horses | | | | | | |
| | | | | | | 47 |
| INDUSTRY: | | | | | | |
| Stripping/absorption | | | | | | 1 |

In several branches of the chemical industry emission reductions of 95% can be achieved. This is possible through the application of stripping and absorption techniques (Technica, 1984). For the total chemical industry causing ammonia emissions the average reduction that can be achieved is set at 50%.

# 2 Low nitrogen feed, stable adaptations, covering manure storage and cleaning of stable air

## 2.1 Introduction

Low nitrogen feed is a combination of various techniques to reduce emissions such as:

- reductions in the level of nitrogen application on grassland or the substitution of grass by silage for dairy cows (Baltussen et al., 1990b; Spiekers and Pfeffer, 1990),
- reductions in the nitrogen content of feed through an improved agreement between the amino acids in the diet and the amino acid requirements of animals (multi-phase feeding) or through changes in the composition of the raw materials and supplementing diets with synthetic amino acids for pigs and poultry (Baltussen et al., 1990a; Lenis, 1989: Spiekers and Pfeffer, 1990),

For various animal categories stable adaptations are possible which reduce the escape of ammonia (Baltussen, 1990a; Oosthoek et al. 1990a). $NH_3$ emissions from stalls can be reduced by limiting the time that manure remains in the stable (e.g. by using manure flushing systems), keeping floors as dry and free of manure as possible, drying manure quickly, minimization of the time during which ammonia is in contact with air or adding acid to manure. Covering storage of manure is another way to prevent the escape of ammonia during the stable and storage period. A third option to control the emissions from the stable is the application of various techniques that clean the stable air: biological filters, biological scrubbers or chemical scrubbers.

## 2.2 The algorithm

The algorithm used in the cost calculation routine includes technology- and animal-specific, as well as country-specific, factors for comparing the costs of abating ammonia emissions per country (see Table 2).

Table 2. Parameters used in the cost calculation routine

| Technology (and animal) specific parameters | |
|---|---|
| cif, civ | parameters of the investment functions |
| fk | annual fixed (maintenance) costs |
| lt | lifetime of the installation |
| Qf | fodder use per animal |
| Qg | heating fuel use |
| Ql | labor use |
| Qw | water use |
| Qe | electricity use |
| Qd | disposal of waste |
| ar | number of animal rounds per year |
| sb | capacity utilization factor |
| cf | fodder price (increase) |
| cg | heating fuel price |
| cl | labor price |
| cw | water price |
| cd | disposal price |
| cfma | fixed costs manure application |
| cvma | variable costs manure application |
| cfmi | fixed cost manure injection |
| cvmi | variable costs manure injection |
| cfms | fixed costs sod manuring |
| cvms | variable costs sod manuring |
| cfmr | fixed costs manure sprinkling |
| cvmr | variable costs manure sprinkling |
| xs | removal efficiency stable |
| xa | removal efficiency application |
| xm | removal efficiency meadow |
| **Country specific parameters** | |
| ce | electricity price |
| ck | fertilizer price |
| Mi | manure production per animal |
| Qma | share manure ploughed down |
| Qmi | share manure injected |
| Qmr | share manure sprinkled |
| Qms | share sod manuring |
| Qmh | volume of manure per hectare |
| ssd | stable size dairy cows |
| sso | stable size other cattle |
| ssp | stable size pigs |
| ql | interest rate |

### 2.2.1 Investment costs

The following description uses the indices i, k, l to indicate the nature of the parameters:

- i the type of animal
- k the control technology
- l the country

The investment function describes the investment costs of the control technology as a function of the stable size:

$$I_{i,k,l} = ci^{f}_{i,k} + \frac{ci^{v}_{i,k}}{ss_{i,l}} \tag{2.1}$$

In which $ci^{f}_{i,k}$ and $ci^{v}_{i,k}$ are the coefficients of the investment function and $ss_{i,l}$ is the number of animal places per stable.

The investment costs are annualized over the lifetime lt of the installation using the interest rate $q_l$:

$$I^{an}_{i,k,l} = \frac{I_{i,k,l} \times (1+q_l)^{lt} \times q_l}{(1+q_l)^{lt}-1} \tag{2.2}$$

### 2.2.2 Fixed operating costs

Fixed operating costs may comprise of maintenance, insurance and administrative overhead. They are presented as a fixed percentage $fk_{i,k}$ of the investments per animal place:

$$OM^{fix}_{i,k,l} = I_{i,k,l} \times fk_{i,k} \tag{2.3}$$

### 2.2.3 Variable operating costs

Variable operating costs may consist of the following elements:

- increase in feed costs per animal due to higher prices of low nitrogen feed,
- costs of natural gas use,
- electricity use,
- water use,

- labor use,
- waste disposal costs.

These variable costs are presented as costs per delivered animal:

$$OM^{var}_{i,k,l} = Q^f_i \times c^f + Q^g_i \times c^g + Q^l_i \times c^l + Q^w_i \times c^w + Q^e_i \times c^e_l + Q^d_i \times c^d \tag{2.4}$$

| | |
|---|---|
| $Q^f_i$ | the quantity of feed per animal |
| $c^f$ | the price (increase) of feed |
| $Q^g_i$ | the quantity of natural gas per animal |
| $c^g$ | the price (increase) of natural gas |
| $Q^l_i$ | the quantity of labor per animal |
| $c^l$ | the price of labor |
| $Q^w_i$ | the quantity of water per animal |
| $c^w$ | the price of water |
| $Q^e_i$ | the quantity of electricity per animal |
| $c^e_l$ | the price of electricity |
| $Q^d_i$ | the quantity of waste per animal |
| $c^d$ | the price (increase) of waste disposal |

