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Ammonia and GHG emission from environmental techniques in livestock farms based on modelization

TESIS DOCTORAL

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Summary

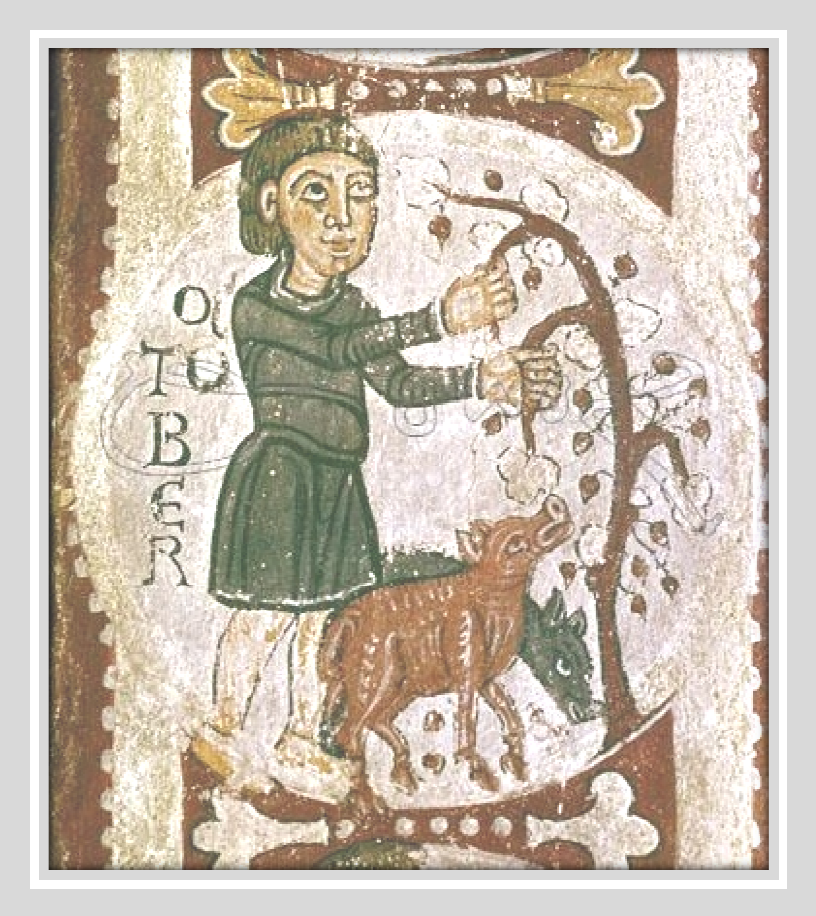
In trying to respond to societal demands for sustainable development, farming systems have to deal with environmental, technical and economic challenges, in which innovative solutions are required. At this regard, poultry and swine farms over certain size must have a permit to operate based on the implementation of Best Available Techniques (BAT) according to Industrial Emissions Directive IED 2010/75/EU. In this thesis we have developed a practical model (BATFARM model) for environmental techniques selection in swine, poultry and dairy cattle farms. The model aims to identify the key stages giving rise to farm gas emissions (ammonia and greenhouse gases, GHG), which would most benefit from implementing environmental techniques and predicts farm emission variation under different scenarios. Designed with methodological rigour and as a user-friendly tool, the model calculates mass balances throughout farm stages in order to estimate manure evolution, related emissions, feed consumptions and animal production. It was set up using a combination of methodologies including emission factors, empirical equations and process-oriented mechanisms. Model testing seems to indicate that the model considers relevant interactions between farm components and captures the effect of factors having an important impact on gas emission. However, results must be interpreted as indicative of the relative emission reduction achieved due to implementation of mitigation practices rather than absolute values. Besides, direct measurements on farm are required to establish emission factors for up-scaling farm level processes in modeling work. In this thesis, the evaluation of the performance of a wet scrubber in a commercial pig farm has been carried out and results have been incorporated in the model. Wet scrubbers seem to be, also in our climatic conditions, very effective for ammonia reduction. However air cleaning may not be a generally applicable technique due to the high implementation cost and it can lose its efficiency if applied without any further measures in downstream farming activities. Precisely, whole-farm models can help in these integrated scenario evaluations and can support in the selection of most effective mitigation option in each particular farm case.

Modelización de emisiones de amoniaco y GEI de técnicas ambientales en explotaciones ganaderas

Resumen

Actualmente, los sistemas agro-ganaderos tienen que hacer frente a desafíos ambientales, técnicos y económicos para tratar de responder a las demandas de desarrollo sostenible de la sociedad, lo cual requiere de soluciones innovadoras. En este sentido, las explotaciones avícolas y porcinas de gran tamaño deben obtener una autorización ambiental basada en la implementación de las Mejores Técnicas Disponibles (MTD) de acuerdo a la Directiva de Emisiones Industriales IED 2010/75 / EU. Esta tesis aborda el desarrollo de un modelo (modelo BATFARM) para la selección de técnicas ambientales en granjas comerciales de ganado porcino, avícola y de vacuno leche. Su objetivo es identificar las etapas clave que contribuyen a la emisión de amoniaco y de gases de efecto invernadero (GEI) en las explotaciones ganaderas, así como la comparación de diferentes escenarios de implementación de MTD. Diseñado con rigor metodológico y como una herramienta fácil de usar, calcula la producción y composición de los estiércoles y purines, las emisiones, los consumos y la producción animal. El modelo combina diferentes metodologías incluyendo factores de emisión y ecuaciones, tanto empíricas como basadas en procesos. Las pruebas realizadas parecen indicar que tanto las interacciones como el efecto de los factores con un impacto relevante en las emisiones, son considerados por el modelo. Sin embargo, los resultados deben ser interpretados como indicativos de la reducción relativa lograda y no como valores absolutos de emisión. La disponibilidad de estudios de investigación aplicada en granjas para determinar factores de emisión es fundamental en la modelización. En el marco de esta tesis se ha llevado a cabo la evaluación de un lavador de aire húmedo en una explotación comercial de ganado porcino, cuyos resultados se han incorporado en el modelo. Los lavadores húmedos parecen ser, también en nuestras condiciones climáticas, muy eficaces en la reducción de amoniaco. Sin embargo, es una técnica costosa que además puede perder su eficacia si no se aplican medidas adicionales en las subsiguientes etapas. Precisamente, los modelos permiten realizar este tipo de evaluaciones a lo largo de todas las fases de producción ganadera, clave para la selección de las técnicas que permitan alcanzar los objetivos de reducción de emisión requeridos en cada caso de la forma más eficaz.

