Emissions inventory of greenhouse gases and ammonia from livestock housing and manure management

M. Samer

(*Cairo University, Faculty of Agriculture, Department of Agricultural Engineering, El-Gammaa Street, 12613 Giza, Egypt*)

Abstract: An emission inventory is a database on the amount of pollutants released into the atmosphere. The anthropogenic emissions of air pollutants and greenhouse gases are detrimental to the environment and the ecosystems. Therefore, reducing emissions is crucial. One key issue is to inventory these emissions, and consequently databases on anthropogenic emissions will be available for making decisions on implementing suitable mitigation strategies. Such investigations aim at developing national emissions inventory for domestic livestock and to identify possible abatement techniques in order to reduce these emissions. Therefore, the objectives of this study are to introduce and define the emissions inventories, review the emission inventory guides, introduce the relation between the emissions inventory and livestock buildings and manure stores and the relevant emission factors and algorithms, review the tools for processing the emissions inventories (e.g. models, software), show the evaluation and improvement methods of emissions inventories, review the emissions abatement techniques, and present examples and paradigms of available national emissions inventories for several countries.

Keywords: emissions inventory, greenhouse gases, methane, nitrous oxide, ammonia, particulate matter, livestock buildings, manure, emission factors, emissions abatement techniques

Citation: M. Samer. 2013. Emissions inventory of greenhouse gases and ammonia from livestock housing and manure management. Agric Eng Int: CIGR Journal, 15(3): 29−54.

1 Introduction

Emissions from agriculture have great environmental impact and relevant political importance, and animal husbandry is a major source of atmospheric pollutants, such as methane (CH_4), nitrous oxide (N_2O), carbon dioxide (CO_2), and ammonia (NH_3). Carbon dioxide, methane and nitrous oxide are greenhouse gases (GHG) that contribute to global warming, while ammonia is an atmospheric pollutant and responsible for eutrophication and soil acidification (Haussermann et al., 2006; Reinhardt-Hanisch, 2008; Samer et al., 2011a,b). Ammonia emissions from agricultural industries are a significant source of atmospheric reactive nitrogen, which can lead to negative environmental consequences such as ecosystem change and formation of fine particulate (Faulkner and Shaw, 2008). Ammonia may also form secondary particulate matter (PM). Particulate emissions also have an adverse impact on human health. Emissions of NH_3, Nitric oxide (NO), and non-methane volatile organic compounds (NMVOCs) arise from the excreta of agricultural livestock deposited in and around buildings and collected as liquid slurry, solid manure or litter-based farmyard manure. NMVOCs and NO are involved in the formation of ozone, which near the surface of the Earth can have an adverse effect on human health and plant growth (EMEP/EEA emission inventory guidebook, 2009; updated June 2010). Most of the N_2O emissions occurred after the deposition of manure on soil during cattle grazing, while CH_4 is mainly produced during the period when cattle manure remained in livestock buildings and in outside storage facilities (Gac et al., 2007). A detailed knowledge of the processes of NH_3 transfer from the manure and transport to the free atmosphere will contribute to development of techniques

---

Received date: 2013-02-19 **Accepted date:** 2013-04-09

Corresponding author: M. Samer, Email: samer@cu.edu.eg.

and housing designs that will contribute to the reduction of NH_3 emission to the atmosphere (Sommer et al., 2006). Livestock NH_3 emissions are higher in areas characterized by intensive livestock production with diet manipulation and land spreading offering the greatest potential for NH_3 abatement options (Carew, 2010). About 94% of global anthropogenic emissions of NH_3 to the atmosphere originate from the agricultural sector of which close to 64% is associated with livestock management (FAO, 2006). Excessive levels of NH_3 emissions contribute to eutrophication and acidification (Schuurkes and Mosello, 1988). Enteric fermentation and manure management account for 35% to 40% of the total anthropogenic CH_4 emissions and 80% of CH_4 release from agriculture (FAO, 2006). Livestock activities contribute with 65% of the global anthropogenic N_2O emissions and account for 75% to 80% of the emission from agriculture (FAO, 2006). Globally, CO_2, CH_4 and N_2O, contribute 60%, 15% and 5%, respectively, to the anthropogenic greenhouse effect (Guo and Zhou, 2007). The Intergovernmental Panel on Climate Change (IPCC, 2007) CH_4 and N_2O are greenhouse gases with global warming potentials of 23 and 296 times that of CO_2, respectively.

Zervas and Tsiplakou (2012) stated that GHG emissions are expected to cause global warming which results in extreme weather changes that could affect crop yields and productivity, food supplies and food prices. It is also expected that climate change will have an impact on animal metabolism and health, reproduction and productivity. On the other hand, the expected increased demand of animal origin products in the coming years will increase the reared animal numbers and consequently GHG emissions. They outlined the main GHGs emitted from livestock which are CO_2, CH_4 and N_2O, coming from respiration, enteric fermentation and manure management respectively, with CH_4 and N_2O having the highest global warming potential. The guidelines of the IPCC (2006) stated that CO_2 emissions from livestock are not estimated because annual net CO_2 emissions are assumed to be zero – the CO_2 photosynthesized by plants is returned to the atmosphere as respired CO_2. A portion of the C is returned as CH_4 and for this reason CH_4 requires separate consideration. Zervas and Tsiplakou (2012) stated that ruminant livestock has the highest contribution to these GHG emissions with small ruminants share being 12.25% of the total GHG emissions from livestock's enteric and manure CH_4, and manure N_2O in CO_2 equivalent, producing 9.45 kg CO_2 equivalent per kg body weight with the respective values for cattle, pigs and poultry being 5.45, 3.97 and 3.25. Since the production systems significantly affect the GHG emissions, the grazing, livestock crop complex, and intensive ones account for 30.5%, 67.29% and 5.51% for total CH_4 emission (from enteric fermentation and manure management) and 24.32%, 68.11% and 7.57% for N_2O respectively. Taking into account the positive and negative impacts of small ruminant livestock production systems on the environmental aspects in general, they recommended that a number of potentially effective measures should be taken and the appropriate mitigation technologies should be applied in order to reduce effectively and essentially the GHG emissions to the atmosphere, with no adverse effects on intensification and increased productivity of small ruminants production systems. However, small ruminants are not key categories in many countries; therefore, this would be the case of countries with a high production of small ruminants in extensive systems.

Following the adoption of the United Nations Economic Commission for Europe (UNECE) Gothenburg protocol (Protocol to the 1979 convention on long range transboundary air pollution to abate acidification, eutrophication and ground-level ozone, United Nations Economic Commissions for Europe (UNECE), Geneva.), the members struggle to achieve significant reduction in national NH_3 emissions. Similarly, the United Nations Framework Convention on Climate Change (UNFCCC) adopted the Kyoto protocol (Protocol that set binding obligations on the industrialized countries to reduce their emissions of greenhouse gases). The protocol was adopted on December 11th, 1997, in Kyoto, Japan, and entered into force on February 16th, 2005. As of September 2011, 191 states have signed and ratified the protocol), the members exert efforts to reducing the GHG emissions. In order to achieve reduction of GHG and

NH_3 emissions, it is essential to inventory these emissions.

An emission inventory is a database of the amount of pollutants released into the atmosphere. The anthropogenic emissions of air pollutants and greenhouse gases are detrimental to the environment and the ecosystems. Therefore, reducing emissions is critical. One key issue is to inventory these emissions, and as consequently emissions databases will be available for making decisions on implementing suitable mitigation strategies. The objectives of such investigations are to develop a national emissions inventory for domestic livestock and to identify possible abatement techniques to reduce these emissions. The inventory can be developed using data derived from farm surveys and national statistical records in a country and through several years, as well as Tier 1 and 2 emission factors for enteric fermentation and manure management as proposed by the Intergovernmental Panel on Climate Change. The developed inventory should be compared with other countries, in terms of pollutant emissions per unit of land area or human population. The results can be then used, with data from other agricultural sectors, to develop a national inventory for the entire agricultural sector of a country (Aljaloud et al., 2011). A product based metric (emissions/kg product) can be launched, which allows analyzing profitability, food security and environmental conflicts. When an emission inventory is developed, it can be thereafter used to develop an abatement/mitigation strategy for livestock farms as well as at national level which focuses on abating the highest emission levels that are released through determined stages of manure management (from housing to field), this to adhere to the commitments imposed by Gothenburg and Kyoto Protocols.

Reidy et al. (2008a) stated that NH_3 emissions from agriculture commonly account for > 80% of the total NH_3 emissions. Accurate agricultural NH_3 emission inventories are therefore required for reporting within the framework of the Gothenburg Protocol of the UN Convention on Long-range Transboundary Air Pollution. To allow a coordinated implementation of the Protocol, different national inventories should be comparable. A core group of emission inventory experts therefore developed a network and joint program to achieve a detailed overview of the best inventory techniques currently available and compiled and harmonized the available knowledge on emission factors (EFs) for nitrogen (N)-flow emission calculation models and initiated a new generation of emission inventories.

Existing emission inventory approaches mainly rely on expert judgment for information on farm and manure management. To detect the relatively small changes of total annual NH_3 emissions required under the Gothenburg Protocol, expert judgment is considered insufficient. Therefore, a new NH_3 emission inventory based on a detailed representative stratified survey on farm and manure management conducted on 1950 farms was presented by Reidy et al. (2008b). Emissions estimated within a National Emission Inventory should be compared to emissions estimated using Guidebook emission factors and National statistics for a specific year or period. The objective is to explore the quality of both methods and to find major differences and similarities (Dröge et al., 2010).

Chadwick et al. (2011) stated that slurry and farmyard manure are an inevitable consequence of livestock products generated from housed animals. These manures are recycled back to land for plants to use the nutrients they contain. However, since they contain inorganic N, microbial sources of C and water, they provide the essential substrates required for the microbial production of N_2O and CH_4. These greenhouse gases can be produced and emitted at each stage of the “manure management continuum”, being the livestock building, manure stores, manure treatment and manure spreading to land. The contribution that manure management makes to total national agricultural emissions of N_2O and CH_4 varies, but it can exceed 50% in countries reporting to the UNFCCC in 2009 (Chadwick et al., 2011). On farm management decisions interact with environmental controls such as temperature and water availability of key microbial processes (i.e., nitrification, denitrification, methanogenesis, CH_4 oxidation), affecting the magnitude of emissions from each stage of the manure management continuum (Chadwick et al., 2011). Reidy et al. (2009) mentioned that each country uses its own excretion model and defines cattle groups in different ways and with different degrees of detail (in breed, feeding regimes). They added that there were fewer published results of NH_3 emissions from solid manure than for slurry and the introduction of litter leads to more complex interactions (e.g. immobilization and mineralization) and greater, and highly variable, emissions of other N gases. In addition, the state of development of the models with respect to immobilization and denitrification is quite different. They concluded that it is important to get a better overview of the existing knowledge, an identification of research gaps and a thorough re-editing of some of the models (especially DYNAMO and MAM) for processes other than NH_3 emission.

Comparing complete inventories to independent efforts in assessing emissions, e.g. atmospheric measurements combined with source apportionment, allows better understanding and quantifying the reliability of inventory data (Winiwarter et al., 2009). A number of emission factors (EFs) can be proposed for developing ammonia emissions inventories (Faulkner and Shaw, 2008). Besides the production facilities, the EFs for animal feeding operations should also be considered (Faulkner and Shaw, 2008). Seasonal distribution of NH_3 emissions should be considered when developing emissions inventory. Significant uncertainty exists in the seasonal distribution of NH_3 emissions since the predominant sources are animal husbandry and fertilizer application. Previous studies that estimated bottom–up and top–down NH_3 emissions have provided the most comprehensive information available about the seasonality of NH_3 emissions. Inverse modeling (top-down) results suggest that the prior NH_3 emission estimates should be increased in summer and decreased in winter, while results for spring and fall are questionable due to precipitation prediction biases. A final conclusion from this study is that total NH_x (NH_3 and aerosol NH4+) air concentration data are essential to quantitative top–down analyses of NH_3 emissions that can extend beyond what is possible using precipitation chemistry data (Gilliland et al., 2006). Fugitive emissions may affect the accuracy of emissions inventory, where fugitive emissions are emissions caused by wind shear, material transfer processes or other mechanical forces from non-point sources, e.g. natural particles like sea salt and windblown dust (Winiwarter et al., 2009).

Merino et al. (2011) stated that quantification of greenhouse gas emissions is reported annually in the National Emission Inventory. They added that a mathematical approach should be used to assess uncertainties of estimated emission factors (EF) in order to update the annual CH_4 and N_2O emissions. Additionally, they mentioned that emissions from manure management have the largest uncertainty due to the high natural variability of manure. The more accurate animal data are available, the lower uncertainty is expected. This is the case in the intensive production systems. Webb et al. (2005) found that among the larger sources of NH_3 emissions, those from buildings housing cattle and pigs on straw were the most uncertain. Sommer et al. (2006) stated that livestock excreta and manure stored in housing, in manure stores, in beef feedlots, or cattle hardstandings are the most important sources of NH_3 in the atmosphere. Inventories have shown that animal housing, stored animal manure, and exercise areas account for about 69–80% of the total emission of NH_3 in Europe (ECETOC, 1994; Hutchings et al., 2001). Therefore, the objectives of this study are to introduce and define the emissions inventories, review the emission inventory guides, introduce the relation between the emissions inventory and livestock buildings and manure stores and the relevant emission factors and algorithms, review the tools for processing the emissions inventories (e.g. models, software), show the evaluation and improvement methods of emissions inventories, review the emissions abatement techniques, and present examples and paradigms of available national emissions inventories for several countries.