### 2.2.4 Unit costs of $NH_3$ control

Based on the above mentioned items the unit costs for the control of $NH_3$ emissions can be calculated. Unit costs are expressed in costs per animal per year by taking into account the number of animal rounds per year $ar_i$ and the utilization factor of the capacity $sb_i$:

$$ca_{i,k,l} = \frac{(I^{an}_{i,k,l} + OM^{fix}_{i,k,l})}{sb_i} + \frac{OM^{var}_{i,k,l} \times ar_i}{sb_i} \tag{2.5}$$

The cost efficiency of the abatement option can only be evaluated if the annual costs are related to the amount of emissions reduced in order to obtain the cost per unit of $NH_3$ removed. In doing so it has to be taken into account that abatement options may simultaneously reduce emissions during stable and storage, application and in the meadow:

In which:

$nh3s_{i,l}$ emission coefficient of stable

$$cn_{i,k,l} = \frac{ca_{i,k,l}}{nh3s_{i,l} \times xs_{i,k} + nh3a_{i,l} \times xa_{i,k} + nh3m_{i,l} \times xm_{i,k}} \quad \textbf{(2.6)}$$

| | |
|---|---|
| $nh3a_{i,l}$ | emission coefficient of application |
| $nh3m_{i,l}$ | emission coefficient meadow |
| $xs_{i,k}$ | efficiency of reduction stable |
| $xa_{i,k}$ | efficiency of reduction application |
| $xm_{i,k}$ | efficiency of reduction meadow |

## 2.3 Low nitrogen feed

Low nitrogen feed is a combination of various techniques to reduce emissions:

- reductions in the level of nitrogen application on grassland in combination with an increase in silage maize for grazing cattle (dairy cows),
- reductions in the albumen content of concentrate feed through changes in the composition of the raw materials and supplementing synthetic amino acids or as a result of directing the feed composition to the specific demand for amino acids (multi-phase feeding) for pigs and poultry.

For dairy cows nitrogen excretion can be lowered if the level of nitrogen application on grassland is reduced from 400 or even 500 kg nitrogen per ha to 200 kg nitrogen per ha and grass silage is partly substituted by silage maize, according to Baltussen et al. (1990b) for the Netherlands. Their calculations show that reductions in stall emissions by 10 to 30 % and in meadow emissions of around 25 % are possible. Spiekers and Pfeffer (1990) indicate that a reduction in the nitrogen excretion would be possible of 10 to 15 %. Whether this is an alternative that is possible in other European countries, with the exception Denmark and the Federal Republic of Germany, is uncertain since levels of nitrogen application of grassland in other European countries are generally far below the level in the Netherlands. Consequently, the user of RAINS is allowed to limit the potential applicability of this alternative.

For pigs multi-phase feeding, in combination with nitrogen poor feed or synthetic amino acids, reduces the nitrogen in the excretion by 5% for fattening pigs and 20% for sows (Baltussen et al., 1990c). Spiekers et al. (1990) even suggests that reductions of up to 35% are possible for fattening pigs and 15% for sows. Lenis (1989) is of the opinion that synthetic amino acids may achieve reductions of 25% for both pigs and sows on the long run.

For laying hens a reduction in the albumen content may reduce the nitrogen excretion by some 10 per cent. Multi-phase feeding and synthetic amino acid are expected to reduce the nitrogen excretion for broilers by 20 per cent (Horne, Van 1990).

Only for pigs the introduction of low nitrogen feed is associated with investment costs. For all other animals costs only consist of higher feed prices. The technology and animal specific data are presented in Klaassen (1991a). Data are based on Baltussen et al. (1990a, 1990b, 1990c) and Van Horne (1990). The investment costs are annualized over the lifetime lt of the installation using the interest rate $q_l$. There are no fixed operating cost. Variable operating costs consist of the increase in feed costs per animal due to the higher prices of low nitrogen feed. These costs are based on changes in the composition of raw materials for feed production for the situation in the Netherlands. Results for the Federal Republic of Germany (Spiekers et al., 1990) however show that the cost increases for pigs in the Netherlands and the Federal Republic of Germany are approximately as high.

## 2.4 Stable adaptations

For various animal categories low emission stable systems are possible which prevent the escape of ammonia. $NH_3$ emissions from stalls can be reduced by limiting the time that manure remains in the stable, keeping floors as dry and free of manure as possible, drying manure quickly, minimization of the time during which ammonia is in contact with air or adding acid to manure (Hannessen, 1990). The preliminary cost estimates used in this study are based on the following systems:

- dairy cows : stable washing and scraping systems, removing manure regularly to a (closed) storage basin
- pigs : manure flushing and scraping systems
- laying hens : manure belt with forced drying of manure
- broilers : forced drying of manure on slatted, littered floors

For most of these systems, especially for pigs and cattle, cost estimates are uncertain since practical experience hardly exists. Therefore the estimates are preliminary.

Washing the stable floor of dairy cow stables and frequently removing the manure to a closed storage system, can reduce ammonia emissions by 50 to 70% (Oosthoek et al., 1990a). Costs consist of the washing system in combination with manure storage capacity (Baltussen, 1990b). Annual costs are still uncertain and therefore accounted for as fixed percentage of the investment (Klaassen, 1991a). For pig stables, Oosthoek et al. (1990a, 1990b) conclude that the reduction in ammonia emissions that can be achieved is 60 to 70%, using a manure flushing system in combination with a replacement pump or drainage system in the stable. Provisional cost estimates were made by Baltussen et al. (1990c) and Hakvoort and Paques (1990). The application of a manure belt with forced drying of manure reduced emissions from laying hens stables by some 60% (Van Horne, 1990). Forced drying of manure of slatted, littered floors, or alternatively so called trampoline floors, are expected to reduce emissions from broiler housing systems by 90 per cent. Costs consist of additional investments, recirculating air, energy use and litter use (Boonen, 1990; Brunnekreef, 1991).