A mis padres



Colegiata de San Isidoro (León), Panteón Real. Pinturas románicas.
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Table of contents

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Introduction, Objectives & Structure

1.1. Introduction

Representing in 2015 the 41% of the Total Agricultural Goods Output of the European Union (UE-28) (Eurostat, 2017), livestock production contributes substantially to the economies of many European countries in terms of employment, export of products and use and maintenance of natural resources. Much of this production is from intensified schemes, as it is the case for most pig, poultry and some cattle productions.

Over the last decades, livestock practices have evolved considerably and traditional farms have been replaced in many cases by intensive systems characterized by large numbers of animals using a relatively small area of land with a high demand of inputs from out of the farm (feeds, energy, water, veterinary services). Pig and poultry farms became the most specialized and intensive as they could rely almost entirely on imported feedstuffs (Bernet & Béline, 2009; Burton & Turner, 2003). This trend towards more specialized and intensive production has had a remarkable impact on environment, mainly due to emissions of pollutants to air (odours, gases, dust, and bioaerosols), discharges to soils and surface waters (nitrogen, phosphorus, heavy metals and drug residues) and raw materials consumption (Burton & Turner, 2003; Hartung, 2007; Herrero et al., 2015; Lassaletta et al., 2016).

According to Steinfeld et al. (2006) livestock is responsible for the 37% of methane (CH_4), 65% of nitrous oxide (N_2O) and 64% of ammonia (NH_3) anthropogenic emissions.

About 30% of the greenhouse gases (GHG) produced by livestock production are attributed to manure management (Bernet & Béline, 2009). Nitrous oxide is mainly emitted from soils following organic and mineral fertilization as an intermediate product of nitrification/denitrification under conditions of low oxygen availability and degradation of organic matter. Methane is emitted as a product of enteric fermentation in ruminant animals, and from decomposition of manure under anaerobic conditions, especially when manure is stored in liquid form.

According to the European Environment Agency and EMEP (European Monitoring and Evaluation Programme), the agricultural sector remains the major source of NH_3 emissions (94% of total EU28 emissions in 2014) which derive mainly

from the livestock sector, particularly from the decomposition of urea in animal wastes (EEA, 2016).

In response to the growing concerns about the environmental impacts from modern farming, society takes action in the form of policy interventions which can involve a mix of regulations, voluntary agreements and economic instruments (Pellini & Morris, 2001).

Directive 2016/2284/EU on the reduction of national emissions sets national reduction commitments for five pollutants (sulphur dioxide, nitrogen oxides, volatile organic compounds, ammonia and fine particulate matter). This new Directive repeals and replaces Directive 2001/81/EC, the National Emission Ceilings Directive (NEC Directive) from the date of its transposition (30 June 2018) ensuring that the emission ceilings for 2010 set in that Directive shall apply until 2020 and establish more ambitious reduction commitments for 2030. Projections for Spanish emissions seem to indicate that emission ceilings for 2030 will be exceeded for all the pollutants if no further abatement measures are implemented (Orio, 2014).

The Integrated Pollution Prevention and Control Directive IPPC (96/61/CE) which has been incorporated into the Industrial Emissions Directive IED 2010/75/EU, combines a mix of compulsory and voluntary technical standards that regulates all forms of emissions into the atmosphere, water and soil. It also considers waste production, water consumption and energy efficiency for different industrial activities to achieve a high level of protection of the environment as a whole. One of the defined sectors under IPPC (IED) in the UE is intensive livestock farms. Farms with more than 40000 poultry, 2000 fattening pig or 750 sow places are considered in regulation. The transposition of the Industrial Emissions Directive IED 2010/75/EU to the Spanish legislation has been carried out in such a way that the Spanish Government acquires a subsidiary responsibility in the application of the Directive, allowing to the different regions modulate the measures, in some cases even including more restrictive laws than the ones found in the European Directive. In Navarre region (Spain), the type of livestock affected by the Directive has been extended to dairy cattle farms over 250 adult cows (Provincial Law 4/2005, of the 22 March, of the Intervention for the Protection of the Environment, BON No. 39 of 1st April 2005). All the facilities under IPPC (IED) must have a permit to operate based on the implementation of Best Available Techniques (BAT) in the whole farm

production process, the Integrated Environmental Authorization. During the period from 2009 to 2011, a total of 19,141 farms required a permit in EU27, from which 2,615 (13.6%) were located in Spain (EC, 2015).

The BAT are defined as the most effective techniques to accomplish high general level of environmental protection under economically and technically viable conditions in a particular farm situation. These BAT are described in the "Reference Document on Best Available Techniques for The Intensive Rearing of Poultry and Pigs" (EC, 2003), so called BREF, a review document providing information on the main techniques that are considered by the Technical Working Group formed by European experts in this field. This document is periodically revised as new techniques emerge and BAT studies are carried out (EC, 2015). Strategies contemplated involve whole farm systems including nutritional techniques, housing design to reduce emission, techniques to reduce water and energy consumptions, to improve manure handling during storage, reduced emission spreading methods and treatment techniques.

Farms affected by the IPPC (IED) have to present a proposal of BAT implementation to the Regional Administration to apply for the Integrated Environmental Authorization considering: the available techniques, the farm impact on environment and the affordable cost. Then the Administration evaluates the proposal and if it is not enough, the farmers will have to negotiate with the regulators the required improvements.

At this regard, first "BAT conclusions" decision has recently been published (Commission Implementing Decision (EU) 2017/302 of 15 February 2017). "BAT conclusions" are the technical basis for national authorities in EU countries to set permit conditions for large poultry or pig farms process and will be published for each BREF reviewed under the IED. The "BAT conclusions" set, for the first time at the EU level, limits for ammonia emissions to air from animal housing and for excretions of nitrogen and phosphorus. Also, other environmental issues such as dust, odour and noise emissions are part of the new "BAT conclusions". Within four years, the authorities must ensure that the permit conditions for the farms concerned are reconsidered and, if necessary, updated according to these new standards.

Moreover, farms affected by the IPPC (IED) have to yearly declare their emissions to the European Pollutant Release and Transfer Register (E-PRTR). In 2014 there were a total of 6176 intensive livestock facilities in EU27 with public information in E-PRTR, 30% of them located in Spain and representing the 28% of total EU27 NH_3 emissions in this register.

The level of BAT implementation in livestock farms is quite variable among different regions. One possible explanation for this divergence is that for many farmers the adoption of the BAT requires an extra effort of innovation and adaptation. Despite huge research carried out during these years studying BAT, research results are often insufficiently exploited and taken up in practice, and innovative ideas from practice are not captured and spread.