2 Definition of emissions inventory

An emission inventory is a database of the amount of pollutants released into the atmosphere. An emission inventory usually includes the total emissions for one or more specific air pollutants, originating from all source categories (agriculture, energy, industry, waste, and some other sources) in a certain geographic area and within a specified time period, usually a specific year. Generally, an emission inventory provides the following information: (1) the types of activities that cause emissions, (2) the chemical or physical identity of the pollutants included, (3) the geographical area covered, (4) the time span over which emissions are estimated, (5) the methodology that was implemented to inventory the emission(s). The emission inventories are developed for scientific applications as well as for policy processes and decision making.

3 Emission inventory guides

There are two guides: the air pollutant emission inventory guidebook of the European Environment Agency (EEA) and the Cooperative programme for monitoring and evaluation of the long-range transmission of air pollutants in Europe (EMEP), and the guidelines of the Intergovernmental Panel on Climate Change (IPCC). Dröge et al. (2010) stated that knowledge of emissions and releases of pollutants to the environment is crucial for every environmental pollution problem. Information on emissions is therefore an absolute requirement in understanding environmental problems and monitoring progress in solving these problems. Dammgen and Webb (2006) mentioned that the reduction of emissions of air pollutants is subject of international conventions, which include reporting of emissions in accordance with guidelines or guidebooks provided. Within the Convention on the Long-range Transboundary Air Pollution (LRTAP), the Atmospheric Emission Inventory Guidebook describes the methodology. With respect to emissions from agricultural sources, in particular from animal husbandry, this guidebook has undergone major modifications: the calculation procedure making use of partial emission factors for the various sources of emissions (animal house, storage, manure application, etc.) were replaced by a mass flow concept for both nitrogen and carbon species. The necessity to update both the Guidebook and the IPCC Guidelines as complementary tools to describe agricultural emissions and mass flows is emphasized.

3.1 EMEP/EEA emission inventory guide

Dröge et al. (2010) mentioned that in the field of long-range transboundary air pollution, countries in both Europe and North-America have agreed to decrease the impacts of air pollution below a level that is damaging human health and ecosystems by ratifying the UNECE Convention on Long-Range Transboundary Air Pollution (LRTAP). The aim of the Convention is “that Parties shall endeavor to limit and, as far as possible, gradually reduce and prevent air pollution including long-range transboundary air pollution. Parties develop policies and strategies to combat the discharge of air pollutants through exchanges of information, consultation, research and monitoring”. The LRTAP Convention and its protocols include important pollutants like NO_x, SO_x, NH_3, NMVOC, CO, particulate matter, heavy metals and persistent organic pollutants. Parties to the LRTAP convention have agreed to annually report atmospheric emissions and are required to set up an emission inventory. As a minimum, parties shall use the latest version of the EMEP/EEA Air Pollutant Inventory Guidebook, but most countries have set up their own inventory, which uses country specific information to supplement the information from the Guidebook.

The EMEP/EEA air pollutant emission inventory guidebook (formerly called the EMEP CORINAIR emission inventory guidebook) provides guidance on estimating emissions from both anthropogenic and natural emission sources. It is designed to facilitate reporting of emission inventories by countries to the UNECE Convention on Long-range Transboundary Air Pollution and the EU National Emission Ceilings Directive. The EEA publishes the Guidebook, with the UNECE’s Task Force on Emission Inventories and Projections (EMEP/EEA, 2009).

3.2 IPCC guidelines for national inventories

The 2006 IPCC Guidelines for National Greenhouse Gas Inventories (2006 IPCC Guidelines) provide methodologies for estimating national inventories of anthropogenic emissions by sources and removals by sinks of greenhouse gases. The 2006 IPCC Guidelines were prepared in response to an invitation by the Parties to the UNFCCC. They may assist Parties in fulfilling their commitments under the UNFCCC on reporting on inventories of anthropogenic emissions by sources and removals by sinks of greenhouse gases not controlled by the Montreal Protocol, as agreed by the Parties. The 2006 IPCC Guidelines are in five volumes. Volume 1 describes the basic steps in inventory development and offers the general guidance in greenhouse gas emissions and removals estimates based on the authors' understanding of accumulated experiences of countries over the period since the late 1980s, when national greenhouse gas inventories started to appear in significant numbers. Volumes 2 to 5 offer the guidance for estimates in different sectors of economy, which are: Energy, Industrial Processes and Product Use, Agriculture, Forestry and Other Land Use, and Waste. Volume 4 (Chapter 10) provides guidance on methods to estimate emissions of methane from Enteric Fermentation in livestock, and methane and nitrous oxide emissions from Manure Management (IPCC, 2006).

4 Livestock buildings and emissions inventory

Sommer et al. (2006) stated that process-based models can be an accurate and cost-effective means of estimating emissions from a discrete source. This is especially relevant for predicting or monitoring the impact of emissions from buildings, outdoor yards, and manure stores of a large livestock production unit on adjacent sensitive area(s). In mechanically ventilated buildings ventilation rate often determines NH_3 emissions. While data on ventilation rate may be available for models of emission from individual buildings or farms, such data will not be available for national-scale models. Surrogates for ventilation rates may be available based on ambient temperature and wind speed and ambient data may also be used to calculate conditions within naturally ventilated buildings. However, to be accurate such meta-models would require detailed information of the number, age, and weight of animals within buildings and again, this may be available to use for individual buildings or farms but will not be available for national-scale models except via census data of total numbers of livestock, buildings, and averages/ distributions of animals within those buildings. Such information is also known as activity data, which, in the context of calculating NH_3 emissions, may be defined as data quantifying agricultural practices that have an influence on NH_3 emissions, for example, housing systems. They concluded that the limiting factor in our ability to model emissions from buildings housing livestock is knowledge of what is in those buildings and how they are managed. They added that the following information is needed to make accurate estimates of national NH_3 emissions from buildings housing livestock, hardstandings, and manure stores: (1) animal numbers, (2) the housing period for all types of cattle and for sheep, (3) the amount of time cattle spend on hardstandings and the proportion of cattle that use them, (4) the proportions of cattle, all classes, housed on slurry- or straw-based systems, (5) The proportions of cattle and pig slurry stored in aboveground tanks, lagoons, and weeping walls, and (6) the adoption of covers for slurry stores.

4.1 Emission factors for livestock barns and manure storages

Animal manure from housing is a mixture of feces and urine, bedding material (straw, wood shavings, sawdust, sphagnum, etc.), spilt feed and drinking water, and water used for washing floors. Housing systems are often adapted to the category of housed animal such as calves, dairy cows, sows, fatteners etc., where most cattle buildings are naturally ventilated (Sommer et al., 2006). The quantification of gaseous emissions from naturally ventilated buildings is sophisticated (Samer et al., 2011a,b,c,d).

Faulkner and Shaw (2008) stated that ammonia emissions from agricultural industries are a significant source of atmospheric reactive nitrogen, which can lead to negative environmental consequences such as ecosystem change and formation of fine particulate. They presented recently developed ammonia emission factors (EFs) from literature for animal feeding operations, including production facilities for beef and dairy cattle, swine, and poultry. Several studies have investigated the emission factors from livestock buildings. Samer et al. (2011a) listed that the emission factors for dairy cows (lactating Holstein Friesian) –through summer- were 63.5, 270, 23842, and 15.5 kg $year^{-1}$ LU^{-1} (LU: livestock unit and is equivalent to 500 kg animal live mass) for NH_3, CH_4, CO_2, and N_2O respectively.

On the other hand, the emission factors –through winter- were 32, 169, 20431, and 17.3 kg $year^{-1}$ LU^{-1} for NH_3, CH_4, CO_2, and N_2O respectively. However, the aforementioned values differ discernibly from study to another depending on the number of experiments, time length, climatic conditions…etc (Samer et al., 2011a,b,c,d; Samer et al., 2012b). Winiwarter et al. (2009) mentioned that due to particulate matter (PM) specific circumstances such as the large number of sources, very different release pathways and differences of the individual particles in terms of chemical composition or size, it is very difficult to appropriately handle measurement conditions to arrive at adequate emission factors, especially when emission points cannot be defined clearly.

Gupta et al. (2007) reported seasonal variation in CH_4 and N_2O emission rate from solid storage of bovine manure in Delhi as well as emission factors and emission inventory from manure management systems in India. Emission flux observed in the year 2002-2003 was 4.29 ± 1, 4.84 ± 2.44 and 12.92 ± 4.25 mg CH_4 kg^{-1} dung day^{-1}, as well as 31.29 ± 4.93, 72.11 ± 16.22 and 6.39 ± 1.76 mg N_2O kg^{-1} dung day^{-1} in winter, summer and rainy seasons, respectively. CH_4 emission factors varied from 0.8 to 3.3 kg hd^{-1} $year^{-1}$ for bovines and were lower than IPCC-1996 default values. N_2O emission factors varied from 3 to 11.7 mg hd^{-1} $year^{-1}$ from solid storage of manure. Inventory estimates were found to about 698 ± 27 Gg CH_4 from all manure management systems and 2.3 ± 0.46 tons of N_2O from solid storage of manure for the year 2000.

O'Shaughnessy and Altmaier (2011) stated that the swine concentrated animal feeding operations (CAFOs) emit hydrogen sulfide (H_2S) from both housing structures and waste lagoons. An average emission flux rate of 0.57 mg m^{-2} s^{-1} was determined for swine CAFO lagoons. Using the average total animal weight (kg) of each CAFO, an average emission factor of 6.06×10^{-7} μg yr^{-1} m^{-2} kg^{-1} was calculated. From studies that measured either building or lagoon emission flux rates, building fluxes, on a floor area basis, were considered equal to lagoon flux rates. Costa and Guarino (2009) investigated yearly emission factor of dust and greenhouse gases through continuous measurements in swine husbandry. They found that the mean yearly emission factor of PM_{10} was measured in 2 g d^{-1} LU^{-1} for the weaning room, 0.09 g d^{-1} LU^{-1} for the farrowing room, 2.59 g d^{-1} LU^{-1} for the fattening room and 1.23 g d d^{-1} LU^{-1} for the gestation room. The highest PM_{10} concentration and emission per LU was recorded in the fattening compartment while the lowest value was recorded in the farrowing room. The yearly emission factor for CO_2 was measured in 5,997 g d^{-1} LU^{-1} for the weaning room, 1,278 g d^{-1} LU^{-1} for the farrowing room, 13,636 g d^{-1} LU^{-1} for the fattening room and 8,851 g d^{-1} LU^{-1} for the gestation room. The yearly emission factor for CH_4 was measured in 24.57 g d^{-1} LU^{-1} for the weaning room, 4.68 g d^{-1} LU^{-1} for the farrowing room, 189.82 g d^{-1} LU^{-1} for the fattening room and 132.12 g d^{-1} LU^{-1} for the gestation room. The yearly emission factor for N_2O was measured in 3.62 g d^{-1} LU^{-1} for the weaning room, 0.66 g d^{-1} LU^{-1} for the farrowing room, 3.26 g d^{-1} LU^{-1} for the fattening room and 2.72 d^{-1} LU^{-1} for the gestation room. It can be inferred that the emission factors of the different gases differs depending not only the barn being investigated but also on specific parts/compartments inside the barn. This should be considered when inventorying the emissions.

4.2 Algorithms for determining emissions from buildings and manure stores

Sommer et al. (2004) developed algorithms for calculating methane and nitrous oxide emissions from manure management. They found that biogenic emissions of CH_4 and N_2O from animal manure are stimulated by the degradation of volatile solids (VS) which serve as energy source and a sink for atmospheric oxygen. They presented algorithms which link carbon and nitrogen turnover in a dynamic prediction of CH_4 and N_2O emissions during handling and use of liquid manure (slurry). A sub-model for CH_4 emissions during storage relates CH_4 emissions to VS, temperature and storage time, and estimates the reduction in VS. A second sub-model estimates N_2O emissions from field-applied slurry as a function of VS, slurry N and soil water potential, but emissions are estimated using default emission factors. Anaerobic digestion of slurry and organic waste produces CH_4 at the expense of VS. Accordingly, the model predicted a 90% reduction of

CH_4 emissions from outside stores with digested slurry, and a >50% reduction of N_2O emissions after spring application of digested as opposed to untreated slurry. Their study indicates that simple algorithms to account for ambient climatic conditions may significantly improve the prediction of CH_4 and N_2O emissions from animal manure. Sommer et al. (2006) developed algorithms for determining ammonia emission from buildings housing cattle and pigs and from manure stores.