The investment function for stable adaptations is the same as for low nitrogen feed. The technology, animal and country specific data are presented in Klaassen (1991a). The investment costs are annualized over the lifetime lt of the installation using the interest rate $q_l$. Fixed operating costs may consist of maintenance, insurance and administrative overhead. They are presented as a fixed percentage of the investment per animal place. Due to a lack of experience with these techniques generally no specification of the variable operating costs was possible yet for pigs and dairy cows. Therefore annual operating costs are assumed to be a fixed percentage of the investment. Variable operating costs consist only of the additional costs of natural gas.

## 2.5 Covering manure storage

Covering the storage of manure prevents 90% of the ammonia emissions (Baltussen et al. 1990b). However, since only part (some 10%) of the total ammonia released during stable and storage actually escapes from the storage the overall removal efficiency is only 10%. Costs only consist of investments (Klaassen, 1991a). The additional investments consist of the costs of the roof or the cover minus the smaller investments in the silo. The silo can be smaller since no rain enters the silo. The investments depend on the size of the silo and thus indirectly on the number of animals per stable (Klaassen, 1991a). Covering of storage is only feasible if storage facilities already exist or are expected as a result of national legislation.

## 2.6 Biofiltration of biological scrubbers

Another possibility to control the emissions from the stable is the application of various techniques that clean the stable air. These techniques can only be applied in case stables are equipped with mechanical ventilation. This is usually the case for poultry but not always for pig stables (Asman, 1990). Techniques are biofiltration, bioscrubbing and chemical scrubbers. Application of biofiltration for poultry stables may be difficult due to dust problems.

Cost estimates show wide ranges (Zeisig and Wolferstetter, 1990; Eggels and Scholtens, 1989; Demmers, 1989; Jol, 1990; van Horne, 1990; Baltussen et al., 1990b). The investment depends on the size of the installation. The technology, animal specific and country specific data are included in Klaassen (1991a). Again investment costs are annualized over the lifetime of the installation using the interest rate. Fixed operating costs are presented as a fixed percentage of the investments per animal place. No country specific prices are incorporated for labor, water and waste disposal prices due to a lack of data on the one hand and the fact that these cost items are relatively less relevant for the total annual costs.

# 3 Low ammonia application of manure

## 3.1 Introduction

To prevent the escape of ammonia during application of manure on arable land or grassland a wide variety of techniques exists (Huijsmans, 1990; Krebbers, 1990; Havinga, 1991):

- direct application (ploughing down) of manure on arable land,
- manure injection (deep) on grassland,
- sod injection (shallow) or sod manuring for manure on grassland,
- sprinkling, trenching or diluting manure on grassland.

Furthermore, the processing of manure to control manure surpluses, as a side effect, reduces ammonia emissions during application. This option, however, is less likely in countries where the manure surplus is less of a problem than in the Netherlands. In addition, the costs of manure processing are too high to justify its application for controlling ammonia emissions only.

The applicability of these techniques (apart from manure processing) depends, amongst other things, on soil type, water availability (sprinkling), and the slope of the soil. Sod manuring can be applied on soils with low carrying capacity (heavy clay soils or peat soils) where manure injection may not be feasible. Dilution of manure is partly practiced in Alpine countries and may be more appropriate for soils in steeply sloped areas.

Costs are expressed per $m^3$ manure applied since these techniques are usually carried out by contractors whose services can be rented by the individual farmer. In addition, this avoids unnecessary complications in the cost calculation routine. Costs per $m^3$ manure depend on, among other things, the technique, the volume of manure applied ($m^3$ per hectare) (Huijsmans, 1991) and the distances between land and storage (Krebbers, 1990; Havinga, 1991). The most important country-specific element is probably the mixture of techniques. Not only are there additional cost but there are also cost savings since less artificial fertilizer has to be applied. It is also possible that, because of the poor uptake of phosphate from injected manure, an additional amount of phosphate fertilizer will have to be applied at the start of the growing season.

Since we assume that these low ammonia application techniques are carried out by specialized firms there are no investments, annualized investments costs or fixed operating costs. The cost only consist of the variable costs of the mixture of techniques (ploughing down, manure injection, sod manuring, sprinkling or manure processing) minus the cost savings.

## 3.2 The algorithm

The costs of direct application or ploughing down per $m^3$ manure are:

$$c^{ma}{}_{l} = c^{fma} + c^{vma} \times Q^{mh}{}_{l} \qquad \textbf{(3.1)}$$

With:

$c^{fma}$ the fixed costs of direct application per $m^3$ manure
$c^{vma}$ the variable costs of direct application
$Q^{mh}{}_{l}$ the amount of manure applied per hectare

The cost of manure injection per $m^3$ manure are:

$$c^{mi}{}_{l} = c^{fmi} + c^{vmi} \times Q^{mh}{}_{l} \qquad (3.2)$$

$c^{fmi}$ the fixed costs of injection per $m^3$ manure
$c^{vmi}$ the variable cost of injection per $m^3$ manure
$Q^{mh}{}_{l}$ the amount of manure applied per hectare

The cost of sod manuring, or shallow injection, per $m^3$ manure are:

$$c^{ms}{}_{l} = c^{fms} + c^{vms} \times Q^{mh}{}_{l} \qquad (3.3)$$

$c^{fms}$ the fixed costs of sod manuring per $m^3$ manure
$c^{vms}$ the variable cost of sod manuring per $m^3$ manure
$Q^{mh}{}_{l}$ the amount of manure applied per hectare

The costs of sprinkling, $c^{mr}$, per $m^3$ manure consist of fixed and variable elements. The fixed costs consist of the investment in the installation. The costs per $m^3$ manure then depend on the manure production per farm, a function of the number of animals:

$$c^{mr}{}_{l} = c^{fmr} + \frac{c^{vmr}}{ss_{dl}} \tag{3.4}$$

$c^{fmr}$ the fixed costs of sprinkling per $m^3$ manure
$c^{vmr}$ the variable cost of sprinkling
$ss_{dl}$ the stable size for dairy cows