The evaluation of the convenience to select a mitigation technique instead of another is not an easy task. Some of these techniques may be erroneously selected leading to pollution swapping (situation when one technique can successfully reduce diffusion of pollutants but it can also have the potential to increase levels of one or more non target pollutants) and others are incorrectly ignored because they present high cost, confusion on their benefits and lack of demonstration on commercially viable farms. Main reasons for the lack of information of BAT effects in commercial farms might be: the availability of farm installations to research, the higher risk of uncontrolled conditions under commercial farm operation and the huge amount of resources (staff and equipments) required for this type of measurements. Therefore, most of abatement techniques have been studied under controlled laboratory or pilot-scale conditions and in North and Center of Europe, being less frequent in Mediterranean countries. These aspects together with a lower pressure for environmental protection might be some of the reasons of lower level of innovative BAT implementation in Southern Europe. This is for example the case of wet scrubbers, a technology widely used in Netherlands, Northern Germany or Denmark, but infrequent to find in highly intensive producing areas in Southern EU countries. As a consequence, very limited information is available on the performance and costs of many innovative technologies in warmer conditions. Understanding these causes of lacking information for farmers and enhancing the implementation of BAT among countries and livestock sectors seems essential to ensure the application of the IED.

Consideration of the effect of the proposed techniques in the whole farm system will be essential for selection of a valid strategy. Decision support tools to explore and simulate the effect of farm mitigation options can be very useful for environmental authorities, farmers and technicians in this task. Whole farm models are also necessary to calculate absolute farm emission to be submitted to E-PRTR, to monitor new environmental standards under BAT conclusions and for national emission inventories and projections.

Different models have been developed in Europe (Schils, Olesen, del Prado, Soussana, 2007), and numerous studies describe models used for national agricultural NH_3 emission inventories (Dämmgen, Lüttich, Döhler, Eurich-Menden, Osterburg, 2003; Hutchings, Sommer, Andersen, Asman, 2001; Menzi, Rüttimann, Reidy, 2003; Reidy et al., 2008; Reidy, Rhim, Menzi, 2008; Reidy et al., 2009; Velthof et al., 2012). Some of these models identify the most cost-effective means of reducing farm pollution (Gooday et al., 2014; Webb et al., 2006).

However, there is a lack of support and knowledge transfer models designed for farm-scale operations that combine a variety of methodologies and uses inputs well known by the farmers rather than requiring additional data collection. Besides, as the availability and the quality of data will constrain model parameterization for specific farms conditions and the quality of results, there is a need of models designed to enable continuously updating of its database as new knowledge becomes available.

In summary, it is essential to review critically the implementation of BAT in the intensive livestock production of the EU and establish effective ways for the farmers to make use of this information. Developing practical models to be used at farm level may be of highest interest. To this aim, we require results based on applied research conducted under real conditions. According to this, the work conducted under this PhD thesis follows this approach.

1.2. Research objectives

The main thesis target is to contribute to BAT implementation in particular situations of commercial farm systems, using consistent procedures to evaluate the techniques for a practical BAT selection.

To this aim several particular objectives have been established:

1. To analyse the main challenges for the implementation of emission abatement technologies in European livestock farms.

2. To develop a whole farm model to be used by policy makers, environmental authorities, farmers and technicians to assess the emissions of N_2O, CH_4 and NH_3 resulting from different strategies and techniques implemented on intensive cattle, pig and poultry farms.

3. To evaluate innovative techniques to reduce gas emission in commercial farms following science-based protocols and integrating them in the whole farm model. In particular, scrubbing systems to clean the air in a gestating sows building in Northern Spain has been monitored.

The thesis has contributed to facilitate a collaborative network with main stakeholders (farmer, advisors and researchers) that have actively participated during the PhD development.

1.3. Thesis structure

According to these objectives, the PhD thesis is structured in 6 chapters that are briefly described below and in Figure 1.

Chapter 1 (this chapter) includes a brief introduction to the topic and establishes the objectives and structure of the PhD thesis.

Chapter 2 analyses current challenges in BAT implementation in European livestock farms (Thesis Objective 1), by providing a summary of the main techniques for reducing the impact on environment from livestock farming, a critical evaluation of the different ways to assess the performance of a given technique and the main conclusions of experts present with respect to future research on the development of effective environmental protection.

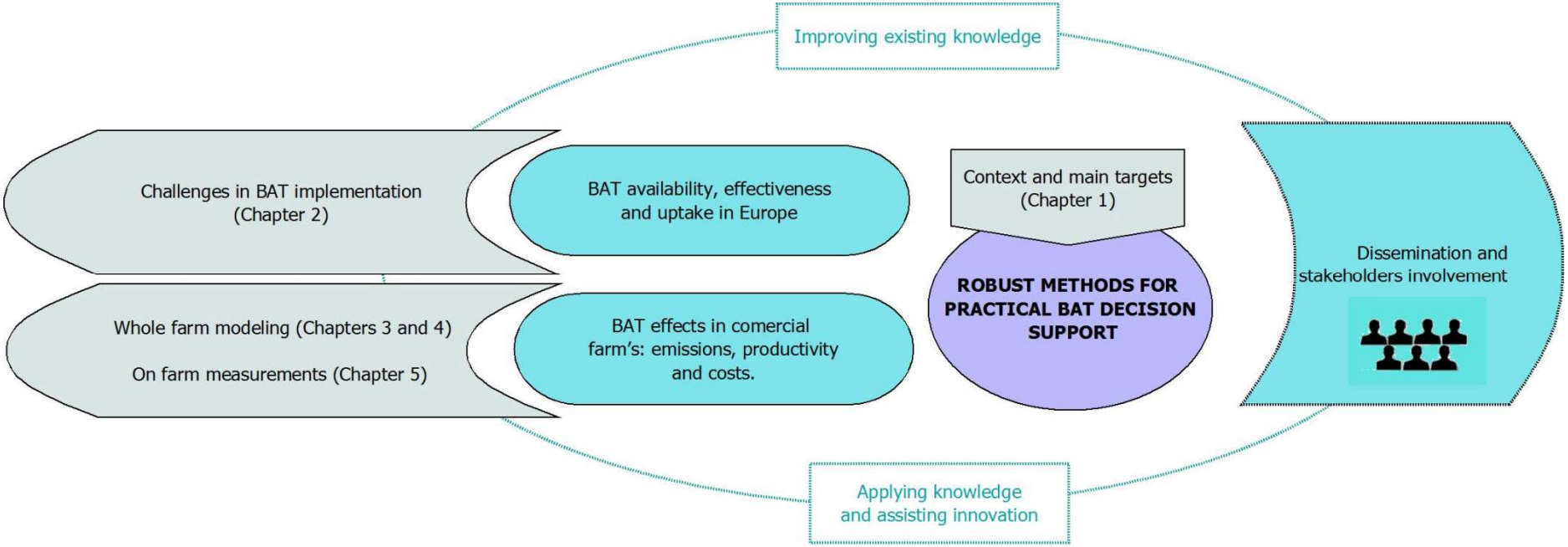
Next two chapters are focused on the model (Thesis Objective 2). Chapter 3 describes the model itself, showing model methods and equations of the different subroutines. Chapter 4 presents model application in swine farms studying how

simulated gas emissions response to systematic variation of management parameters; conducting N and C mass balance; comparing model results against literature data; simulating scenarios to illustrate potential applications and finally showing model evaluation by main stakeholders.