Keener and Zhao (2008) presented a modified N-balance method, which, although it does not distinguish for losses of N as N_2, NH_3 or NO_x's, it does determine accurately the upper limit on NH_3-N emissions for both forced and naturally ventilated livestock facilities. The method is based on nitrogen and ash contents and ratio of the inputs (feed, bedding, water, and animals) and outputs (products and waste) of the animal-production systems and does not require measuring total masses of manure from the system. Generalized equations for all classes of livestock operations were presented. This N-balance method is simple, low cost and accurate in predicting the upper limits on NH_3 emissions in the air from livestock facilities.

4.3 Emissions inventory based on farm survey

Reidy et al. (2008b) stated that existing emission inventory approaches mainly rely on expert judgment for information on farm and manure management. To detect the relatively small changes of total annual ammonia (NH_3) emissions required under the Gothenburg Protocol, expert judgment is considered insufficient. They presented, therefore, NH_3 emission inventory based on a detailed representative stratified survey on farm and manure management conducted on 1950 farms. The survey data was used to calculate NH_3 emissions with the emission model DYNAMO for each farm participating in the survey. This allowed the effects of the variability of farm and manure management parameters among farms on the NH_3 emissions to be fully taken into account. Weighted emission factors per animal for 24 livestock categories and 36 farm classes were used to upscale to the national inventory. The stratified sampling and the individual farm calculations allow the comparison of emissions from specific regions and altitudes and the study of the variability among farms. This emission inventory approach permits a more detailed analysis of the regional distribution of NH_3 emissions as well as a more robust and standardized monitoring of the future development of emissions.

National inventories of N emissions to the atmosphere from cattle depend on reliable information about husbandry practices, with appropriate spatial and temporal resolution. Sheppard and Bittman (2011) quantified the prevalence of pasture and forage management practices that impact N intake and NH_3 emissions, with implications for N_2O emissions using a survey of 1400 beef cattle farmers approximately. The survey data were coupled to a mass balance model to inventory NH_3 emissions by accounting for total ammoniacal (i.e., ammonia and ammonium) N (TAN) from excretion through to land spreading. As inputs, the model requires excretion fractions, census of agriculture animal populations and detailed farm management practices from the survey. They stated that the N intake by grazing cattle is especially difficult to quantify. Early season grazing, when forages have elevated crude protein (>300 g CP/kg dry matter), indicated potential for elevated emissions in May and June. Such exceptional crude protein concentrations during spring grazing may be important for N emissions, especially NH_3. They added that beef cattle were reported by producers to spend 7%–30% of their time congregated, such as around water sources, feed or shade trees. This level of biosphere activity was attributed in the model to ～20% increases in NH_3 emissions relative to open pasture, and similar or higher effects would be expected for N_2O emissions.

4.4 Whole farm systems models

Crosson et al. (2011) stated that in order to comply with the United Nations Framework Convention on Climate Change (UNFCCC) greenhouse gas (GHG) emissions reporting requirements, the Intergovernmental Panel on Climate Change (IPCC, 2006) developed guidelines for calculating national GHG inventories in a consistent and standard framework. Although appropriate for national level accounting purposes, IPCC methodologies lack the farm level resolution and holistic approach required for whole farm systems analysis. Thus, whole farm systems modeling is widely used for farm level analysis. Whole farm systems models are an appropriate tool to develop and measure GHG mitigation strategies for livestock farms. Therefore, developing a methodology for estimating emission at farm level reaching the national level would be an asset. Variation in farm system parameters, and the inherent uncertainties associated with emission factors, can have substantial implications for reported agricultural emissions and thus, uncertainty or sensitivity analysis in any modeling approach is needed. Although there is considerable variation among studies in relation to quality of farm data, boundaries assumed, emission factors applied and co-product allocation approach, we suggest that whole farm systems models are an appropriate tool to develop and measure GHG mitigation strategies for livestock farms.

5 Tools for processing emission inventories

5.1 Mass-Flow models

The mass-flow approach to estimating NH_3 emissions has evolved from initially estimating only NH_3 emissions (Cowell and ApSimon, 1998), to including processes such as immobilization, mineralization, nitrification and denitrification in order to properly quantify the TAN flow (Webb and Misselbrook, 2004) and to improve the accuracy of the NH_3 emission calculations. Webb and Misselbrook (2004) described a mass-flow approach to estimating NH_3 emissions from livestock production at national scale. NH_3 is emitted from a pool of ammoniacal-N (TAN) in livestock excreta. This pool is not added to during manure management, but is depleted by losses as gaseous emissions and leachate and by immobilization in litter. At each stage of manure management, a proportion of TAN will be lost, mainly as NH_3, and the rest passed on to the next stage. This approach enables rapid and easy estimation of the consequences of abatement at one stage of manure management (upstream) on NH_3 losses at later stages of manure management (downstream). Such a model facilitates scenario analysis of abatement options and cost-curve production. Model output is most sensitive to variation in estimates of the length of the housing period for cattle. Thus, the collation of accurate data on factors such as the length of the housing period and other ‘activity’ data, are as important in compiling accurate inventories of national emissions as improving the accuracy of emission factors. Priorities for research should be to accurately quantify the relationship between NH_3 emissions from livestock buildings and the proportion of the day those buildings are occupied, and to characterize and quantify the transformations of N that take place during storage of litter-based manures.

Gac et al. (2007) developed a methodology based on (1) national data concerning livestock and rearing practices and (2) a mass-flow approach which was developed to quantify NH_3, CH_4 and N_2O emissions resulting from manure management. They performed a literature review to determine emission factors for each animal type and each management stage. Afterwards, they have developed a Microsoft Access® database containing these emission factors, the census data and manure compositions, allowing the calculation of gaseous emissions by the mass-flow approach. From this database, a national gas emissions inventory resulting from manure management was drawn up.

Reidy et al. (2009) and Reidy et al. (2008a) compared six N-flow models used to calculate national NH_3 emissions from litter-based systems and slurry-based systems, respectively, in different European countries using standard data sets. It was found that output of the models tested proved to be much more variable for solid manure than for slurry. The variability of emissions found in practice is likely to be much greater for straw-based systems than for slurry systems. They added that the differences in estimates of NH_3 emissions decreased as estimates of immobilization and other N losses increased. Since immobilization and denitrification depend also on the C:N ratio in manure, there would be advantages to include C flows in mass-flow models. This would also provide an integrated model for the estimation of emissions of methane, non-methane VOCs and carbon dioxide. Estimation of these would also enable an estimate of mass loss, calculation of the N and TAN concentrations in litter-based manures and further validation of model outputs.

Wang et al. (2011) developed a detailed inventory framework to estimate N_2O and CH_4 emissions from UK agriculture using the IPCC approach. The inventory framework model was illustrated by combining relevant emission factors with agricultural census data for England, Wales, Scotland and Northern Ireland for the year 2000 to derive country-specific emission estimates which were summed to derive the UK total. The framework enables simple assessment to be made of the impact on national emissions of using different emission factors (EFs) (e.g. site- or local-specific compared with IPCC default factors). The framework was used to calculate the average annual emissions of nitrous oxide N_2O and CH_4 for specific livestock and crops, and amounts lost through volatilization, leaching and runoff for each country in the UK. The framework provides a simple, realistic and transparent approach to estimating national emissions and can easily be updated.

Bannink et al. (2011) stated that the protocol for the National Inventory of agricultural greenhouse gas emissions in The Netherlands includes a dynamic and mechanistic model of animal digestion and fermentation as an Intergovernmental Panel on Climate Change (IPCC) Tier 3 approach to estimate enteric CH_4 emission by dairy cows. The model predicts CH_4 emission by considering various dietary characteristics, including the types of carbohydrate, protein, fat, intrinsic degradation characteristics of feeds, as well as ruminal fractional passage rates, fluid volume and acidity, instead of assuming a fixed CH_4 energy conversion factor in the Tier 2 approach.

Pouliot et al. (2012) highlighted the similarities and differences in how emission inventories and datasets were developed and processed across North America and Europe for the Air Quality Model Evaluation International Initiative (AQMEII) project and then characterized the emissions for the two domains. They focused specifically on the creation of "model-ready" gridded emission datasets for 2006 across the two continental study domains. They discussed the practice of creating and processing the two inventories with a focus on emission factors, spatial allocation, temporal variability, speciation of PM and VOCs, and the mechanics of distributing the data and supporting emission algorithms to the modeling community. The spatial and temporal distribution on common scales was compared for the pollutants of primary concern: NO_x, VOCs, SO_2, $PM_{2.5}$, CO, and NH_3. Because of differences of population distribution, emissions across North America tend to be more heterogeneous in spatial coverage than in Europe. The temporal patterns in the estimated emissions are largely the result of assumptions used to characterize human activity, with the exception of "natural" emissions, which are modulated by meteorological variability, and emissions from large electric generating units in the U.S., which have the benefit of continuous emission monitors that provide hourly resolved profiles. Emission estimates in both study domains are challenged by several important but poorly characterized emission source sectors, notably road dust, agricultural operations, biomass burning, and road transport. One important outcome of this comparison of 2006 emissions between Europe and North America is the greater understanding provided into how the emission estimates developed for the AQMEII project impact regional air quality model performance. On the other hand, the Emission Database for Global Atmospheric Research (EDGAR) provides global past and present-day anthropogenic emissions of air pollutants and greenhouse gases by country and on a $0.1^o \times 0.1^o$ grid. The current development of EDGAR is a joint project of the European Commission JRC Joint Research Centre and the Netherlands Environmental Assessment Agency.

Velthof et al. (2012) stated that NH_3 emissions from agriculture are quantified using a nitrogen (N) flow approach, in which the NH_3 emission is calculated from the N flows and NH_3 emission factors. Because of the direct dependency between NH_3 volatilization and Total Ammoniacal N (TAN; ammonium-N + N compounds readily broken down to ammonium) an approach based on TAN is preferred to calculate NH_3 emission instead of an approach based on total N. They developed a TAN-based NH_3-inventory model, called NEMA

(National Emission Model for Ammonia). The total N excretion and the fraction of TAN in the excreted N are calculated from the feed composition and N digestibility of the components. TAN-based emission factors were derived or updated for housing systems, manure storage outside housing, manure application techniques, N fertilizer types, and grazing.

Hellsten et al. (2008) provided an updated spatial NH_3 emission inventory for the UK for the year 2000, based on an improved modeling approach and the use of updated input datasets of a model called AENEID1 (Atmospheric Emissions for National Environmental Impacts Determination) which its methodology constituted the basis for the updated model AENEID2 that distributes NH_3 emissions from a range of agricultural activities, such as grazing and housing of livestock, storage and spreading of manures, and fertilizer application, at a 1-km grid resolution over the most suitable landcover types.

5.2 Software systems for emission inventories

Samaali et al. (2007) stated that emission databases need to be processed for several purposes, e.g., policy and scientific, and so require flexible and fast handling. As a result, the creation and use of convenient interactive tools allowing easy and fast processing are of high importance. They presented software devoted to emission inventories processing and applied to the actualization of the ESCOMPTE inventory. The ESCOMPTE inventory is a yearly high-resolution (1 km^2) emission inventory created for use in chemical transport models to study ozone pollution. This tool includes several functions allowing the update of emission inventories at different temporal and spatial resolutions based on the reference ESCOMPTE inventory. The actualized yearly and hourly inventories are found consistent with the hypothesis involved in emission inventory creation. Due to its flexible architecture, several other applications related to air quality analysis can be performed such as the creation of emission scenarios allowing the evaluation of abatement strategies and regulation effects on anthropogenic emissions. Also, the architecture of the software allows an easy application to other emission inventories provided that they have similar or adaptable structure compared to the ESCOMPTE one.

Winiwarter and Schimak (2005) stated that atmospheric emission inventories are important tools for environmental decision making. The need to include transparency and reproducibility in emission calculation also fostered the development of environmental software systems for emission inventories. In general such software systems are organized as databases which contain as much emission related information as is available. For their own concept they introduced a layer of emission information as interim results, which were stored as specific tables. While this structure is needed for allowing computations at an acceptable response time, it is also used to differentiate detail levels between a user's sphere and an expert's sphere. In the user's sphere, it is expected that any user adapts the evaluation algorithms without the need to know about limitations in emission calculations. Still also the expert's sphere is openly accessible, allowing for full transparency of the calculation procedures. Considering uncertainty of input data and the impossibility to perform full validation, much of the information contained in an inventory will not contribute to improve the results in terms of emissions. Nevertheless, this information is still valued as a contribution to an expert system, which may become crucial at a different application of an inventory. Emission inventory systems generally contribute to comparability between data from diverse inputs, especially as they are able to support error-checking procedures, an important task due to the large number of data providers.