The costs of manure processing, $c^{mp}$, cannot be fully attributed to ammonia emission control since the technique is primarily directed at controlling nitrate and phosphate surpluses. Therefore only a fraction of the costs (fcn3) is attributed to ammonia:

$$c^{mp}{}_{l} = fcn3 \times c^{fmp} \tag{3.5}$$

$c^{fmp}$ the costs of processing per $m^3$ manure
fcn3 fraction of costs attributed to ammonia

In addition to the costs of low ammonia application of manure there are also costs savings due the reduction in fertilizer use. Per animal these costs savings are:

$$OM^{nf}{}_{i,k,l} = nh3a_{i,l} \times xa_{i,k} \times (1 - S^{mp}{}_{l}) \times 0.5 \times \frac{14}{17} \times c^{k}{}_{l} \times \frac{sb_i}{ar_i} \tag{3.6}$$

With:
$nh3a_{i,l}$ the emission coefficient for application
$xa_{i,k}$ the removal efficiency of application
$c^{k}{}_{l}$ the fertilizer price
$S^{mp}{}_{l}$ the share of manure processed
$sb_i$ the rate of utilization
$ar_i$ the number of animal rounds per year

The factor 14/17 is used to recalculate the emission reduction expressed in kg $NH_3$ into kg nitrogen. It is expected that the ammonia that is not emitted does not fully lead to equal savings in fertilizer. Krebbers (1990) is of the opinion that the effectiveness of the nitrogen uptake by grassland increases by a factor of two. Therefore only half of the ammonia is assumed to lead to savings in fertilizer use. For that part of the manure that is processed ($S^{mp}_l$) there are no savings in fertilizer use.

The total annual costs of the low ammonia application techniques are:

$$OM^{var}_{i,k,l} = (c^{ma} \times S^{ma}_l + c^{mi} \times S^{mi}_l + c^{ms} \times S^{ms}_l + c^{mr} \times S^{mr}_l + c^{mp} \times S^{mp}_l) \times M_{i,l} - OM^{nf}_{i,k,l} \tag{3.7}$$

In which:

$S^{ma}_l$ the share of manure directly applied
$S^{mi}_l$ the share of manure injected
$S^{ms}_l$ the share of manure sod manured
$S^{mr}_l$ the share of manure sprinkled
$S^{mp}_l$ the share of manure processed
$M_{i,l}$ the production of manure per animal

Based on the above mentioned items the unit costs for the control of $NH_3$ emissions can be calculated. Unit costs are expressed in costs per average present animal by taking into account the number of animal rounds per year $ar_i$ and the capacity utilization factor $sb_i$:

$$ca_{i,k,l} = \frac{OM^{var}_{i,k,l} \times ar_i}{sb_i} \tag{3.8}$$

The cost efficiency of the abatement option can only be evaluated if the annual costs are related to the amount of emissions reduced in order to obtain the cost per unit of $NH_3$ removed. In doing so it has to be taken into account that (combinations of) abatement options may simultaneously reduce emissions during stable and storage, application and in the meadow:

$$cn_{i,k,l} = \frac{ca_{i,k,l}}{nh3s_{i,l} \times xs_{i,k} + nh3a_{i,l} \times xa_{i,k} + nh3m_{i,l} \times xm_{i,k}} \tag{3.9}$$

In which:

| | |
|---|---|
| $nh3s_{i,l}$ | emission coefficient of stable |
| $nh3a_{i,l}$ | emission coefficient of application |
| $nh3m_{i,l}$ | emission coefficient meadow |
| $xs_{i,k}$ | efficiency of reduction stable |
| $xa_{i,k}$ | efficiency of reduction application |
| $xm_{i,k}$ | efficiency of reduction meadow |

## 3.3 The costs of low ammonia application

Direct application of manure, or ploughing down, can reduce ammonia emission by 80 to 90 per cent in comparison to superficial application. The removal efficiency of manure injection is 90 to 99 per cent. The reduction to be achieved by sod manuring, or shallow injection, varies between 75 and 99 per cent. Sprinkling, trenching or the dilution of manure has a removal efficiency of 75 to 90 per cent (Havinga, 1991; Huijsmans, 1990; Huijsmans and Bruins, 1990; Krebbers, 1989, 1990). When manure is processed the reduction would be 100 per cent.

The net costs for direct application are 0 to 7 DM/$m^3$ and for manure injection 0 to 5 DM/$m^3$. Sod manuring costs vary between 3 and 7 DM/$m^3$. Sprinkling is more expensive: costs are 6 to 18 DM/$m^3$. Manure processing costs around 25 to 35 DM/$m^3$ (Klaassen, 1991a).

The costs are, amongst other things, dependent on the amount of manure applied. Country specific elements are: the shares of the different low ammonia application techniques, the volume of manure per hectare and the fertilizer price, and the manure production per animal. As default values the share of manure ploughed down is assumed to be equal to the share of arable land, and the share of manure injected is equal to the share of grassland in each country. For the time being the default value for the shares of sod manuring and sprinkling are set zero due to a lack of data. The user of RAINS is allowed to change these values. Only in the Netherlands is manure processing assumed to take place. Since the costs of manure processing are much higher than the other techniques it is not applied for ammonia control but geared towards controlling manure (mineral) surpluses. Therefore, the fraction of the costs of manure processing attributed to ammonia control is zero in this specific example. Klaassen (1991a) supplies more details on the parameters.

# 4 Costs of combinations

The options which are available per animal category can also be applied in combination. In that case the costs are simply the sum of the costs of the separate options. More details on the options which are allowed for and the associated removal efficiencies are given in Klaassen (1991a) .

# 5 Industrial process emissions control

The total annual costs of controlling ammonia emissions from industrial processes are estimated at DM 1250 per ton $NH_3$ removed. The removal efficiency is 50% (Technica, 1984).