Chapter 5 presents the results from the monitorization of two air scrubbers in a commercial pig farm in Northern Spain under different washing water management to assess the potential use and limitations of this technique and feed the whole farm model (Thesis Objective 3).

The last chapter of the thesis (Chapter 6) includes a general discussion and a summary of main conclusions of this thesis.

Fig. 1. Thesis structure. Links among the different chapters and thesis objectives.



1.4 References

Bernet N. and Béline F. (2009). Challenges and innovations on biological treatment of livestock effluents. Bioresource Technology 100, 5431-5436.

Burton C.H. and Turner C. (2003). Manure management — treatment strategies for sustainable agriculture. 2nd Edition. Silsoe Research Institute, Wrest Park, Silsoe, Bedford (UK), 490 pp.

Dämmgen U., Lüttich M., Döhler H., Eurich-Menden B. and Osterburg B. (2003). GAS-EM – a procedure to calculate gaseous emissions from agriculture. Landbauforschung Völkenrode 52, 19-42.

EC (2003). Reference Document on Best Available Techniques for intensive Rearing of Poultry and pigs. Integrated Pollution Prevention and Control (IPPC), July 2003, 341 pp.

EC (2015). Best Available Techniques (BAT) Reference Document for the Intensive Rearing of Poultry or Pigs. Final Draft - August 2015, European Commision, 911 pp.

EEA (2016). European Union emission inventory report 1990–2014 under the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP). Technical report No 16/2016, 122 pp.

Eurostat (2017). http://epp.eurostat.ec.europa.eu

Gooday R.D., Anthony S.G., Chadwick D.R., Newell-Price P., Harris D., Duethmann D., Fish R., Collins A.L. and Winter M. (2014). Modelling the cost-effectiveness of mitigation methods for multiple pollutants at farm scale. Science of the Total Environment 468–469, 1198–1209.

Hartung, J. (2007). Assessment of environmental effects of airbone emissions and waste effluents from livestock production. Proceedings of the XIII International Society of Animal Hygiene Congress, Tartu (Estonia), 695-701.

Herrero, M., Wirsenius, S., Henderson, B., Rigolot, C., Thornton, P., Havlík, P., De Boer, I. and Gerber, P.J. (2015). Livestock and the environment: what have we learned in the past decade?. Annual Review of Environment and Resources 40, 177-202.

Hutchings N.J., Sommer S.G., Andersen J.M. and Asman W.A.H., (2001). A detailed ammonia emission inventory for Denmark. Atmospheric Environment 35, 1959-1968

Lassaletta, L., Billen, G., Garnier, J., Bouwman, L., Velazquez, E., Mueller, N.D. and Gerber, J.S. (2016). Nitrogen use in the global food system: past trends and future trajectories of agronomic performance, pollution, trade, and dietary demand. Environmental Research Letters 11, 095007.

Menzi H., Rüttimann L. and Reidy B. (2003). DYNAMO: a new calculation model for dynamic emission inventories for ammonia. International Symposium on Gaseous and Odour Emission from Animal Production Facilities, Horsens (Denmark), 378-381.

Orío, A. (2014). Nueva Directiva de Techos Nacionales de Emisión de Contaminantes a la Atmósfera. Subdirección General de Calidad del Aire y Medio Ambiente Industrial. Jornada Ganadería y Medioambiente, Ministerio de Agricultura, Alimentación y Medio Ambiente, Madrid, 27 de noviembre de 2014.

Pellini T. and Morris J. (2001). A framework for assessing the impact of the IPPC directive on the performance of the pig industry. Journal of Environmental Management 63, 325–333.

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

Reidy B., Rhim B. and Menzi H. (2008). 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, 3266-3276.

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

Schils R.L.M., Olesen J.E., del Prado A. and Soussana J.F. (2007). A review of farm level modelling approaches for mitigating greenhouse gas emissions from ruminant livestock systems. Livestock Science 112, 240-251.

Steinfeld H.,Gerber P., Wassenar T., Castel V., Rosales M. and Haan Cd. (2006). Livestock's long shadow. Environmental Issues and Options. Food and Agricultural Organisation, Rome, 390 pp.

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

Webb J., Ryan M., Anthony S.G., Brewer A., Laws J., Aller M.F. and Misselbrook T.H. (2006). Cost-effective means of reducing ammonia emissions from UK agriculture using the NARSES model. Atmospheric Environment 40, 7222–7233.

Best available technology for European livestock farms:

Availability, Effectiveness and Uptake

Published as:

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Best available technology for European livestock farms: Availability, Effectiveness and Uptake

Abstract

Concerns over the negative environmental impact from livestock farming across Europe continue to make their mark resulting in new legislation and large research programmes. However, despite a huge amount of published material and many available techniques, doubts over the success of national and European initiatives remain. Uptake of the more cost-effective and environmentally-friendly farming methods (such as dietary control, building design and good manure management) is already widespread but unlikely to be enough in itself to ensure that current environmental targets are fully met. Some of the abatement options available for intensive pig and poultry farming are brought together under the European IPPC/IED directive where they are listed as Best Available Techniques (BAT). This list is far from complete and other methods including many treatment options are currently excluded. However, the efficacies of many of the current BAT-listed options are modest, difficult to regulate and in some cases they may even be counterproductive with respect to other objectives ie pollution swapping. Evaluation of the existing and new BAT technologies is a key to a successful abatement of pollution from the sector and this in turn relies heavily on good measurement strategies. Consideration of the global effect of proposed techniques in the context of the whole farm will be essential for the development of a valid strategy.

Keywords

Livestock Farming, Manure, Emissions, IPPC Directive, Measurements, BAT Evaluation.