6 Evaluation and Improvement of Emissions Inventories

The 2006 IPCC Guidelines for National Greenhouse Gas Inventories (2006 IPCC Guidelines) provide General Guidance and Reporting (in Volume 1) which includes: Approaches to Data Collection, Uncertainties, Methodological Choice and Identification of Key Categories, Time Series Consistency, Quality Assurance/ Quality Control (QA/QC) and Verification, Precursors and Indirect Emissions, and Reporting Guidance and

Tables. The contents of this volume provide general guides to develop high quality emissions inventories (IPCC, 2006). Similarly, The EMEP/EEA air pollutant emission inventory guidebook provides guides on: Key category analysis and methodological choice, Data collection, Time series consistency, Uncertainties, Inventory management, Improvement and quality assurance and control QA/QC (EMEP/EEA, 2009). On the other hand, the Executive Body for the Convention on Long-Range Transboundary Air Pollution (2007a), reported methods and procedures for the technical review of air pollutant emission inventories.

In the early phase of the evolvement of the emissions inventories, Rypdal and Flugsrud (2001) developed a sensitivity analysis as a tool for systematic reductions in GHG inventory uncertainties. They elucidated that a greenhouse gas emission inventory consists of a large number of input parameters, many of which have high uncertainties. The Kyoto protocol requires accurate emission data. It is, consequently, important to reduce the inventory uncertainty by improving the input parameters and methodologies in a cost-efficient manner. They defined a key parameter as one that has significant effect on the total emissions or trend and their uncertainty. Key parameters may be ranked according to their contribution to total emissions and trend uncertainty. Quantitative uncertainty estimates of emissions are not available in many countries. In order to evaluate key parameters in such inventories, they proposed a simplified approach based on thresholds. The simplified threshold approach gives insight into the inventory and identifies the key parameters. More sophisticated methods of sensitivity analysis assessments are, however, useful in order to seek specific improvements. The key parameters for determination of total emissions are the large and uncertain sources. Smaller emission sources may be key parameters for the trend determination if their source level is strongly increasing or decreasing.

Battye et al. (2003) mentioned that in terms of the potential change in mass of NH_3 emissions over regional domains, the uncertainty for soils and vegetation, and livestock waste represents a large amount, where these uncertainties stem from variations in measured emission factors. All of the sources of NH_3 emissions are subject to variability and uncertainty. The soils and vegetation category represents one of the most important sources of uncertainty, whereas Warneck (1988) stated that plants will either absorb or give off ammonia depending on the concentration of an ammonium ion in the plant and the concentration of ammonia gas in the surrounding air. Accordingly, Battye et al. (2003) stated that the emissions estimates used in atmospheric models for vegetation should probably be linked to predicted concentrations of free ammonia, which would vary through the year and across the modeling domain. They added that the lack of monitoring data on ammonia gas is another important data gap. Without data on ammonia gas, there is little that can be done to evaluate the validity of the ammonia inventory. This lack of data also complicates any evaluations of modeled equilibria among NH_3, NH_4^+, NO_3^- and SO_4^{2-}. They added that, in terms of the mass of emissions, uncertainties for the livestock waste category are very large. In addition, livestock emissions are highest in rural areas and in the summer. Uncertainties in NH_3 emissions would have the strongest impact in winter, when the formation of NH_4NO_3 particulate is favored by colder temperatures, and in urban areas, where NH_3 may be the limiting component in the formation of particulate NH_4NO_3. Although additional measurements would be helpful, a better characterization of the timing of livestock waste and fertilizer application would also reduce the uncertainty of seasonal emissions estimates.

Ramirez et al. (2008) stated that the main contributors to overall uncertainty of inventories are found to be related to N_2O emissions from agricultural soils, the N_2O implied emission factors of Nitric Acid Production, CH_4 from managed solid waste disposal on land, and the implied emission factor of CH_4 from manure management from cattle.

Dröge et al. (2010) mentioned that emissions estimated within the Dutch Emission Inventory were compared to emissions estimated using Guidebook emission factors and Dutch statistics for the year 2005. The objective was to explore the quality of both methods and to find major differences and similarities. The comparison shows that for most sources, emission estimates are within uncertainty ranges for both methodologies, especially for sources where a higher Tier (more detailed) methodology is used to estimate the emissions. This is in line with the Guidelines which indicate that for key categories a more detailed methodology should be used. The comparison also shows some surprising differences, such as large differences in emission factors (especially Tier 1) and missing sources which have not been included in the Guidebook. This comparison is shown to be a useful tool to identify areas where improvements and further research are necessary.

Staebler et al. (2009) investigated NH_3 emissions plume from a beef feedlot in order to improve the Canadian NH_3 emission inventories and air quality forecasting capabilities. Lindley et al. (2000) investigated the uncertainties in the compilation of spatially resolved emission inventories. These investigations indicated that there are a number of variations between inventory results in terms of overall emission magnitudes and their spatial distribution. O'Shaughnessy and Altmaier (2011) conducted investigations to determine optimal settings applied to the plume dispersion model, AERMOD, for accurately determining the spatial distribution of hydrogen sulfide (H_2S) concentrations in the vicinity of swine concentrated animal feeding operations (CAFOs). They found that when simulating the spatial distribution of H_2S around a hypothetical large swine CAFO (1 M kg), concentrations within 0.5 km from the CAFO exceeded 25 ppb and dropped to 2 ppb within 6 km of the CAFO. These values compare to a level of 30 ppb that has been determined by the State of Iowa as a threshold level for ambient H_2S levels.

Bannink et al. (2011) stated that uncertainty of model predictions of CH_4 emission is determined mostly by errors in feed intake estimation, in the representation of the stoichiometry of production of volatile fatty acids (VFA) from fermented substrate, and in the acidity of rumen contents. Further uncertainty of predicted CH_4 emission can be due to errors in estimation of dietary composition of ingredients and in chemical compositions of dietary components. They added that prediction of CH_4 should not solely focus on representing effects of nutrition on overall digestion and apparent feed utilization by cows, but that additional attention is needed to address effects of nutrition on intra-ruminal fermentation conditions, and their effects on formation of VFA and the rumen hydrogen balance.

7 Paradigms of national emissions inventories

Hyde et al. (2003) stated that the estimated total emissions from Irish agriculture were 89.9 and 91.8 kt NH_3-N for 1991 and 2010, respectively. Cattle farming account for more than 75% of total emissions. The largest emission factors found included 46.9 g NH_3-N lu^{-1} d^{-1} for cattle housing, 29.5 g NH_3-N lu^{-1} d^{-1} for pig housing and 150 g NH_3-N lu^{-1} d^{-1} for housed broilers (lu being equivalent to 500 kg live weight). They added that abatement strategies should be implemented to achieve the agreed national emission reduction target by 12% in 2010. They also found that strategies for reducing emissions from the land spreading of manure offer the greatest potential to achieve target levels.

Yamaji et al. (2003) estimated the CH_4 emissions from livestock in South, Southeast, and East Asia and found to be about 29.9 Tg CH_4 in 2000 using the Food and Agriculture Organization database and district-level data on regional activity and emission factors, considering regional specificities. These emissions consisted of 25.9 Tg CH_4 from enteric fermentation and 4.0 Tg CH_4 from livestock manure management systems. India had the greatest production, with 11.8 Tg CH_4 from livestock, primarily cattle and buffaloes. China was also a high-emission country, producing about 10.4 Tg CH_4. They found that the total CH_4 emissions from livestock increased by an average of 2% per annum from 1965 to 2000 in Asia.

Gac et al. (2007) developed a national gas emissions inventory resulting from manure management and the data was drawn up for the year 2003 in France. Total NH_3 emissions were estimated at 382 kt N, mainly arising from cattle (72%). Greenhouse gas emissions were estimated at 14.0 Tg CO_2-Equation for N_2O and 10.2 Tg CO_2-Equation for CH_4. Most of the N_2O emissions occurred after the deposition of manure on soil during cattle grazing, while the CH_4 was mainly produced during the period when cattle manure remained in livestock buildings and in outside storage facilities.

Zhou et al. (2007) presented a systematic estimation of CH_4 and N_2O emissions during 1949–2003, based on the local measurement and IPCC guidelines, in order to investigate the GHG emissions from enteric fermentation and manure management of livestock and poultry industry in China. As far as GHG emissions are concerned among livestock, swine is found to hold major position followed by goat and sheep, while among poultry chicken has the major place and is followed by duck and geese. CH_4 emissions from enteric fermentation are estimated to have increased from 3.04 Tg in 1949 to 10.13 Tg in 2003, an average annual growth rate of 2.2%, and CH_4 emissions from manure management have increased from 0.16 Tg in 1949 to 1.06 Tg in 2003, an annual growth rate of 3.5%, while N_2O emissions from manure management have increased from 47.76 to 241.2 Gg in 2003, with an annual growth rate of 3.0%. The total GHG emissions have increased from 82.01 Tg CO_2 Equation in 1949 to 309.76 Tg CO_2 Equation in 2003, an annual growth rate of 2.4%. The estimation of livestock CH_4 and N_2O emissions in China from 1949 to 2003 is shown to be consistent with a linear growth model, and the reduction of GHG emissions is thus considered an urgent and arduous task for the Chinese livestock industry.

Aljaloud et al. (2011) developed a national emissions inventory for Saudi Arabia using data derived from farm surveys and national statistical records from 2003 to 2007, as well as Tier 1 and 2 emission factors for enteric fermentation and manure management as proposed by the Intergovernmental Panel on Climate Change. They reported that the national CH_4 emission inventory in Saudi Arabia, developed using Tier 1 default factors, increased from 87,069 t in 2003 to 100,971 t in 2007, with a CO_2 equivalent of 2,176,735 and 2,524,287 t, respectively. The contribution to the inventory varied considerably among provinces, ranging from 0.1% (Northern) to 40.6% (Riyadh) in 2007 when horse data were not included. Among animal species, dairy cattle and sheep were the biggest CH_4 producers (37.7% and 35.5%, respectively), followed by camels (13.4%), goats (11.4%), chickens (1.6%) and horses (0.4%). National CH_4 emissions consisted of 0.87 from enteric fermentation and 0.13 from manure storage. Tier 2 CH_4 emission factors were developed for dairy cattle and sheep, with each species divided into two groups (i.e., specialized high genetic merit and traditional low genetic merit), and each group of cattle and sheep were further divided into subgroups according to their physiological states. These factors for enteric fermentation and manure management were all different from Tier 1 default factors, and resulted in a decrease of 2088 t in CH_4 emission inventories (CO_2 equivalent of 52,200 t) from cattle and sheep when estimated using Tier 2, rather than Tier 1, factors. In comparison with other countries, the national CH_4 emission inventory from domestic animals in Saudi Arabia is relatively small in terms of CH_4 emissions per unit of land area or human population. These results can be used, with data from other agricultural sectors, to develop a national greenhouse gas inventory for the entire agricultural sector of Saudi Arabia.

Velthof et al. (2012) stated that the NEMA results show that the total NH_3 emission from agriculture in the Netherlands in 2009 was 88.8 Gg NH_3-N, of which 50% from housing, 37% from manure application, 9% from mineral N fertilizer, 3% from outside manure storage, and 1% from grazing. Cattle farming are the dominant source of NH_3 in the Netherlands (about 50% of the total NH_3 emission). The NH_3 emission expressed as percentage of the excreted N was 22% of the excreted N for poultry, 20% for pigs, 15% for cattle, and 12% for other livestock, which is mainly related to differences in emissions from housing systems. The calculated NH_3 emission was most sensitive to changes in the fraction of TAN in the excreted manure and to the emission factor of manure application. From 2011, NEMA has been used as official methodology to calculate the national NH_3 emission from agriculture in The Netherlands.

8 Emissions abatement techniques

Around 75% of ammonia (NH_3) emissions come from livestock production (Reinhardt-Hanisch, 2008).

Emissions occur at all stages of manure management: from buildings housing livestock; during manure storage; following manure application to land; and from urine deposited by livestock on pastures during grazing Ammoniacal nitrogen (total ammoniacal-nitrogen, TAN) in livestock excreta is the main source of NH_3. At each stage of manure management TAN may be lost, mainly as NH_3, and the remainder passed to the next stage. Hence, measures to reduce NH_3 emissions at the various stages of manure management are interdependent, and the accumulative reduction achieved by combinations of measures is not simply additive. This TAN-flow concept enables rapid and easy estimation of the consequences of NH_3 abatement at one stage of manure management (upstream) on NH_3 emissions at later stages (downstream), and gives unbiased assessment of the most cost-effective measures. Rapid incorporation of manures into arable land is one of the most cost-effective measures to reduce NH_3 emissions, while covering manure stores and applying slurry by band spreader or injection are more cost-effective than measures to reduce emissions from buildings. These measures are likely to rank highly in most European countries (Webb et al., 2005). The implementation of urease inhibitors is effective in reducing ammonia emissions from cattle and pig slurry (Reinhardt-Hanisch, 2008). She added that the basic investigations on urease inhibitors afforded an important contribution to the expansion of knowledge in this area, and will lead on the other hand to develop new techniques in order to reduce the NH_3 emissions from livestock housing. Rahman et al. (2011) evaluated the effectiveness of the Digest3+3© microbial additive for reducing odor and pollutant gas emission from a swine gestation-farrowing operation, where the additive was used to treat the deep pits to be compared with other untreated pits. However, they found no significant differences in terms of odor, NH_3, and H_2S concentrations and emissions between treated and untreated units. Overall, the microbial treatment had very little effect in reducing odor, ammonia, and hydrogen sulfide emission. Wheeler et al. (2011a) stated that amendments can be practical and cost-effective for reducing NH_3 and GHG emissions from dairy manure. They found that six amendment products that acted as microbial digest, oxidizing agent, masking agent or adsorbent significantly can reduce NH_3 by more than 10%. They added that microbial digest/enzymes with nitrogen substrate appeared effective in reducing CH_4 fluxes. For both CH_4 and CO_2 fluxes, aging the manure slurry for 30 d can significantly reduce gas production.