# 6 Some results

## 6.1 Average costs per ton ammonia abated

The average costs per ton ammonia of the different ammonia control options are presented in Table 3 and Table 4. A comparison of both Tables shows that the cost of low ammonia application are partly different in both countries. In Finland, cost savings on fertilizer use are somewhat higher since the fertilizer price and the emission coefficient for application for some animal categories (dairy cows e.g) is higher. In the Netherlands the amount of manure applied per hectare is higher which reduces the costs. Costs of stable adaptations (dairy cows and pigs), covering manure storage (dairy cows, other cattle) and biofiltration (pigs) are lower in the Netherlands mainly because the size of the stables is larger. Costs of low nitrogen feed and stable adaptations for poultry (hens and broilers) are the same in both countries. Costs of biofiltration of bio-scrubbing for poultry are somewhat smaller in Finland because the electricity price is lower.

Table 3. Average costs in Finland

| Costs per ton ammonia abated (DM/ton $NH_3$) | | | | | |
|---|---|---|---|---|---|
| Control options | Dairy cows | Control options | Pigs | Laying Hens | Broilers |
| LNF | 12777 | LNF | 20834 | 5834 | 6626 |
| SA | 29623 | SA | 24582 | 4798 | 9022 |
| CS | 77473 | BF | 36692 | 26055 | 65592 |
| LNA | 11349 | LNA | 3951 | 1218 | 153 |
| LNF+SA | 23440 | LNF+SA | 26012 | 5481 | 9340 |
| LNF+CS | 24180 | LNF+BF | 36372 | 23380 | 48412 |
| LNF+LNA | 13539 | LNF+LNA | 9030 | 2160 | 2202 |
| SA+LNA | 18659 | SA+LNA | 11899 | 2441 | 3602 |
| CS+LNA | 19128 | BF+LNA | 19162 | 11370 | 23795 |
| LNF+SA+LNA | 19763 | LNF+SA+LNA | 12327 | 3106 | 4964 |
| LNF+CS+LNA | 19140 | LNF+BF+LNA | 22271 | 11853 | 24542 |
| | Other cattle | | Indus-try | | |
| CS | 38707 | Stripping | 1250 | | |
| LNA | 13490 | | | | |
| CS+LNA | 16523 | | | | |
| | | | | | |
| LNF: low nitrogen feed. SA: Stable adaptation. CS: covering manure storage. BF: biofiltration. LNA: low ammonia application. | | | | | |

Average costs per ton $NH_3$ abated range from 153 DM/ton $NH_3$ (low ammonia application broilers) to more than 77000 DM/ton $NH_3$ (dairy cows covered storage) in Finland. In the Netherlands average costs per ton $NH_3$ vary between minus 123 DM/ton $NH_3$ (low ammonia application broilers) and 72000 DM/ton $NH_3$ (broilers biofiltration).

Table 4. Average costs in Netherlands

| Costs per ton ammonia abated (DM/ton $NH_3$) | | | | | |
|---|---|---|---|---|---|
| Control options | Dairy cows | Control options | Pigs | Laying Hens | Broilers |
| LNF | 11662 | LNF | 20671 | 5834 | 6626 |
| SA | 37111 | SA | 25199 | 5122 | 9022 |
| CS | 44290 | BF | 33472 | 28533 | 71827 |
| LNA | 3913 | LNA | 1562 | 364 | -123 |
| LNF+SA | 22523 | LNF+SA | 26311 | 5734 | 9340 |
| LNF+CS | 15307 | LNF+BF | 33491 | 25470 | 52733 |
| LNF+LNA | 7781 | LNF+LNA | 6776 | 1382 | 1965 |
| SA+LNA | 12319 | SA+LNA | 10193 | 1989 | 3433 |
| CS+LNA | 6477 | BF+LNA | 15709 | 11879 | 25872 |
| LNF+SA+LNA | 14063 | LNF+SA+LNA | 13634 | 2667 | 4799 |
| LNF+CS+LNA | 9561 | LNF+BF+LNA | 18852 | 12353 | 26555 |
| | Other cattle | | Indus-try | | |
| CS | 21517 | Stripping | 1250 | | |
| LNA | 3109 | | | | |
| CS+LNA | 4238 | | | | |
| | | | | | |
| LNF: low nitrogen feed. SA: Stable adaptation. CS: covering manure storage. BF: biofiltration. LNA: low ammonia application. | | | | | |

## 6.2 Future emissions without abatement

The development of the $NH_3$ emissions without control is shown for two example countries, Finland and the Netherlands, in Tables 5 and 6. These emissions show the potential that can be abated. Finland is selected because it is presently collecting data on the costs of controlling ammonia. The Netherlands was chosen since most of the data is based on Dutch experience. The projections are based RAINS (Klaassen, 1991b). Finnish $NH_3$ emissions will be 39 kilotons in 2000 if no control would take place. That is 30% lower than the 1980 level of 56 kilotons. This mainly results from the reduction in the livestock population. Ammonia emissions in the Netherlands would slightly decrease (10 per cent) compared to the 1980 level of 224 kilotons. The most important sources of ammonia are dairy cows, other cattle and fertilizer. In the Netherlands emissions from pigs are also relevant. Application of manure is the most important source of livestock ammonia in both countries. Other sources consist of human respiration and other, anthropogenic emissions like cats and dogs and fur animals.