2.1. Introduction

2.1.1. The environmental impacts of livestock farming

Concern over the negative impacts from livestock farming across Europe is certainly not new. Many studies have been carried out on the assessment of the detrimental effects of modern farming systems and the possible abatement methods that might be implemented. These issues relate to air and water quality, and the consequential impacts on ecosystems and biodiversity (Gerber et al., 2013; Hoffmann, 2011) and also to the potential impact on human health (Gilchrist et al., 2007; Seedorf & Hartung, 1999). In particular, nitrogen (N) losses from the livestock farming process is present in many forms of pollution including nitrate (NO_3^-) leaching contributing to eutrophication (Jarvie, Neal, Withers, Wescott, Acornley, 2005; Moreau et al., 2013), and ammonia (NH_3) emissions from livestock manures (Fangmeier, Hadwiger-Fangmeier, Van der Eerden, Jäger, 1994) with recognized detrimental effects on soil condition, forests and biodiversity (Steinfeld et al., 2006). Furthermore, the presence of surplus nitrate in certain soils can lead to the production and emission of nitrous oxide (N_2O) (Fangmeier et al., 1994; Sommer, Østergård, Løfstrøm, Andersen, Jensen, 2009). Livestock production activities also contribute to greenhouse gas emission (GHG), especially methane (CH_4) from enteric fermentation, and both CH_4 and N_2O from manure management (Chadwick et al., 2011; Steinfeld et al., 2006). The accumulation of copper and zinc in soils may impose a toxicity risk to plants and micro-organisms (Dourmad & Jondreville, 2007). Livestock production accounts for an estimated 14% and 64% of world GHG and NH_3 emissions, respectively (Gerber et al., 2013, Steinfield et al., 2006) and 78% of NH_3 emissions in Europe (EEA, 2010).

2.1.2. Abatement of pollution by the development of regulatory measures

The environmental impacts from modern farming, has led to a series of international protocols, European directives and national regulations. Control of NH_3 emissions comes under the EU National Emissions Ceilings Directive (EC, 2001) resulting from the Gothenburg Protocol (United Nations Convention on Long-range Transboundary Air Pollution - CLRTP, UNECE, 1999). Emissions of CH_4 and N_2O from livestock farming are regulated by the Kyoto Protocol under the United Nations Framework Convention on

Climate Change - UNFCCC (UN, 1997). Nitrate and phosphorus loading of water resources are addressed by the EU Nitrates Directive (ECC, 1991) and the EU Water Framework Directive (EC, 2000). The Nitrate Directive sets limits for the period of time of permitted manure application and a standard of 50 mg L^{-1} for NO_3^- concentrations in surface and ground waters, whilst the Water Framework Directive sets phosphorus concentration standards of 50 to 120 µg L^{-1} for good ecological status. Under this legislative framework, European member states have implemented national programmes to achieve their obligations to reduce NO_3^- losses (to water) and NH_3 and GHG emissions (to air). These measures are based on official documents that specify the current scientific knowledge and best techniques to reduce pollution: for NO_3^- "Good Agricultural Practice for nitrates" (EEC, 1991), and, for NH_3 the "Guidance document for preventing and abating NH_3 emissions from agricultural sources" (UNECE, 2014).

An even more prescriptive approach to implement abatement measures has emerged from the Integrated Pollution Prevention and Control Directive IPPC 96/61/EC (EC, 1996), which has been incorporated in to the Industrial Emissions Directive 2010/75/EU (EC, 2010). This directive sets common rules for licensing industrial activities with the broad objective of environmental protection. One of the defined sectors is intensive livestock farms (currently those with more than 40000 places for poultry or 2000 places for fattening pigs) which must have an operating permit that describes the whole environmental performance of the farm. This takes in to account pollution of air, water and land, waste production and resource utilization (including water consumption and energy efficiency). The operating permit is only given if the farmer demonstrates the appropriate use of "best available technologies not entailing excessive costs" (BAT) which are listed and described in the official "Reference Document on Best Available Techniques for The Intensive Rearing of Poultry and Pigs" or BREF (EC, 2003).

2.1.3. Challenges to the implementation of effective abatement strategies

Despite the extensive concerted effort by governments and researchers over many years, success in protecting the environment is still questioned on the basis of (a) the efficacy of the individual measures proposed (with respect to the level of abatement achieved and cost), (b) the suitability of some methods for certain farm types, (c) the actual level of uptake of the related technology and (d) the conflict and incompatibility of methods for differing objectives. This last point highlights the risk of counter-productivity

in applied measures: for example, prohibiting manure application in the winter (such as specified by the EU Nitrates Directive) require longer manure storage with the increased risk of CH_4 emissions. Furthermore, there is the additional pressure of using manure as an organic fertilizer but without compromising food quality or safety (Burton, 2009). Thus there may be some justification in the hesitation to implement methods when there is no standard tool or prescriptive document on the precise selection and application of the appropriate BAT method.

2.1.4. A technical workshop to review the application of BAT technologies

In this context the BATFARM project (Interreg-Atlantic Area, Project, 2009-1/071) was launched in 2009 with the objective of evaluating proposed methods against local and environmental criteria. This project reached its final stages late in 2013. Amongst its many outputs was the organization of a European workshop "Reconciling livestock management to the environment - applying Best Available Technique (BAT): from the lab to the farm" which took place in IRSTEA, Rennes (France) in March 2013 (Loyon & Burton, 2013). The workshop brought together 40 leading scientific researchers and IPPC/IED inspectors for the purpose of critically reviewing the effectiveness of abatement techniques (BAT listed and others) in the context of livestock housing and manure management with the objective of achieving measurable environmental protection.

This review paper is based on the submitted material, presentations and related discussions of this workshop. It provides (i) a summary of the main techniques for reducing the impact on environment from livestock farming, (ii) a critical evaluation of the different ways to assess the performance of a given technique and (iii) the main conclusions of experts present with respect to future research on the development of effective environmental protection.

2.2. Reducing emission by animal diet modification

2.2.1. General principles

Manipulation of the animal diet can be a practical way to limit the impact of livestock on the environment by controlling (i) the amount and composition of manure produced and the associated gaseous emissions from manure and (ii) the enteric CH_4 production.

Diet manipulation is listed as BAT in the BREF document for pig and poultry farms (EC, 2003), as one of the techniques given in the UNECE Guidance Document for reducing ammonia from agriculture (UNECE, 2014) and also in the recent FAO review (Hristov et al., 2013) of techniques to reduce non-CO_2 GHG emission from livestock production.