Carew (2010) stated that diet manipulation to improve animal N utilization efficiency and land spreading techniques to inject or incorporate manure into the soil are the most effective measures to reduce livestock NH_3 emissions compared to housing and manure storage techniques. While biofilters, air scrubbers and urine/feces separation techniques offer potential opportunities to lower emissions, these options are expensive and impractical to implement on commercial farms. Further research is needed to understand the factors limiting livestock producers adopting mitigating strategies to reduce emissions since a whole-farm system approach can provide a modeling framework to evaluate the feasibility and cost-effectiveness of abatement measures. Sommer et al. (2004) stated that daily flushing of slurry from cattle houses would reduce total annual CH_4 + N_2O emissions by 35% (CO_2 Equation), and that cooling of pig slurry in-house would reduce total annual CH_4 + N_2O emissions by 21% (CO_2 Equation). Giltrap et al. (2010) investigated the effect of NI dicyandiamide (DCD) on transformations of N to nitrate (NO_3^-) and subsequent reduction to N_2O in a grazed pasture system receiving cow urine, where the DCD was able to decrease the nitrification rate. Based on this study, an issue can be raised: is the DCD able to be used as an inhibitor of N_2O emissions from floor inside a livestock building?

Liquid manure storage facilities are sources of gaseous emissions of NH_3 and greenhouse gases especially CH_4 and N_2O. Additives can reduce gaseous emissions from swine waste lagoons and pits. The additives have the potential to reduce methane emissions from anaerobic swine lagoons (Shah and Kolar, 2012). Different materials for covering liquid manure storage facilities have been investigated and are in use for mitigating odor and ammonia emissions (Sommer et al., 1993; Williams, 2003). These materials abate also methane and nitrous oxide emissions (Berg et al., 2006). Manipulating the balance between ammonia and ammonium by lowering the pH-value of slurry is another measure to reduce emissions (Stevens et al., 1989; Oenema and Velthof, 1993; Hendriks and Vrielink, 1997; Kroodsma and Ogink, 1997; Martinez et al., 1997; Beck and Burton, 1998; Pedersen, 2003).

Ammonia and methane emissions can be controlled by pH-value. Manipulating the pH-value of slurry has an effect on the balance between ammonia and ammonium. The pH-values of untreated slurries range between seven up to eight usually. Lowering the pH reduces the gaseous emission. From former investigations it is known that a slurry pH around 5.5 can reduce ammonia emission by 80% to 90% (Al-Kanani et al., 1992; Berg et al., 2006; Husted et al., 1991; Li et al., 2006; Pain et al., 1990; Stevens et al., 1989). A slurry pH below 4.5 nearly avoids ammonia emission (Hartung and Phillips, 1994). The pH-value influences the activities of microorganisms. Higher methane production occurs, when the pH-value is between 6 and 7 (Lay et al., 1997). A slurry pH below 6 is necessary to reduce methane emission and below 5 impede methane formation (Berg et. al, 2006). Whereas the use of inorganic acids has several disadvantages, using organic acids is a promising possibility to reduce not only ammonia but also methane and nitrous oxide emissions (Berg and Hoernig, 1997; Berg and Pazsiczki, 2003; Berg, 2003). Previous studies have evaluated additives for reducing gaseous emissions from manure in 5 laboratory using glass jars or plexiglass tanks and a multi-gas monitor (Wheeler et al., 2011b; Reinhardt-Hanisch, 2008; Berg et al., 2006). Zhang et al. (2008) confirmed the generally positive impacts of anaerobic and aerobic treatment on the reductions of methane and volatile organic compounds (VOCs). However, the effects of anaerobic and aerobic treatment varied over the time of storage, especially for VOCs. They indicated that to achieve significant reductions in VOC emission the storage time of anaerobic digester or aerobic reactor effluent should be limited to no more than 84 days.

On the other hand, adjusting the housing system and the management has the potential to decrease the emissions. For instance, Dekker et al. (2011) stated that using organic laying hen husbandry in aviary systems instead of single-tiered systems has the potential to reduce emissions of NH_3, N_2O, and CH_4; further reductions might be realized by changes in litter management. Regarding manure management, Haeussermann et al. (2006) stated that methane emissions can be significantly reduced by complete slurry removal between the fattening periods and subsequent cleaning of the slurry pits in pig housing. Additionally, the release of methane from indoor slurry storage can be influenced by availability of oxygen and volatile solids, pH value, substrate temperature, retention time, and presence of inhibiting compounds. These factors should be further investigated to develop emissions abatement techniques. Special considerations should be given to avoid increasing specific gas emissions while abating another one. For instance, Brink et al. (2001) found that abating agricultural emissions of NH_3 may cause releases of N_2O from this sector up to 15% higher than in the case of no NH_3 control. Ngwabie et al. (2011) stated that increased knowledge of the factors that affect emissions from livestock barns may lead to a better understanding of daily (between different days) and diurnal (within a specific day) variations in emissions, an improvement of mitigation methods and a refinement of emission models. Animal activity, animal weight, indoor air temperature and relative humidity have influence on carbon dioxide, methane, and ammonia emissions. Emission variations emphasized the need for measurements during different times within the day and during the growing period in order to obtain reliable data for assessing abatement techniques.

Sommer and Hutchings (2001) reviewed ammonia emission from field applied manure and its reduction. They found that the use of trail hoses, pre- or post-application cultivation, and reduction in slurry viscosity, choice of application rate and timing and slurry injection are considered as reduction techniques. The most effective methods of reducing ammonia emissions were concluded to be incorporation of the animal slurry and farmyard manure or slurry injection. Incorporation should be as close to the application as possible, especially after slurry application, as loss rates are high in the first hours after application. Injection is a very efficient reduction technique, provided the slurry is applied at rates that can be contained in the furrows made by the injector tine.

Melse et al. (2009) stated that the EU emission norms for PM have come into effect recently, may be limited for continuation and/or expansion of intensive livestock operations in the near future, alongside existing ammonia and odor emission standards. Therefore, the implementation of dust abatement techniques is crucial. Mostafa (2008) listed several means for reduction of dust in/from livestock houses, such as: (1) dust suppression with spraying oil and/or water, (2) ionization, (3) cleaning the air by oxidization using chemical compounds called “oxidants”, (4) use of windbreaks, (5) implementation of de-dusters, scrubbers and filters (dry and wet).

The EMEP/EEA air pollutant emission inventory guidebook provides greenhouse emission abatement measures for animal husbandry and manure management in form of Best Available Technique (BAT) for each case, type of manure, land use, limits of applicability, emission reduction (%), availability for different farms (EMEP/EEA, 2009). On the other hand, the Executive Body for the Convention on Long-Range Transboundary Air Pollution (2007b) provided a guidance document on control techniques for preventing and abating emissions of ammonia. Moreover, the European Commission (2003) provided an Integrated Pollution Prevention and Control (IPPC) reporting on the BAT for intensive rearing of poultry and pigs.

9 Discussion

Based on a literature review of several studies, Crosson et al. (2011) criticized the IPCC methodologies which provide a set of generalized guidelines for compiling and reporting national inventories and, as such, provide a transparent and consistent framework for comparing national GHG emissions at various times. However, limitations with respect to this national and sectoral approach undermine the usefulness of the methodology for modeling at the farm level. In this respect, whole farm modeling is widely employed for farm level GHG emissions modeling. Despite the increased flexibility and sensitivity of these models to capture farm level activity, concerns and variation relating to data sources, boundaries (i.e., which emission sources are included), emission factors and allocation approaches exist, and they limit direct comparison among studies. Thus, it is critical that assumptions made in this regard are clearly outlined. Furthermore, uncertainty analysis has a key role to play given the range of GHG emission factors reported and the inherent uncertainty involved in agricultural processes. Nevertheless, in terms of developing and assessing mitigation policies to reduce GHG emissions from livestock systems, whole farm approaches provide the most robust and comprehensive approach to developing and implementing effective strategies.

On the other hand, the livestock housing system and design greatly affect the livestock’s microclimate (especially temperature and air velocity near animals) which has a direct influence on gas emissions from livestock barns. Several studies investigated the effect of barn design on animals’ microenvironment, where the barn design and accordingly the management of the different components (including cooling or heating systems) inside the barn have a great effect on the indoor bioenvironment. The most important –in this context- bioenvironmental components are the temperature and the air velocity which when increase or decrease the emissions rates are directly affected (Samer, 2004; Hatem et al., 2004a,b; Hatem et al., 2006; Samer, 2008; Samer et al., 2008a,c; Ngwabie et al., 2010; Samer, 2010a,b,c; Samer, 2011a,c; Pereira et al., 2012; Samer, 2012a; Samer et al., 2012a,b; Samer et al., 2013a; Samer 2013), where the highest average NH_3 emission coincided with higher environmental temperatures (Adviento-Borbe et al., 2010). Ammonia emissions are about twice higher at manure temperatures of about 25ºC than emissions at about 15ºC (Ngwabie et al., 2010). Decreasing velocity, turbulence intensity and liquid temperature are shown to reduce the ammonia emission rates. The emission rates are more sensitive to the change of velocity at a low velocity compared to change of velocity at a higher velocity range, which corresponds with the conclusion that the boundary layer thickness of velocity increases sharply when velocity is changed from 0.2 m s^{-1} to 0.1 m s^{-1}. In addition, the emission rates are more sensitive to the change of temperature at a higher temperature than at a lower liquid temperature range (Rong et al., 2009). Pereira et al. (2012) stated that increasing temperature from 5 to 35°C significantly increases potential NH_3, CO_2 and CH_4 emissions from excreta of the housed cattle, but it does not significantly influence N_2O emissions. Also, the diet supplied to lactating cows leads to significantly higher NH_3, N_2O and CO_2 emissions relative to heifers and dry cows. On the other hand, seasonal variations of gaseous emissions, regional conditions (i.e. climatic conditions: hot or cold climates) have direct impact on the gaseous emissions flow rates which increase in summer (Gilliland et al., 2006; Haeussermann et al., 2006; Samer, 2011d). Further, the design of the implemented ventilation system, which is mainly natural ventilation system for several livestock types, as well as the ventilation rate or the air exchange rate have direct effect of the amount of emissions flow rates of the different gases (Samer et al., 2011a,b,c,d,e,f,g; Samer et al., 2012c). The flux of emissions of harmful gases from naturally ventilated buildings is dependent on wind velocity (both speed and direction) and turbulence levels inside and over the outside of the building, thus emission mass flow is highly variable and difficult to estimate (Ngwabie et al., 2009; Van Buggenhout et al., 2009; Samer et al., 2013b). Morsing et al. (2008) hypothesized that the effects on gas emissions are as a consequence of changing airflow patterns and different types of flow in the boundary layer between the slurry and ventilation air. Odors and gases emitted from animal houses are strongly related to airflows. Haeussermann et al. (2006) found a clear influence of the indoor temperature – and therefore of the ventilation strategy – on the level of the methane emission rate for mean daily temperatures above 25 °C. On the other hand, the factors-of-influence (FOI) that strongly influence the dispersion of NH_3 are: NH_3-mass-flow, internal and external temperatures, mean and turbulent wind components in horizontal and vertical directions, atmospheric stability, and exhaust air height where the continuous measurement of NH_3 remains a challenging and costly enterprise, in terms of capital investment, running costs or both (Von Bobrutzki et al., 2010; Von Bobrutzki et al., 2011). Additionally, the floor type plays an important role where according to Pereira et al. (2011) the gaseous emissions are significantly greater from the solid floor relative to the slatted floor at all temperatures (5, 15 and 25°C). Low emission values can only be achieved by reducing the emission source surfaces, by decreasing temperature and air velocity near the source, and minimizing air volumetric flow rates throughout the livestock buildings (Bjorneberg et al., 2009; Blanes-Vidal et al., 2007; Gay et al., 2003). Nevertheless, the type of manure management and the design of the handling system (Samer et al., 2008d; Samer, 2011b; Samer, 2012b) will have direct effect on the emissions level. For instance, Moset et al. (2012) found that there was relevant transformation of the more degradable organic matter (OM) into soluble OM during the first week of aged fattening pig slurry storage. This transformation is more pronounced in the slurry with a higher initial OM concentration in raw slurry (RS) than in separated slurry (SS), indicating a higher hydrolytic, acidogenic and acetogenic activity, as well as higher rate of urea mineralization and nitrogen denitrification rate at the beginning of the storage period in RS than in SS. Special considerations for the emission rate should be given if a biogas plant has been implemented in the livestock farm, where such implementation influences considerably the emissions rate of the different gases as the manure is handled differently from aerobic stores (Samer, 2010d; Samer, 2012b). For instance, the type of manure management in caged-layer poultry has a great impact on the level of emissions, where belt/composting has less than half the total emissions of conventional deep-pit caged-layer systems (Keener and Zhao, 2008). Another issue should be considered which is the farmstead layout especially regarding the orientation of different buildings in respect to the prevailing wind as well as the distance between the adjacent buildings (Samer et al., 2008b), where such layout parameters will affect the immission and transmission of the gases and eventually affect the accuracy of the emission factors. The abovementioned statements confirm the need to relate the emissions inventories to whole farm systems and treating each farm as a special case where farm survey would be an effective method, which agree with Crosson et al. (2011). Sommer et al. (2006) stated that a detailed knowledge of the processes of NH_3 transfer from the manure and transport to the free atmosphere will contribute to development of techniques and housing designs that will contribute to the reduction of NH_3 emission to the atmosphere.