Table 5. Ammonia emissions in Finland (kton $NH_3$)

| Process<br>Source | STABLE AND STORAGE OF MANURE | APPLICATION OF MANURE | PASTURE PERIOD | TOTAL |
|---|---|---|---|---|
| LIVESTOCK | | | | |
| Dairy cows | 5.0 | 4.2 | 2.9 | 12.1 |
| Other cattle | 3.4 | 2.8 | 1.9 | 8.1 |
| Pigs | 2.9 | 3.3 | 0 | 6.2 |
| Laying hens | 0.5 | 0.6 | 0 | 1.1 |
| Broilers | 0.4 | 0.7 | 0 | 1.1 |
| Sheep | 0.1 | 0.1 | 0 | 0.2 |
| Horses | 0.1 | 0.1 | 0.1 | 0.4 |
| SUBTOTAL | 12.4 | 11.8 | 4.9 | 29.1 |
| Fertilizer | | | | 2.8 |
| Industry | | | | 1.6 |
| Other sources | | | | 5.5 |
| TOTAL | | | | 39.0 |

Table 6. Ammonia emissions in the Netherlands (kton $NH_3$ in 2000)

| Process<br>Source | STABLE AND STORAGE OF MANURE | APPLICATION OF MANURE | PASTURE PERIOD | TOTAL |
|---|---|---|---|---|
| LIVESTOCK | | | | |
| Dairy cows | 14.1 | 23.1 | 19.8 | 57.0 |
| Other cattle | 9.6 | 16.4 | 7.3 | 33.3 |
| Pigs | 27.3 | 34.3 | 0.0 | 61.6 |
| Laying hens | 4.9 | 6.2 | 0.0 | 11.1 |
| Broilers | 2.7 | 4.3 | 0.0 | 7.1 |
| Sheep | 0.8 | 1.5 | 0.0 | 4.5 |
| Horses | 0.3 | 0.2 | 2.1 | 0.7 |
| SUBTOTAL | 59.8 | 86.1 | 29.4 | 175.2 |
| Fertilizer | | | | 10.2 |
| Industry | | | | 12.1 |
| Other sources | | | | 11.6 |
| TOTAL | | | | 209.1 |

## 6.3 Cost functions and cost minimization

RAINS offers the user two possibilities to reduce the emissions:

- optimization: i.e. reaching emission or deposition targets at minimal costs,
- scenario analysis: calculating the costs of a variety of combinations of control options, on any part of the emissions.

Figures 2 and 3 present the optimal, least cost combination of abatement options (the national cost functions) for Finland and the Netherlands. Both the marginal and the total costs function are shown.

The marginal costs in Finland range from 153 DM/ton $NH_3$ abated to around 138718 DM/ton $NH_3$ removed. Relatively cheap options are low ammonia-application, stripping/absorption of industrial process emissions and stable adaptations for laying hens and broilers. More expensive are options which include biofiltration for pigs or covering manure storage for cattle. With best available technologies, 17.5 kilotons of $NH_3$ could be removed in the year 2000; 22 kilotons would be left. This would imply a reduction of some 45% over the uncontrolled emissions in 2000 and a reduction of 60% over the 1980