2.2.2. Diet manipulation for reducing excretion of N and P and ammonia emissions.

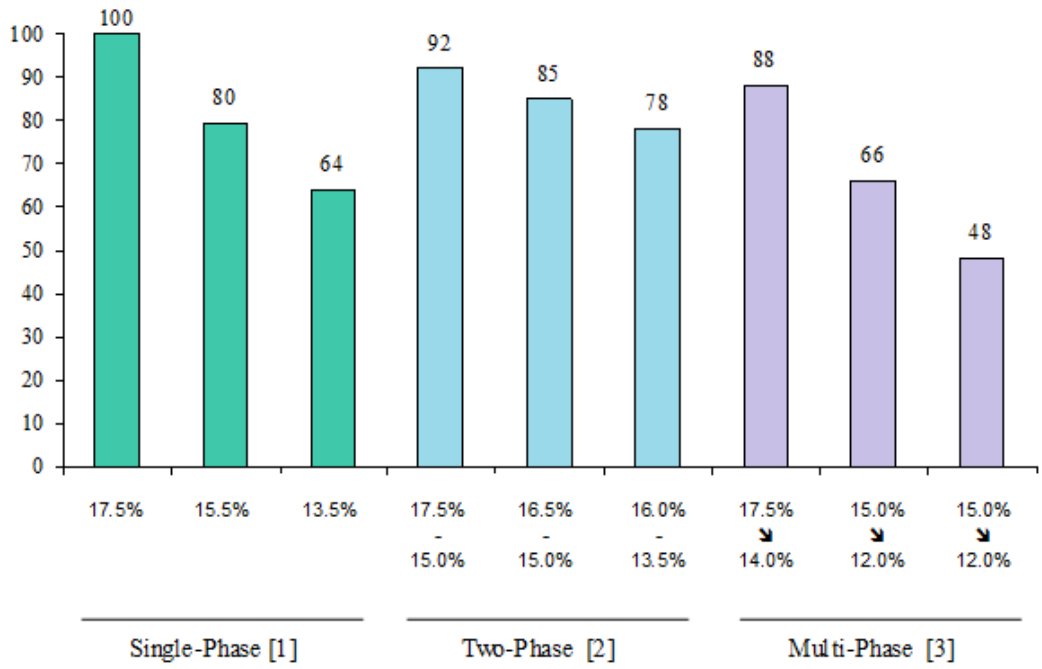
Modifications to the animal diet can affect the level of nitrogen, phosphorous and trace element excretion (Dourmad & Jondreville, 2007; Hristov et al., 2013) without penalizing animal health, welfare or performance (Dourmad & Jondreville, 2007; Nahm, 2007; Veldkamp, Star, van der Klis, van Harn, 2012). In pigs, a low crude protein (CP) diet supplemented with amino acids can reduce N excretion by 25-50% (Figure 1) and lead to a lower pH and thus a reduction in subsequent NH_3 emission (Dourmad & Jondreville, 2007). It has been frequently suggested that lower CP diet can also reduce N_2O emissions since NH_3 is a precursor of its formation but laboratory-scale experiments failed to support this hypothesis (Clark, Moehn, Edeogu, Price, Leonard, 2005; Le, Aarnink, Jongbloed, 2009; Osada, Takada, Shinzato, 2011). Under farm conditions with fattening pigs on litter, Philippe, Laitat, Canart, Vandenheede and Nicks (2007) reported that whilst NH_3 emissions fell by 26% when the crude protein was reduced by 18% the subsequent N_2O emissions doubled.

Reducing the crude protein content in poultry feed is also possible (Veldkamp et al., 2012). It is important that the optimal amino acid pattern for each poultry species is ensured by adding supplements. Ammonia reductions up to 50% have been reported when the crude protein content of the diets were decreased in this way. However, such changes will increase the overall feed costs because of the relatively high price of the amino acid supplements. For example, the decrease of the crude protein content in laying hen diets by 30 g/kg at constant digestible lysine content will increase feed costs by around 16%. A decrease of 5, 10 and 15 g/kg crude protein content in broiler diets suggest an increase in feed costs of 5, 12 and 19%, respectively.

In pig production, the inclusion of dietary fibre reduces NH_3 emission by shifting the nitrogen from urine to feces due to promotion of bacterial growth in the large intestine (Jarret, Cerisuelo, Peu, Martinez, Dourmad, 2012; Philippe, Cabaraux, Nicks, 2011).

However, dietary fibre inclusion also increases enteric CH_4 production depending on the specific fibre added (Philippe et al., 2008). A significant reduction in NH_3 emission can also be achieved by lowering the dietary electrolyte balance or supplementing with acidifying salts such as benzoic acid or $CaSO_4$ (Philippe et al., 2011).

Fig. 1. Effect of dietary crude protein (CP) content and protein feeding strategy on N excretion of fattening pigs (100% = excretion with single-phase feeding of a 17.5% CP diet) (Dourmad and Jondreville, 2007). Single-phase: the same diet is fed over the whole fattening period (30-110 kg body weight); Two-Phase: two different diets are fed during the growing (30-65 kg LW) and the finishing (65-110 kg BW) periods. Multiphase: the composition of the diet is changed each day/week by mixing two diets in different proportions to fit evolution of animals'protein requirements.



For dairy cattle, decreasing N excretion can be achieved by an improvement of rumen protein metabolism leading to lower NH_3 and N_2O production. Two methods are recommended: firstly, decreasing crude protein content in the diet to 14% (from 17-18%) which is often practiced, so that less N is excreted whilst both maintaining milk production, and secondly, limiting the digestion of ingested food meal in the rumen. These measures can cut NH_3 emissions from excreted manure by up to 70% (Ndegwa, Hristov, Arogo, Sheffield, 2008; Pellerin et al, 2013). A reduction of dietary protein content has not considered for beef production systems because the protein content in their feed is rarely excessive.

Another abatement technique based on diet control is to improve the biological availability of certain nutrients such phosphorus by phytase addition to the feed (Dourmad & Jondreville, 2007). Other feed additives, such as the mineral zeolite, plant extracts rich in tannins and saponins, and probiotics, have all been tested with varying success to reduce NH_3 losses from both pig manure (Philippe et al., 2011) and cattle manure (Eckard, Grainger, Klein, 2010; Ndegwa et al., 2008).

In order to comply with the IED/IPPC directive and the Gothenburg protocol, the main feeding strategies proposed to reduce N excretion and related NH_3 emissions are listed as, (i), phase feeding, (ii) low-protein feeding, (iii) increase in the non-starch polysaccharides of the feed and (iv), supplementation with additives to reduce manure pH. The UNECE draft guidance document (2014) specifies reducing the protein content in the diet (by matching more closely to the animal needs) as one of the most cost-effective ways of cutting NH_3 emissions which may also reduce the subsequent emission of N_2O. For each percent point decrease in protein content of the animal feed, total NH_3 emission (from animal housing, manure storage and the application to land) is cut by 5 to 15% due to the reduced ammoniacal nitrogen in the manure produced (UNECE, 2014). A recent European survey on the current abatement strategies in pig and poultry production revealed the most common methods in use as phase feeding and low-protein animal diets. (Loyon, Burton, Guiziou, 2009).