10 Conclusions

Current emissions inventories lack of detailed information on emissions from livestock buildings. They focus on emissions from animal husbandry (mainly enteric fermentation) as well as manure storages and field application of manure. However, livestock barns encompass large contaminated areas that form an important emitting source of pollutants. Besides, manure can be stored inside the livestock barns for short periods and this is also a considerable source of emissions. One key issue is to integrate the emission factors, mass flow models and algorithms that consider emissions from livestock barns with those of animal husbandry and manure management (handling, stores, and field application) to allow developing comprehensive emissions inventories.

On the other hand, this study focuses on emissions and emissions inventories from animal husbandry, manure management and livestock buildings. Nevertheless, these emissions should be integrated with those from soils in order to form agricultural emissions inventory and further integrated with the other main sources of emissions (waste, energy, and industry) in order to make a regional/territorial emissions inventory for a specific region/province. Subsequently, the different regional emissions inventories are then compiled together to make the national inventory.

According to the issues raised in this study, it can be further concluded that:

1) It is crucial to consider the emissions from the livestock buildings in the emissions inventories, where the enteric fermentation and the manure stores are not the only sources of emissions from animal husbandry. One key issue is to consider the manure management system inside the building (slatted floor, manure scrapper, litter-based system…), floor type, building design, and ventilation system where these parameters have discernible influence on the emissions flow rates from livestock building. Therefore, each farm should be treated as special case, where some farms may have applied emissions abatement techniques and other farms may have not, and therefore farm survey is necessary to be integrated into the emissions inventorying procedures. Furthermore, the emission rates depends not only the barn design and the housing system but also on specific parts/compartments inside the barn (e.g. the different rooms inside a swine barn). This should be considered when inventorying the emissions.

2) The contribution to a national emissions inventory varies discernibly among the different states, provinces or governorates within the same country depending on the geographical distribution of the livestock farms, where the geographical area which encompasses the largest number of livestock would be the most contributing area to the emissions inventory. However, the management practices (especially manure handling) are a detrimental factor which if effective in mitigating the emissions, they minimize the contribution of this specific area to the national inventory. The other factor is the climatic conditions (temperature, wind velocity). On the other hand, the composition of animals' feed ration should be considered.

3) Focusing on "whole farm system approach" would be an asset. Every farm should yearly submit data and info to the agricultural and environmental authorities. It is practical to use the submitted data from each farm (e.g., number, age, weight, species of the animals as well as the implemented manure handling systems, manure field application methods, use of chemical fertilizers etc.) to develop an emission inventory for each farm and then to develop an agricultural emissions inventory to a certain geographical area based on the emissions inventories of the farms located in this area, and then the higher level onwards reaching the country level. The developed

agricultural emissions inventory should be integrated to other inventories in the same area (e.g., inventories on Energy, Industrial Processes and Product Use, Forestry, Waste, and other emissions sources) to develop the national emissions inventory of a certain pollutant.

4) The calculation procedure making use of partial emission factors for the various sources of emissions (animal house, storage, manure application, etc.) is being replaced by a mass flow concept.

5) Developing emission algorithms and then environmental software for making emissions inventories would be an asset.

6) Developing processing tools for emissions inventories, especially software systems, would be an asset.

7) Emissions of the harmful gases (NH_3, CH_4 and N_2O) should be reported annually in the national emissions inventory which should be updated annually.

8) A national emissions inventory should be compared with other countries, in terms of pollutant emissions per unit of land area or human population.

9) Inventorying the emissions of the different gases allows developing emissions mitigation strategies.

10) The uncertainty of emissions inventories represents a challenge. Therefore, continuous evaluation and improvement of emissions inventories is essential.

11) It is insufficient to inventory the emissions, rather investigating the plume, dispersion, spatial distribution and fate of emissions is necessary to improve the emissions inventories. Afterwards, developing air quality forecasting tools would be an asset.

12) Developing emissions inventory of particulate matter (PM) will be essential in the next decade.

13) Inventorying volatile organic compounds (VOCs), odors, nitric oxide (NO), hydrogen sulfide (H_2S), and non-methane volatile organic compounds (NMVOCs) might be the state-of-the-art in the near future.

14) Understanding of the processes of CH_4, N_2O and NH_3 mass transfer from the manure and transport to the free atmosphere will contribute to development of emissions abatement techniques and housing designs, and will contribute to the reduction of gaseous emissions to the atmosphere.

References

Adviento-Borbe, M. A. A.,Wheeler, E. F., N. E. Brown, P. A. Topper, R. E. Graves, V. A. Ishler, and G. A. Varga. 2010. Ammonia and greenhouse gas flux from manure in freestall barn with dairy cows on precision fed rations. *Transactions of the ASABE*, 53(4): 1251-1266.

Aljaloud, A. A., T. Yan, and A. M. Abdukader. 2011. Development of a national methane emission inventory for domestic livestock in Saudi Arabia. *Animal Feed Science and Technology*, 166–167(2011): 619– 627.

Al-Kanani, T., E. Akochi, A. F. Mackenzie, I. Alli, and S. Barrington. 1992. Organic and inorganic amendments to reduce ammonia losses from liquid hog manure. *Journal of Environmental Quality*, 21(4): 709–715.

Bannink, A., M. W. van Schijndel, and J. Dijkstra. 2011. A model of enteric fermentation in dairy cows to estimate methane emission for the Dutch National Inventory Report using the IPCC Tier 3 approach. *Animal Feed Science and Technology*, 166–167(2011): 603– 618.

Battye, W., V. P. Aneja, and P. A. Roelle. 2003. Evaluation and improvement of ammonia emissions inventories. *Atmospheric Environment*, 37(2003): 3873–3883.

Beck, J., and C. Burton. 1998. Manure treatment techniques in Europe—result of a EU Concerted Action. In: Proceedings of the International Conference on Agricultural Engineering, AgEng 98, Oslo, Norway, August 24–27, pp. 211–212.

Berg, W. 2003. Reducing ammonia emissions by combining covering and acidifying liquid manure. In: Proceedings of the Third International Conference on Air Pollution from Agricultural Operations, Raleigh, NC, USA, October 12–15, pp. 174–182.

Berg, W. 1999. Technology assessment—livestock management. Anim. Res. Dev. 50, 98–109.

Berg, W., and I. Pazsiczki. 2003. Reducing emissions by combining slurry covering and acidifying. In: Proceedings of the International Symposium on Gaseous and Odour Emissions from Animal Production Facilities, Horsens, Denmark, June 1–4, pp. 460–468.

Berg, W., and G. Hoernig. 1997. Emission reduction by acidification of slurry—investigations and assessment. In: Voormans, J. A. M., Monteny, G. J. (Eds.), Proceedings of the

International Symposium on Ammonia and Odour Control from Animal Production Facilities, Vinkeloord, The Netherlands, October 6–10, pp. 459–466.

Berg, W., R. Brunsch, and I. Pazsiczki. 2006. Greenhouse gas emissions from covered slurry compared with uncovered during storage. *Agriculture, Ecosystems and Environment*, 112(2-3): 129–134.

Bjorneberg, D. L., A. B. Leytem, D. T. Westermann, P. R. Griffiths, L. Shao, and M. J. Pollard. 2009. Measurements of atmospheric ammonia, methane, and nitrous oxide at a concentrated dairy production facility in southern Idaho using open-path FTIR spectrometry. *Transactions of the ASABE*, 52(5): 1749-1756.

Blanes-Vidal, V., P. A. Topper, and E. F. Wheeler. 2007. Validation of ammonia emissions fromdairy cow manure estimated with a non-steady-state, recirculation flux chamber with whole building emissions. *Transactions of the ASABE*, 50(2): 633-640.

Brink, C., C. Kroeze, and Z. Klimont. 2001. Ammonia abatement and its impact on emissions of nitrous oxide and methane - Part 2: application for Europe. *Atmospheric Environment*, 35(36): 6313–6325.

Carew, R. 2010. Ammonia emissions from livestock industries in Canada: Feasibility of abatement strategies. *Environmental Pollution*, 158(8): 2618-2626.

Chadwick, D., S. Sommer, R. Thorman, D. Fangueiro, L. Cardenas, B. Amon, and T. Misselbrook. 2011. Manure management: Implications for greenhouse gas emissions. *Animal Feed Science and Technology*, 166–167(2011): 514– 531.

Costa, A., and M. Guarino. 2009. Definition of yearly emission factor of dust and greenhouse gases through continuous measurements in swine husbandry. *Atmospheric Environment,* 43(8): 1548–1556.

Cowell, D. A., and H. M. ApSimon. 1998. Cost-effective strategies for the abatement of ammonia emissions from European agriculture. *Atmospheric Environment*, 32(3): 573–580.

Crosson, P., L. Shalloo, D. O’Brien, G. J. Lanigan, P. A. Foley, T. M. Boland, and D. A. Kenny. 2011. A review of whole farm systems models of greenhouse gas emissions from beef and dairy cattle production systems. *Animal Feed Science and Technology*, 166-167 (2011): 29-45.

Dammgen, U., and J. Webb. 2006. The development of the EMEP/CORINAIR Guidebook with respect to the emissions of different nitrogen and carbon species from animal production. *Agriculture, Ecosystems and Environment*, 112(2-3): 241–248.

Dekker, S. E. M., A. J. A. Aarnink, I. J. M. de Boer, and P. W. G. Groot Koerkamp. 2011. Emissions of ammonia, nitrous oxide, and methane from aviaries with organic laying hen husbandry. *Biosystems Engineering*, 110(2): 123-133.

Dröge, R., J. J. P. Kuenen, M. P. J. Pulles, and D. C. Heslinga. 2010. The revised EMEP/EEA Guidebook compared to the country specific inventory system in the Netherlands. *Atmospheric Environment*, 44(29): 3503-3510.

ECETOC. 1994. Ammonia emissions to in Western Europe. Technical Report 62. European Centre for Ecotoxicology and Toxicology of Chemicals, Brussels.

EMEP/EEA. 2009. EMEP/EEA air pollutant emission inventory guidebook: Animal husbandry and manure management. The European Environment Agency (EEA) and the Cooperative programme for monitoring and evaluation of the long-range transmission of air pollutants in Europe (EMEP). http://eea.europa.eu/emep-eea-guidebook

European Commission. 2003. Integrated Pollution Prevention and Control (IPPC): Intensive Rearing of Poultry and Pigs. Reference Document on Best Available Techniques for Intensive Rearing of Poultry and Pigs.

Executive Body for the Convention on Long-Range Transboundary Air Pollution. 2007a. Methods and Procedures for the Technical Review of Air Pollutant Emission Inventories Reported under the Convention and its Protocols. Economic Commission for Europe, Economic and Social Council, United Nations.

Executive Body for the Convention on Long-Range Transboundary Air Pollution. 2007b. Guidance Document on Control Techniques for Preventing and Abating Emissions of Ammonia. Economic Commission for Europe, Economic and Social Council, United Nations.

FAO. 2006. Livestock's role in climate change and air pollution. Available at: ftp://ftp.fao.org/docrep/fao/010/a0701e/A0701E03.pdf. Accessed December 2010.

Faulkner, W. B., and B. W. Shaw. 2008. Review of ammonia emission factors for United States animal agriculture. *Atmospheric Environment*, 42(27): 6567–6574.

Gac, A., F. Béline, T. Bioteau, and K. Maguet. 2007. A French inventory of gaseous emissions (CH_4, N_2O, NH_3) from livestock manure management using a mass-flow approach. *Livestock Science*, 112(3): 252–260.

Gay, S. W., D. R. Schmidt, C. J. Clanton, K. A. Janni, L. D. Jacobson, and S. Weisberg. 2003. Odor, total reduced sulfur, and ammonia emissions from animal housing facilities and manure storage units in Minnesota. *Transactions of the ASABE*, 19(3): 347-360.

Gilliland, A. B., K. Wyat Appel, R. W. Pinder, and R. L. Dennis. 2006. Seasonal NH_3 emissions for the continental united states: Inverse model estimation and evaluation. *Atmospheric Environment*, 40(26): 4986–4998.