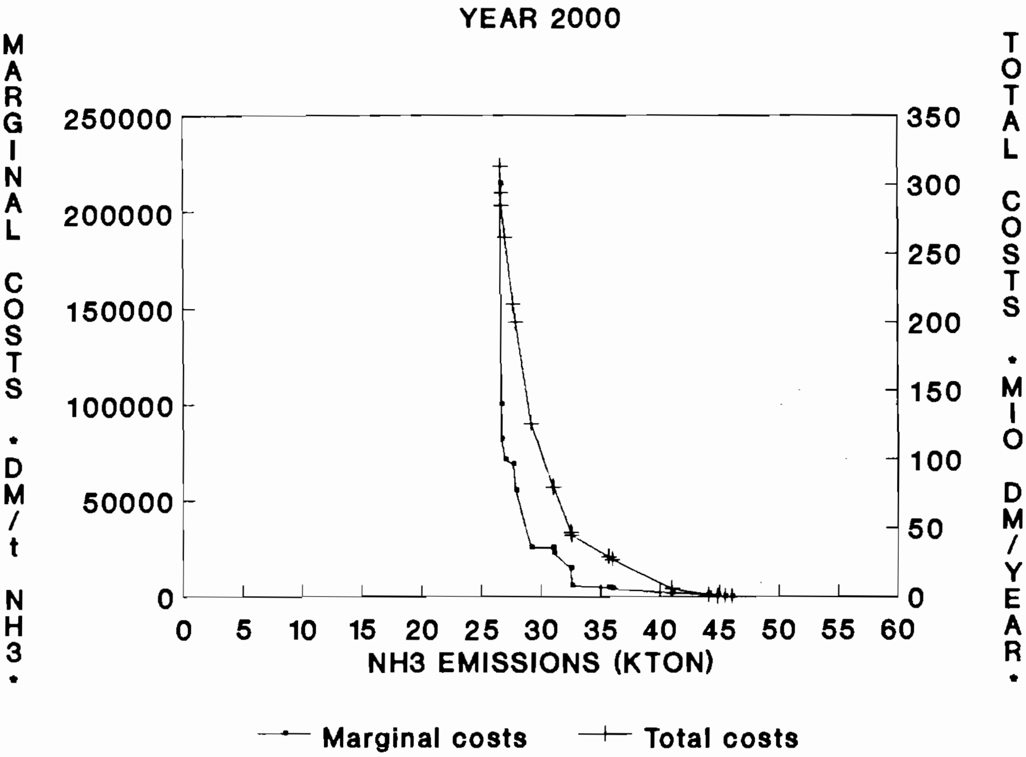


**Figure 2.** Cost function Finland

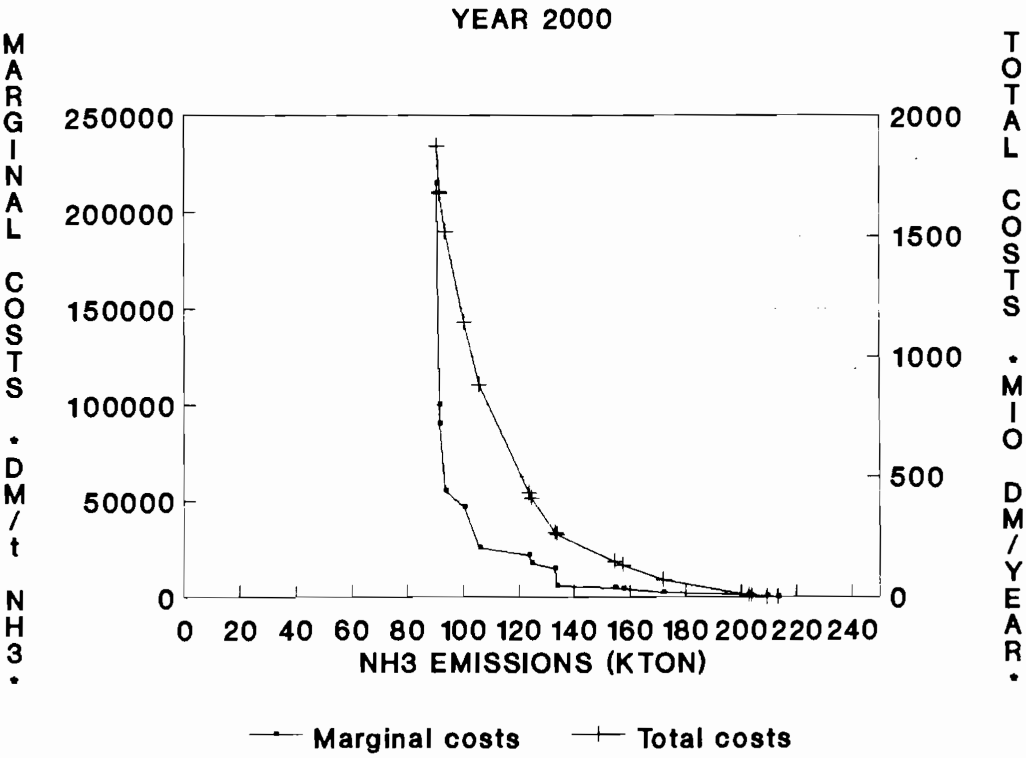


**Figure 3.** Cost function Netherlands

emissions. The associated costs would be 290 million DM. A 30 % reduction of the emissions over the 1980 level would imply an emission ceiling of 39 kilotons in 2000. This result can be achieved without control measures since unabated emissions are already expected to drop to 39 kilotons in 2000. Figure 2 shows that a reduction of the emissions from 39 to 33 kilotons is relatively cheap. After that point marginal as well as total costs will increase sharply. This being so because more expensive techniques like stable adaptations and biofiltration will have to be applied.

Marginal costs for the Netherlands range between -123 DM/ton $NH_3$ abated and 215000 DM/ton $NH_3$ removed. Relatively cheap options are low N-application, stripping/absorption of industrial process emissions and stable adaptations for laying hens. More expensive are options which include biofiltration for pigs. With best available technologies, 129 kilotons of $NH_3$ could be removed in the year 2000; 80 kilotons would be left. This would imply a reduction of 62% over the uncontrolled emissions in 2000. The reduction over the 1980 level would be 65%. This implies that the goal of the Netherlands policy, a reduction of the ammonia emissions by 70% in the year 2000 would not be feasible. However, one has to take into account that the overall application of direct ploughing down of manure and manure injection will reduce artificial fertilizer use. Consequently $NH_3$ emissions from fertilizer use will also decline. If fertilizer use would be halved, 17 kilotons of ammonia would be avoided and a reduction of 70% would be in reach. The associated annual costs would be 1774 million DM. A 30 % reduction of the emissions over the 1980 level would imply an emission ceiling of 157 kilotons in 2000. Marginal costs would be 3104 DM/ton $NH_3$ abated and the total costs would be around 73 million DM. Figure 3 shows that a reduction to 120 kilotons ammonia would be relatively cheap. After that point marginal as well as total costs tend to increase drastically since relatively expensive techniques will have to be introduced.

## 6.4 Scenario analysis and abatement potential

The second mode to operate RAINS is the scenario mode. The user can specify any combination of the options which are provided in RAINS and calculate the resulting costs and emissions. This allows the user to take into account that specific techniques can only be applied partly, or even not at all, or will be applied in specific regions only. Table 7 provides an example for the Netherlands. Table 7 shows that low ammonia application in combination with stable adaptations (SA+LNA) is applied on part of the dairy cows (800 x 1000 heads). Of the total number of cows 807000 apply low ammonia application in combination with covering of manure storage (CS+LNA). The combination which is shown would reduce ammonia emissions from dairy cows to 31 kilotons in 2000. Total ammonia emissions would fall to 184 kilotons. The associated costs would be 242 million DM/year.

Table 7. Control strategies for dairy cows (The Netherlands)

| Control options (in 1000 heads): | | | | Emission |
|---|---|---|---|---|
| No control | LNF | SA | CS | NH3 (kton) |
| 0 | 0 | 0 | 0 | 31 |
| LNA | LNF+SA | LNF+CS | LNF+LNA | Total NH3 (kton) |
| 0 | 0 | 0 | 0 | 184 |
| SA+LNA | CS+LNA | LNF+SA+LNA | LNF+CS+LNA | Annual Costs (mio DM/year) |
| 800 | 807 | 0 | 0 | 242 |

LNF: low nitrogen feed. SA: stable adaptations. CS: covered manure storage. LNA: low ammonia application of manure.

## 6.5 Discussion

A number of factors influence the results of the analysis. First of all, forecasts on livestock population and fertilizer use and the emission coefficients determine the level of unabated emissions in 2000. Forecasts on livestock population might differ as a result of changes in population growth, income per capita, export performance, agricultural policy and consumer preferences. Secondly, cost estimates of stable adaptations for pigs and dairy cows are uncertain due to the lack of practical experience. By contrast, cost estimates for low ammonia application seem more firm, although it is not quite sure to which extent techniques as manure injection and direct ploughing down can be applied in all countries in Europe. The user, however, may limit the applicability of the technique.

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# DISCUSSION AND CONCLUSIONS ON COSTS AND CONTROL OPTIONS

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## 1 Introduction

The discussion on costs of options to control ammonia emissions centered on the following items:

* Which options are available to control ammonia emissions?
* To what extent are the control techniques applicable in every country in Europe?