2.2.3. Diet strategies for reducing enteric methane emission by cattle.

Various different feed additives and biotechnologies have been tested for reducing CH_4 production from the rumen (Hristov et al., 2013; Martin, Morgavi, Doreau, 2010). High-concentrate diets and lipid supplementation are considered the most effective means for lowering such emissions (Martin et al., 2010) but these diets also have some negative side effects (Doreau, Makkar, Lecomte, 2013). Lipid supplementation of diets fed to lactating dairy cows reduced production of CH_4 by 4% for each 10 g/kg of added fat in the diet (Martin et al., 2010). Amongst the available fat sources, linseed is one of the most efficient for cutting the emission (Martin et al., 2010) with the potential to also increase the omega-3 fatty acid content of milk and meat produced (Doreau, Martin, Eugène, Popova, Morgavi, 2011). Other sources of unsaturated lipids such as rapeseed may also be used. Amongst the list of feed additives, only nitrate supplement has been proven to be efficient

over a wide range of diets (Doreau et al., 2011, Hristov et al., 2013) but its use may be limited by risks to the animal (in the case of overdose) and by the poor image of nitrate.

2.3. Abatement methods relating to livestock housing

2.3.1. Principles of good housing design

Abatement techniques for livestock housing are largely based on limiting the factors giving rise to the emission, most often NH_3 and to a lesser extent, CH_4 and N_2O. Emissions mostly arise from the manure in the building and (in the cast of cattle) also from enteric fermentation. Ammonia comes principally from the urine (or uric acid for poultry) which contains the majority of the volatile N excreted, whilst the solid manure fraction is more likely to be the source for CH_4 production, and to some extent, N_2O (Chadwick et al., 2011; Sommer et al., 2006). The generation of these three gases is influenced by the floor type, the ventilation system, the building temperature and the manure characteristics (Philippe et al., 2007; Chadwick et al., 2011). The reduction of emission from housing is largely pursued by good management practices (such as the frequent removal, and/or drying of manure), by maintaining good conditions in the building (adequate ventilation and temperature) and the use of end-of-pipe techniques such as air-scrubbing. In some cases, floor type may also have an influence by reducing the emitting surfaces area. Air-scrubbing systems to reduce NH_3 emissions from pig and especially poultry housing are described in the BREF reference document (EC, 2003) but they are rarely applicable to cattle housing because these are usually naturally ventilated in contrast to pig and poultry systems that often closed with forced ventilation.

Decreasing the duration of cattle in housing is a possible technique to reduce NH_3, CH_4 and N_2O emissions, implying an increase in the grazing period by up to 20 days per year (depending on the region). This can lead to overall lower emissions due to (a), the aerobic conditions in the pasture which reduces CH_4 emissions, (b) the separation of urine and faces excreted which may reduce N_2O emissions and (c) less fuel consumption for manure spreading (Pellerin et al., 2013).

2.3.2. Specific techniques applying to cattle housing

Livestock housing can be broadly categorized as either slurry-based or solid manure-based systems (Bioteau, Burton, Guiziou, Martinez, 2010; Burton & Turner, 2003). In some regions, housing may also be associated with outdoor yard areas used for cattle exercise or feeding (Misselbrook, Webb, Chadwick, Ellis, Pain, 2001). In the UNECE Guidance Document for abating agricultural ammonia emissions (UNECE, 2014) the listed techniques for reducing NH_3 from a reference farm system (defined as a cubicle house or tied system) are, (i) use of grooved floors, (ii) optimal barn air conditioning with roof insulation, (iii) increasing the grazing period and (iv), acid or water air scrubbers in those houses with forced ventilation. The implied reduction in emissions range from 10% (for increased grazing) to 95% where air scrubbers are used. Other cited techniques including increasing the frequency of manure removal, additional (targeted) bedding for cattle housing and the frequent washing down of dairy cow collecting yards. Scraping systems combined with improved floor design, such as grooves or sloping floors to facilitate rapid urine removal to slurry storage, can be effective with up to 40% reduction in emission (Swiestra, Braam, Smits, 2001) but the effect is variable because some excreta often will remain on the surface. Larger reductions in emission (up to 90%) can be achieved through frequent washing down of dirtied surfaces with water but this increases water consumption and the volume of manure produced (Misselbrook, Webb, Gilhespy, 2006).

For housing systems using bedding, litter management and in particular, keeping the litter surface as dry as possible is crucial. Use of additional straw bedding and in particular by targeting the additional bedding to the dirtier areas such as around feed barriers or water troughs, can enable reductions in emission by up to 50% (Gilhespy et al., 2009). However, this will both increase operation costs and the volume of solid manure to be managed when removed from the house. Nor should it be overlooked that ammonia emissions may increase where excessive bedding leads to composting activity and subsequent warming. It might be expected that different bedding materials lead to different emission rates (Misselbrook & Powell, 2005). Nonetheless, reported emission factors from national inventories don't reveal a great deal of difference between the systems - indeed, the variation between countries in northern Europe appears to be greater than the differences among the systems themselves (Table 1).

2.3.3. Specific techniques applying to poultry housing

The main building system used in Europe for fattening birds is a solid floor with litter, whilst the common system for layers is still mostly based on various cage designs but with increasing numbers of alternatives including percheries, floor housing and free range (Loyon et al., 2009; UNECE, 2014).

Table 1. Examples of ammonia emission factors (for cattle housing) for a selection of European countries given as a percentage of total ammoniacal nitrogen available (Reidy et al., 2008 & 2009).

Country	Dairy cow cubicle house % of TAN	Beef cattle deep litter house % of TAN
Denmark	17	12
Germany	20	20
Netherlands	15	17
UK	31	23
Switzerland	17	37

For both types of poultry production, the main strategies or techniques recommended for the reduction of emissions is similar to that for cattle: (a) good manure management (especially with respect to the collection and removal of the droppings) and (b) the control or treatment of the building air (Ndegwa et al., 2008; UNECE, 2014). Drying the manure of laying hens by ventilating the deep-pit or channel systems can reduce NH_3 emission by up to 30% whilst an even greater reduction up to 80% can be achieved by the use of belts conveyors to remove the manure to covered stores (UNECE, 2014). The higher reduction corresponds to the case when manure has been dried to 60-70% on the belts through forced ventilation.