Giltrap, D. L., J. Singh, S. Saggar, and M. Zaman. 2010. A preliminary study to model the effects of a nitrification inhibitor on nitrous oxide emissions from urine-amended pasture. *Agriculture, Ecosystems and Environment*, 136(3-4): 310–317.

Guo, J., C. Zhou. 2007. Greenhouse gas emissions and mitigation measures in Chinese agroecosystems. *Agricultural and Forest Meteorology*, 142(2-4): 270–277.

Gupta, P. K., A. K. Jha, S. Koul, P. Sharma, V. Pradhan, V. Gupta, C. Sharma, and N. Singh. 2007. Methane and nitrous oxide emission from bovine manure management practices in India. *Environmental Pollution*, 146(1): 219-224.

Haeussermann, A., E. Hartung, E. Gallmann, and T. Jungbluth. 2006. Influence of season, ventilation strategy, and slurry removal on methane emissions from pig houses. *Agriculture, Ecosystems and Environment*, 112(2006): 115–121.

Hartung, J., and V. R. Phillips. 1994. Control of gaseous emissions from livestock buildings and manure stores. *Journal Agricultural Engineering Research*, 57(3): 173–189.

Hatem, M. H., R. R. Sadek, and M. Samer. 2006. Effects of Shed Height and Orientation on Dairy Cows` Microclimate, Cooling System Efficiency and Milk Productivity. Proceedings of XVI CIGR World Congress, pp. 413-414, VDI Verlag, Düsseldorf, Germany.

Hatem, M. H., R. R. Sadek, and M. Samer. 2004a. Shed height effect on dairy cows microclimate. *Misr Journal of Agricultural Engineering*, 21(2): 289 - 304.

Hatem, M. H., R. R. Sadek, and M. Samer. 2004b. Cooling, shed height and shed orientation affecting dairy cows microclimate. *Misr Journal of Agricultural Engineering*, 21(3): 714 - 726.

Hellsten, S., U. Dragosits, C. J. Place, M. Vieno, A. J. Dore, T. H. Misselbrook, Y. S. Tang, and M. A. Sutton. 2008. Modelling the spatial distribution of ammonia emissions in the UK. *Environmental Pollution*, 154(3): 370-379.

Hendriks, J. G. L., and M. G. M. Vrielink. 1997. Reducing ammonia emission from pig houses by adding or producing organic acids in pig slurry. In: Voormans, J.A.M., Monteny, G.J. (Eds.), Proceedings of the International Symposium on Ammonia and Odour Control from Animal Production Facilities, Vinkeloord, The Netherlands, October 6–10, pp. 493–501.

Husted, S., L. S. Jensen, and S. S. Jorgensen. 1991. Reducing ammonia loss from cattle slurry by the used of acidifying additives: the role of the buffer system. *Journal of the Science of Food and Agriculture*, 57(3): 335–349.

Hutchings, N. J., S. G. Sommer, J. M. Andersen, and W. A. H. Asman. 2001. A detailed ammonia emission inventory for Denmark. *Atmos. Environ*, 35(11): 1959–1968.

Hyde, B. P., O. T. Carton, P. O'Toole, and T. H. Misselbrook. 2003. A new inventory of ammonia emissions from Irish agriculture. *Atmospheric Environment*, 37(1): 55–62.

Intergovernmental Panel on Climate Change (IPCC). 2006. IPCC guidelines for national greenhouse gas inventories. In: Eggleston, H., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), The National Greenhouse Gas Inventories Programme, Intergovernmental Panel on Climate Change. IGES, Japan.

Intergovernmental Panel on Climate Change (IPCC). 2007. Climate Change 2007: Mitigation. Contribution of Working Group III to the 4th Assessment Report of the Intergovernmental Panel on Climate Change. B. Metz, O. R. Davidson, P. R. Bosch, R. Dave, and L. A. Meyer, eds. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Keener, H. M., and L. Zhao. 2008. A modified mass balance method for predicting NH_3 emissions from manure N for livestock and storage facilities. *Biosystems Engineering*, 99(1): 81 – 87.

Kroodsma, W., N. W. M. Ogink. 1997. Volatile emissions from cow cubicle houses and its reduction by immersion of the slats with acidified slurry. In: Voormans, J.A.M., Monteny, G. J. (Eds.), Proceedings of the International Symposium on Ammonia and Odour Control from Animal Production Facilities, Vinkeloord, The Netherlands, October 6–10, pp. 475–483.

Lay, J.J.; Li, Y.Y. & Noike, T. 1997. Influences of pH and moisture content on the methane production in high-solids sludge digestion. *Water Research*, 31(6), 1518–1524.

Li, H., H. Xin, and R. T. Burns. 2006. Reduction of ammonia emission from stored poultry manure using additives: zeolite, Al+ clear, Ferix-3, and PLT. ASAE Paper No. 064188, 2006 ASABE Annual International Meeting, Portland, OR, USA.

Lindley, S. J., D. E. Conlan, D. W. Raper, and A. F. R. Watson. 2000. Uncertainties in the compilation of spatially resolved emission inventories - evidence from a comparative study. *Atmospheric Environment,* 34(2000): 375-388.

Martinez, J., J. Jolivet, F. Guiziou, and G. Langeoire. 1997. Ammonia emissions from pig slurries: evaluation of acidification and the use of additives in reducing losses. In: Voormans, J. A. M., Monteny, G. J. (Eds.), Proceedings of the International Symposium on Ammonia and Odour Control from Animal Production Facilities, Vinkeloord, The Netherlands, October 6–10, pp. 485–492.

Melse, R. W., N. W. M. Ogink, and W. H. Rulkens. 2009. Overview of European and Netherlands' regulations on airborne emissions from intensive livestock production with a focus on the application of air scrubbers. *Biosystems Engineering*, 104(3): 289 – 298.

Merino, P., E. Ramirez-Fanlo, H. Arriaga, O. del Hierro, A. Artetxe, and M. Viguria. 2011. Regional inventory of methane and nitrous oxide emission from ruminant livestock in the Basque Country. *Animal Feed Science and Technology*, 166–167(2011): 628– 640.

Morsing, S., J. S. Strøm, G. Zhang, and P. Kai. 2008. Scale model experiments to determine the effects of internal airflow and floor design on gaseous emissions from animal houses. *Biosystems Engineering*, 99(1): 99-104.

Moset, V., M. Cambra-Lopez, F. Estelles, A. G. Torres, and A. Cerisuelo. 2012. Evolution of chemical composition and gas emissions from aged pig slurry during outdoor storage with and without prior solid separation. *Biosystems Engineering*, 111 (2012): 2-10.

Mostafa, E. 2008. Improvement of air quality in laying hens barn using different particle separation techniques. PhD diss. Bonn, Germany: University of Bonn.

Ngwabie, N. M., K. H. Jeppsson, S. Nimmermark, C. Swensson, and G. Gustafsson. 2009. Multi-location measurements of greenhouse gases and emission rates of methane and ammonia from a naturally-ventilated barn for dairy cows. *Biosystems Engineering*, 103(1): 68-77.

Ngwabie, N. M, K.-H. Jeppsson, G. Gustafsson, and S. Nimmermark. 2010. Influences of environmental factors and the addition of wood shavings on ammonia and odour emissions from fresh livestock manure. *Agric Eng Int: CIGR Journal*, 12(3): 68－81.

Ngwabie, N. M., K.-H. Jeppsson, S. Nimmermark, and G. Gustafsson. 2011. Effects of animal and climate parameters on gas emissions from a barn for fattening pigs. *Applied Engineering in Agriculture*, 27(6): 1027-1037.

Oenema, O., and G. L. Velthof. 1993. Denitrification in nitric-acid-treated cattle slurry during storage. *Neth. J. Agric. Sci.,* 41(1993): 63–80.

O'Shaughnessy, P. T., R. Altmaier. 2011. Use of AERMOD to determine a hydrogen sulfide emission factor for swine operations by inverse modeling. *Atmospheric Environment*, 45(27): 4617-4625.

Pain, B. F., R. B. Thompson, Y. J. Rees, and J. H. Skinner. 1990. Reducing gaseous losses of nitrogen from cattle slurry applied to grassland by the use of additives. *Journal of the Science of Food and Agriculture*, 50(2): 141–153.

Pedersen, P. 2003. Reduction of gaseous emissions from pig houses by adding sulphuric acid to the slurry. In: Proceedings of the International Symposium on Gaseous and Odour Emissions from Animal Production Facilities, Horsens, Denmark, June 1–4, pp. 257–263.

Pereira J, D. Fangueiro, T. Misselbrook, D. Chadwick, J. Coutinho, H. Trindade. 2011. Ammonia and greenhouse gas emissions from slatted and solid floors in dairy cattle houses: a scale model study. *Biosystems Engineering*, 109(2): 48-57.

Pereira, J., T. H. Misselbrook, D. R. Chadwick, J. Coutinho, and H. Trindade. 2012. Effects of temperature and dairy cattle excreta characteristics on potential ammonia and greenhouse gas emissions from housing: A laboratory study. *Biosystems Engineering*, 112(2): 138-150.

Pouliot, G., T. Pierce, H. D. van der Gon, M. Schaap, M. Moran, and U. Nopmongcol. 2012. Comparing emission inventories and model-ready emission datasets between Europe and North America for the AQMEII project. *Atmospheric Environment*, 53(2012): 4-14.

Rahman, S., T. DeSutter, and Q. Zhang. 2011. Efficacy of a microbial additive in reducing odor, ammonia, and hydrogen sulfide emissions from farrowing-gestation swine operation. *Agric Eng Int: CIGR Journal*, 13(3): 1-9.

Ramirez, A., C. de Keizer, J. P. Van der Sluijs, J. Olivier, and L. Brandes. 2008. Monte Carlo analysis of uncertainties in the Netherlands greenhouse gas emission inventory for 1990－2004. *Atmospheric Environment*, 42(35): 8263－8272.

Reidy, B., U. Dammgen, H. Dohler, B. Eurich-Menden, F. K. van Evert, N. J. Hutchings, H. H. Luesink, H. Menzi, T. H. Misselbrook, G.-J. Monteny, and J. Webb. 2008a. Comparison of models used for national agricultural ammonia emission inventories in Europe: Liquid manure systems. *Atmospheric Environment*, 42(14): 3452–3464.

Reidy, B., B. Rhim, and H. Menzi. 2008b. A new Swiss inventory of ammonia emissions from agriculture based on a survey on farm and manure management and farm-specific model calculations. *Atmospheric Environment*, 42(14): 3266–3276.

Reidy, B., J. Webb, T. H. Misselbrook, H. Menzi, H. H. Luesink, N. J. Hutchings, B. Eurich-Menden, H. Dohler, and U. Dammgen. 2009. Comparison of models used for national agricultural ammonia emission inventories in Europe: Litter-based manure systems. Atmospheric Environment, 43(9): 1632–1640.

Reinhardt-Hanisch, A. 2008. Grundlagenuntersuchungen zur Wirkung neuartiger Ureaseinhibitoren in der Nutztierhaltung, in German (Basic research on the effects of novel urease inhibitors in animal housing). PhD diss. Stuttgart, Germany: University of Hohenheim.

Rong, L., P. V. Nielsen, and G. Zhang. 2009. Effects of airflow and liquid temperature on ammonia mass transfer above an emission surface: Experimental study on emission rate. *Bioresource Technology*, 100(20): 4654–4661.

Rypdal, K., and K. Flugsrud. 2001. Sensitivity analysis as a tool for systematic reductions in greenhouse gas inventory uncertainties. *Environmental Science & Policy*, 4(2-3): 117–135.

Samaali, M., S. Francois, J.-F. Vinuesa, and J.-L. Ponche. 2007. A new tool for processing atmospheric emission inventories: Technical aspects and application to the ESCOMPTE study area. *Environmental Modelling and Software*, 22(12): 1765-1774.

Samer, M. 2013. Towards the implementation of the Green Building concept in agricultural buildings: a literature review.

Agricultural Engineering International: CIGR Journal, 15(2): 25-46.

Samer, M., M. Hatem, H. Grimm, R. Doluschitz, and T. Jungbluth. 2013a. A software for planning loose yards and designing concrete constructions for dairy farms in arid and semi-arid zones. *Emirates Journal of Food and Agriculture*, 25(3): 238-249.

Samer, M., H.-J. Müller, M. Fiedler, W. Berg, and R. Brunsch. 2013b. Measurement of ventilation rate in livestock buildings with radioactive tracer gas technique: theory and methodology. *Indoor and Built Environment*, DOI: 10.1177/1420326X13481988.

Samer, M. 2012a. Reconstruction of Old Gutter-Connected Dairy Barns: A Case Study. Proceedings of the 2012 American Society of Agricultural and Biological Engineers (ASABE) Annual International Meeting, 29.07-01.08.2012, Paper No. 121341061, Dallas, Texas, USA.

Samer, M. 2012b. Biogas Plant Constructions, pp. 343-368. *In*: Biogas, S. Kumar (ed.), ISBN 978-953-51-0204-5. Rijeka, Croatia: InTech.