## 2 Control options

An overview of control options for ammonia emissions from livestock farming and industry is presented in Table 1. Overviews of control options and the associated costs are included in this volume in the contributions of Wijnands and Klaassen. Low $NH_3$ application (LNA) stands for a wide range of combinations such as: direct application of manure (or ploughing down), manure injection (deep or shallow), turf impregnation or sod manuring, irrigation or sprinkling and fertigation or dilution of manure. Compare also the contribution of Havinga in this volume. For some animals within the category other cattle several abatement techniques are also possible. Stable adaptations and low nitrogen fodder might also be possible for part of the young cattle. Bio filtration, or cleaning of stable air in general, can also be feasible for fattening calves. During the discussion two other control options were mentioned. First of all, spreading of manure could be carried out when climatological conditions (such as dark, rainy and cool weather) reduce the volatilization of ammonia. Secondly, manure might be used to produce biological gas (methane). This, however, is not regarded as a very good solution: ammonia emission during stall and storage are hardly removed and emissions during application might increase since the nitrogen content of the slurry is higher.

Table 1. Abatement options

| Livestock farming | | | | | |
|---|---|---|---|---|---|
| | Fodder | Stable and storage | | | Appli-cation |
| | LNF | SA | CS | BF | LNA |
| Dairy cows | x | x | x | | x |
| Other cattle | | | x | | x |
| Pigs | x | x | | x | x |
| Laying hens | x | x | | x | x |
| Broilers | x | x | | x | x |
| Sheep | | | | | |
| Horses | | | | | |
| Industry | | | | | |
| | Stripping/absorption | | | | |

LNF: low nitrogen fodder
SA: stable adaptation
CS: covering manure storage
BF: bio filtration
LNA: low $NH_3$ application

# 3 Potential applicability of control techniques

Low ammonia application consists of a wide variety of options (see Table 2 and the contribution of Havinga in this volume). Application of these techniques is interesting since the cost efficiency appears to be high. However, as the discussion showed, their application is not universal. In Finland for example clay soils do not allow the use of heavy machines, such as the ones used for injection of manure on grassland, because the soil might set. Direct application however is possible since the share of arable land is relatively high.

Table 2. Low (poor) $NH_3$ application

| | |
|---|---|
| 1. | Direct application (ploughing down) |
| 2. | Injection (deep or shallow) |
| 3. | Turf impregnation or sod manuring |
| 4. | Manure spreading harrows |
| 5. | Irrigation |
| 6. | Fertigation/dilution |

Irrigation might lead to run-off to the surface water. In the Netherlands twenty years of experience were needed to develop special, lighter machines which could be used for difficult soil types and circumstances. In the Federal Republic of Germany the introduction of low ammonia application equipment also depends on the presence of constructors who offer farmers the possibility to use the machines. In hilly areas there is usually much grass used for dairy cow manure spreading. Since this area is rather small, not many technical limitations are expected. In Switzerland physical limitations, such as the slope of the soil and stones, however, are relevant. In the mountains there are many animals and only a small amount of arable land. Much slurry is applied on grassland. Dilution is already applied in Switzerland and further extension is not likely. In Italy injection is difficult in heavy soils. During the rain season the soil is not passable. Slurry is usually applied on arable land before ploughing down. The slurry is applied from the road. In summary, direct application of manure on arable land seems less problematic as injection of manure on grassland. Little experience exists with injection and physical (soil type) and socio-economic circumstances (costs, experience, constructors) may limit its application.

Low nitrogen fodder consists of:

dairy cows: lower N-application of grass land in combination with changes in the fodder composition (e.g. increase in silage maize),

pigs/poultry: multi-phase feeding according to the specific needs of the animal in the growing process and addition of synthetic amino acids.

The contribution of Spiekers in this volume offers details on this option. During discussion the possibilities for dairy cows appeared to be influenced by a number of circumstances. The question arose whether it would be possible to increase silage maize or beet production. In hilly regions where grass land is dominating this might be difficult. In Switzerland there is probably only a limited possibility to reduce the N-content in grassland since the clover in the grassland has to be maintained. This influences grassland production per hectare. Moreover, N-input is already relatively low. More possibilities offers the reduction of concentrate use which is rather high in the Federal Republic of Germany, the Netherlands, Denmark and the United Kingdom. This requires analysis of silage maize and tuning with concentrate use. The N-input can probably be reduced in many countries since an amount of 300 kg total N/hectare is the maximum necessary. For pigs and poultry there are generally many possibilities to change the composition of the fodder in order to reduce the N-content.

The introduction of stable adaptations, such as flushing and manure scraping systems, seems to be in an early stage of development (see Voermans contribution to this volume). Apart from uncertainty in technical possibilities and costs, the lifetime of existing stables might determine the speed of application of this techniques. The lifetime of stables might differ per country and is, for example, believed to be more than 15 years in Germany. More relevant than the technical lifetime is the economic lifetime since it is expected that the ammonia poor stable systems will generally be introduced in new stables, although its application in existing stables is not impossible. From a technical point of view the systems seem to be fairly universally applicable, be it with country specific modifications.

Bio-filtration, or similar equipment to control the stable air, can not be applied in stables which have natural ventilation. This is generally the case for dairy cow sheds and part of the pig sties in Europe. Poultry stables generally have mechanical ventilation. Although cheap (German) systems do exist, bio-filtration is expensive and does not always function appropriately.

Covering manure storage might also reduce ammonia losses. In Italy, pig slurry has a low dry matter content and storage takes place outside the stall, without coverage. Solid manure is usually stored outside, without covered storage. In Finland peat is mixed with solid manure which reduces ammonia losses. The question remains to what extent (existing) coverage effectively reduces ammonia emissions.

**Ammonia Emissions in Europe:
Emission Factors and Abatement Costs**

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