Samer, M., M. Hatem, H. Grimm, R. Doluschitz, and T. Jungbluth. 2012a. An expert system for planning and designing dairy farms in hot climates. *Agricultural Engineering International: CIGR Journal*, 14(1): 1-15.

Samer, M., C. Ammon, Loebsin, M. Fiedler, W. Berg, P. Sanftleben, and R. Brunsch. 2012b. Moisture balance and tracer gas technique for ventilation rates measurement and greenhouse gases and ammonia emissions quantification in naturally ventilated buildings. *Building and Environment*, 50(4): 10-20.

Samer, M., W. Berg, M. Fiedler, K. von Bobrutzki, C. Ammon, P. Sanftleben, and R. Brunsch. 2012c. A comparative study among H_2O-balance, heat balance, CO_2-balance and radioactive tracer gas technique for airflow rates measurement in naturally ventilated dairy barns. Proceedings of the Ninth International Livestock Environment Symposium (ASABE ILES IX), Paper No. ILES12-0079, ASABE, Valencia, Spain.

Samer, M. 2011a. Effect of cowshed design and cooling strategy on welfare and productivity of dairy cows. *Journal of Agricultural Science and Technology A*, 1(6): 848-857.

Samer, M. 2011b. How to construct manure storages and handling systems. *IST Transactions of Biosystems and Agricultural Engineering*, 1(1): 1-7.

Samer, M. 2011c. Implementation of cooling systems to enhance dairy cows' microenvironment. *Journal of Environmental Science and Engineering*, 5(12): 1654-1661.

Samer, M. 2011d. Seasonal variations of gaseous emissions from a naturally ventilated dairy barn. *Misr Journal of Agricultural Engineering*, 28(4): 1162-1177.

Samer, M., C. Loebsin, M. Fiedler, C. Ammon, W. Berg, P. Sanftleben, and R. Brunsch. 2011a. Heat balance and tracer gas technique for airflow rates measurement and gaseous emissions quantification in naturally ventilated livestock buildings. *Energy and Buildings*, 43(12): 3718-3728.

Samer, M., M. Fiedler, H. J. Müller, M. Gläser, C. Ammon, W. Berg, P. Sanftleben, and R. Brunsch. 2011b. Winter measurements of air exchange rates using tracer gas technique and quantification of gaseous emissions from a naturally ventilated dairy barn. *Applied Engineering in Agriculture*, 27(6): 1015-1025.

Samer, M., H.-J. Müller, M. Fiedler, C. Ammon, M. Gläser, W. Berg, P. Sanftleben, and R. Brunsch. 2011c. Developing the ^{85}Kr tracer gas technique for air exchange rate measurements in naturally ventilated animal buildings. *Biosystems Engineering*, 109(4): 276-287.

Samer, M., W. Berg, H.-J. Müller, M. Fiedler, M. Gläser, C. Ammon, P. Sanftleben, and R. Brunsch. 2011d. Radioactive ^{85}Kr and CO_2-balance for ventilation rate measurements and gaseous emissions quantification through naturally ventilated barns. *Transactions of the ASABE*, 54(3): 1137-1148.

Samer, M., C. Loebsin, K. von Bobrutzki, M. Fiedler, C. Ammon, W. Berg, P. Sanftleben, and R. Brunsch. 2011e. A computer program for monitoring and controlling ultrasonic anemometers for aerodynamic measurements in animal buildings. *Computers and Electronics in Agriculture*, 79(1): 1-12.

Samer, M., M. Fiedler, C. Loebsin, W. Berg, H.-J. Müller, M. Gläser, C. Ammon, P. Sanftleben, and R. Brunsch. 2011f. Tracer gas technique to estimate the ventilation rate through a naturally ventilated dairy barn. *Landtechnik*, 66(4): 286–288.

Samer, M., W. Berg, M. Fiedler, H.-J. Müller, M. Gläser, C. Ammon, R. Brunsch, C. Loebsin, O. Tober, and P. Sanftleben. 2011g. Implementation of Radioactive ^{85}Kr for Ventilation Rate Measurements in Dairy Barns. Proceedings of the 2011 American Society of Agricultural and Biological Engineers (ASABE) Annual International Meeting, pp. 847-863, Paper No. 1110679, Louisville, Kentucky, USA.

Samer, M. 2010a. Adjusting Dairy Housing in Hot Climates to Meet Animal Welfare Requirements. *Journal of Experimental Science,* 1(3): 14-18.

Samer, M. 2010b. How to Rectify Design Flaws of Dairy Housing in Hot Climates? Proceedings of XVII CIGR World Congress, Quebec City, Canada. Book of Abstracts, p. 79. The Canadian Society for Bioengineering (CSBE), Winnipeg, Manitoba, Canada.

Samer, M. 2010c. Developing and Implementing a Software Program for Configuring Three Dairy Corral Designs. *Journal of Experimental Science,* 1(3): 19-22.

Samer, M. 2010d. A Software Program for Planning and Designing Biogas Plants. *Transactions of the ASABE*, 53(4): 1277-1285.

Samer, M., Grimm, H., Hatem, M., Doluschitz, R., Jungbluth, T. 2008a. Mathematical Modeling and Spark Mapping of Shade Structures for Corral Systems in Hot Climates. Proceedings of CIGR International Conference of Agricultural Engineering, Iguassu Falls City, Brazil.

Samer, M., H. Grimm, M. Hatem, R. Doluschitz, and T. Jungbluth. 2008b. Mathematical Modeling and Spark Mapping of Dairy Farmstead Layout in Hot Climates. *Misr Journal of Agricultural Engineering*, 25(3): 1026 -1040.

Samer, M., H. Grimm, M. Hatem, R. Doluschitz, and T. Jungbluth. 2008c. Spreadsheet Modeling to Size Dairy Sprinkler and Fan Cooling System. Proceedings of Eighth International Livestock Environment Symposium (ASABE ILES VIII), pp. 701-708, ASABE, Iguassu Falls City, Brazil.

Samer, M., Grimm, H., Hatem, M., Doluschitz, R., Jungbluth, T. 2008d. Mathematical Modeling and Spark Mapping for Construction of Aerobic Treatment Systems and their Manure Handling System. Proceedings of International Conference on Agricultural Engineering, Book of Abstracts p. 28, EurAgEng, Hersonissos, Crete, Greece.

Samer, M. 2008. An expert system for planning and designing dairy farms in hot climates. PhD Dissertation, University of Hohenheim, Stuttgart, Germany, VDI-MEG, ISSN 0931-6264, Script 472.

Samer, M. 2004. Engineering Parameters Affecting Dairy Cows Microclimate and their Productivity under Egyptian Conditions. M.Sc. Thesis, Cairo University, 176 p.

Schuurkes, J., and R. Mosello. 1988. The role of external ammonium inputs in freshwater acidification. *Aquatic Sciences* – *Research Across Boundaries*, 50(1): 71-86.

Shah S. B. and P. Kolar. 2012. Evaluation of additive for reducing gaseous emissions from swine waste. *Agric. Eng. Int.: CIGR Journal*, 14(2): 10-20.

Sheppard, S. C., and S. Bittman. 2011. Farm survey used to guide estimates of nitrogen intake and ammonia emissions for beef cattle, including early season grazing and piosphere effects. *Animal Feed Science and Technology*, 166–167(2011): 688-698.

Sommer, S. G., Christensen, B. T., Nielsen, N. E., Schjorring, J. K., 1993. Ammonia volatilization during storage of cattle and pig slurry—effect of surface cover. *J. Agric. Sci.*, 121(1): 63–71.

Sommer, S.G., Pedersen, S.O., Sogaard, H. T. 2000. Greenhouse gas emissions from stored livestock slurry. *J. Environ. Qual.*, 29(2000): 744–751.

Sommer, S. G., S. O. Petersen, H. B. Møller. 2004. Algorithms for calculating methane and nitrous oxide emissions from manure management. *Nutrient Cycling in Agroecosystems*, 69(2): 143-154.

Sommer, S. G., G. Q. Zhang, A. Bannink, D. Chadwick, T. Misselbrook, R. Harrison, N. J. Hutchings, H. Menzi, G. J. Monteny, J. Q. Ni, O. Oenema, and J. Webb. 2006. Algorithms determining ammonia emission from buildings housing cattle and pigs and from manure stores. *Advances in Agronomy*, 89(2006): 261-335.

Staebler, R. M., S. M. McGinn, B. P. Crenna, T. K. Flesch, K. L. Hayden, S.-M. Li. 2009. Three-dimensional characterization of the ammonia plume from a beef cattle feedlot. *Atmospheric Environment*, 43(38): 6091–6099.

Stevens, R. J., Laughlin, R. J., Frost, J. P. 1989. Effect of acidifying with sulphuric acid on the volatilisation of ammonia from cow and pig slurries. *J. Agric. Sci.*, 113(3): 389–395.

Van Buggenhout, S., Van Brecht, A., Eren O¨ zcan, S., Vranken, E., Van Malcot, W., Berckmans, D. 2009. Influence of sampling positions on accuracy of tracer gas measurements in ventilated spaces. *Biosystems Engineering*, 104(2): 216-223.

Velthof, G. L., C. van Bruggen, C. M. Groenestein, B. J. de Haan, M. W. Hoogeveen, J. F. M. Huijsmans. 2012. A model for inventory of ammonia emissions from agriculture in the Netherlands. *Atmospheric Environment*, 46(2012): 248-255.

Von Bobrutzki, K., Braban, C. F., Famulari, D., Jones, S. K., Blackall, T., Smith, T. E. L., Blom, M., Coe, H., Gallagher, M., Ghalaieny, M., McGillen, M. R., Percival, C. J., Whitehead, J. D., Ellis, R., Murphy, J., Mohacsi, A., Pogany, A., Junninen, H., Rantanen, S., Sutton, M. A., and Nemitz, E. 2010. Field inter-comparison of eleven atmospheric ammonia measurement techniques. *Atmospheric Measurements Techniques*, 3 (2): 91-112.

Von Bobrutzki, K., Muller, H.-J., Scherer, D. 2011. Factors affecting the ammonia content in the air surrounding a broiler farm. *Biosystems Engineering*, 108 (4): 322-333.

Wang, J., L. M. Cardenas, T. H. Misselbrook, S. Gilhespy. 2011. Development and application of a detailed inventory framework for estimating nitrous oxide and methane emissions from agriculture. *Atmospheric Environment*, 45(7): 1454- 1463.

Warneck, P. 1988. Chemistry of the Natural Atmosphere. Academic Press, San Diego, CA.

Webb, J., T. H. Misselbrook. 2004. A mass-flow model of ammonia emissions from UK livestock production. *Atmospheric Environment*, 38(14): 2163–2176.

Webb, J., H. Menzi, B. F. Pain, T. H. Misselbrook, U. Dammgen, H. Hendriks, and H. Dohler. 2005. Managing ammonia emissions from livestock production in Europe. *Environmental Pollution*, 135(3): 399–406.

Wheeler, E. F., M. A. A. Adviento-Borbe, R. C. Brandt, P. A. Topper, D. A. Topper, H. A. Elliott, R. E. Graves, A. N. Hristov, V. A. Ishler, and M. A. V. Bruns. 2011a. Amendments for mitigation of dairy manure ammonia and greenhouse gas emissions: preliminary screening. *Agric Eng Int: CIGR Journal*, 13(2): 1-14.

Wheeler, E. F., P. A. Topper, R. C. Brandt, N. E. Brown, A.

Adviento-Borbe, R. S. Thomas, and G. A. Varga. 2011b. Multiple-Chamber Instrumentation Development for Comparing Gas Fluxes from Biological Materials. *Applied Engineering in Agriculture*, 27(6): 1049-1060.

Williams, A. 2003. Floating covers to reduce ammonia emissions from slurry. In: Proceedings of the International Symposium on Gaseous and Odour Emissions from Animal Production Facilities, Horsens, Denmark, June 1–4, pp. 283–291.

Winiwarter, W., and G. Schimak. 2005. Environmental software systems for emission inventories. *Environmental Modelling & Software*, 20(12): 1469-1477.

Winiwarter W., T. A. J. Kuhlbusch, M. Viana, and R. Hitzenberger. 2009. Quality considerations of European PM emission inventories. *Atmospheric Environment*, 43(25): 3819–3828.

Yamaji, K., T. Ohara, and H. Akimoto. 2003. A country-specific, high-resolution emission inventory for methane from livestock in Asia in 2000. *Atmospheric Environment*, 37(31): 4393–4406.

Zervas, G., and E. Tsiplakou. 2012. An assessment of GHG emissions from small ruminants in comparison with GHG emissions from large ruminants and monogastric livestock. *Atmospheric Environment*, 49(2012): 13-23.

Zhang, R., J. A. McGarvey, Y. Ma, and F. M. Mitloehner. 2008. Effects of anaerobic digestion and aerobic treatment on the reduction of gaseous emissions from dairy manure storages. *International Journal of Agriculture and Biological Engineering*, 1(2): 15－20.

Zhou, J. B., M. M. Jiang, and G. Q. Chen. 2007. Estimation of methane and nitrous oxide emission from livestock and poultry in China during 1949–2003. *Energy Policy*, 35(7): 3759–3767.