The other main technique to reduce NH_3 emission from poultry housing is to treat exhaust air either by trickling bio-filters (reduction up to 70%) or with acid scrubbers (reductions between 70 and 90%) (Melse, Ploegaert, Ogink, 2012; UNECE, 2014). Such air treatment can also remove fine dust and odour. For poultry fattening houses (broiler, turkey, duck) the main techniques listed in the UNECE Guidance are (i) the reduction of water losses from the drinking system by using a nipples instead of bell drinkers, (ii) the treatment of the exhaust air and (iii) using forced manure drying by internal air enabling a reduction in NH_3 of 40-60%. The use of certain additives (such as aluminium sulphate and

those based on certain micro-organisms) is also listed the UNECE Guidance with unsubstantiated claims of a reduction in NH_3 emissions by up to 70%.

2.3.4. Specific techniques applying to pig housing

Most pig housing systems are based on slatted floors but some use of solid floors with bedding (Loyon et al., 2009; Philippe et al., 2011). For both types, a large range of techniques exist to reduce the emission of NH_3, CH_4 and N_2O which are similar to those for poultry (Philippe et al., 2011; Philippe & Nicks, 2015; UNECE, 2014). The UNECE Guidance document lists several techniques with respect to NH_3 emissions: (i) frequent removal of manure, (ii) minimizing the extent of dirtied floor, (iii) cooling of the manure stored in the building and (iv) air scrubbing. For liquid manure systems, most of the studies indicated lower emissions with partly slatted floors. However, in some cases, authors reported similar or even higher emission with a reduced proportion of slatted floor principally due to fouling on the solid part of the floor (Guingand & Granier, 2001; Philippe et al., 2013). Strategies to prevent soiling of the solid floor include the control of building conditions (temperature, ventilation), the pen design (location of partitions, feeding and watering facilities), appropriate animal density and the use of smooth inclined flooring. The overall impact of the proportion of slatted floor area on CH_4 and N_2O emissions is unclear (Guingand, Quiniou, Courboulay, 2010; Lägue et al., 2004; Philippe et al., 2013).

Several designs of the building slurry pit and manure removal strategy have been proposed to cut emissions. The reduction of the slurry pit liquid surface area with sloping pit walls may reduce NH_3 emission. Frequent manure removal, flushing and separating urine from faeces (by V-shaped scraper or conveyor belts) can reduce NH_3 loss by 50% and CH_4 emission by 10 to 90 % but their effect on N_2O emissions is negligible (de Vries, Aarnink, Groot Koerkamp, De Boer, 2013). Implementation of frequent manure removal techniques is relatively easy but installing flushing or under-slat separating strategies will require major modifications to existing houses. For new buildings, such systems are economically attractive as pits are not needed and operational costs are lower (Philippe & Nicks, 2015). For solid floor systems, the type and quantity of bedding can influence emissions as for cattle and poultry housing with lower NH_3 and N_2O emissions suggested in the case of generous bedding (Philippe & Nicks, 2015).

2.4. Principles of good manure storage

For manure storage outside the building, the reduction techniques listed in the UNECE guidance document for NH_3 emissions are based around (i), decreasing the exposed liquid surface area by installing covers or encouraging crusting and/or increasing the pit depth, (ii) decreasing the source strength by lowering the pH and (iii) minimizing mixing. As noted by Oenema, Velthof, Klimont, Winiwarter, (2012), these basic principles have been well-known for decades with no fundamentally new ideas emerging in the last decade. The most effective cover for slurry stores is a permanent solid roof (Sommer, Christensen, Nielson, Schjørring, 1993). Other options include the use of a floating plastic sheet or floating layers made up from peat, oil, straw, or polystyrene spheres amongst other materials (Hartung & Phillips, 1994, Loyon, Guiziou, Picard, Saint Cast P., 2007). Any of these barriers can reduce N loss from slurry storage by 80 to 90% so long as a complete cover is maintained throughout the storage period. The effectiveness is greatly compromised when surface cracks develop or when the covering material sinks into the slurry (Loyon, Guiziou, Picard, 2007; Rotz, 2004). The cheapest method is to take advantage of the natural crust formation on slurry, which is influenced by both the total dry matter content and the nature of the slurry solids present. Crusting forms less readily with pig slurry (than cattle) and it is unlikely to occur on any stores with a slurry dry matter content of below 1%. In cooler regions, the crust may indeed sink during winter due to reduced microbial activity generating fewer gas bubbles which otherwise aids buoyancy. The benefits are challenged especially in the case of an applied surface layer of straw (or natural crust) which may in fact be a source of N_2O and possible CH_4 emissions (Sommer, Petersen, Sogaard, 2000; Vanderzaag, Gordon, Glass, Jamieson, 2008). The use of oil covers have been shown to reduce emissions in pilot studies (Portejoie, Martinez, Guiziou, Coste, 2003) but in practice, this method is of limited benefit because the oil may be consumed by microorganisms, dissolved in the liquid or driven from part of the exposed surface by the wind.

A number of additives have been considered that may reduce the NH_3 emission from stored manure (Rotz, 2004) but are not listed in the UNECE guidance document due to scant evidence of their efficacy (Van der Stelt, Temminghoff, Van Vliet, Van Riemsdijk, 2007). For solid manure, mixing in additional fresh straw has the potential to reduce greenhouse gas emissions (especially N_2O) by 32% based on small scale stores (Chadwick et al., 2011; Yamulki, 2006). Compacting farmyard manure (FYM) may also be a method to

reduce NH_3 emission but this could increase N_2O emissions due to a higher manure density (Petersen & Sommer 2011; Sommer, Webb, Kupper, Groenestein, 2010).

2.5. The role of manure processing technologies

Despite optimized diets, improved livestock housing design and good manure management practice, many regions of intensive livestock produce quantities of manure that simply exceed the local land capacity. Therefore, some form of manure processing may be essential in order to meet the environmental protection criteria. This implies the reduction of the nutrient load or the production of exportable products, yet it is normally unattractive due to the costs implied (Petersen et al., 2007). Many manure processing systems already exist for livestock farming which can be grouped under mechanical/physical separation (Burton; 2007), aeration or anaerobic digestion and chemical methods (Burton & Turner, 2003).

A recent European survey (Foged et al., 2011) revealed anaerobic digestion as the most commonly used option from amongst a list of 45 livestock manure processing technologies considered. Nevertheless, the anaerobic digestate itself often requires further processing as little of the original nutrient content is removed. Manure has a fertilizer value, thus nutrient recovery strategies are generally preferred, as they result in a useful product that can reduce farming costs. Several technologies have been proposed to recover the N and P component including stripping/absorption (Laureni, Palatsi, Llovera, Bonmatí, 2013), chemical precipitation (Cerillo, Palatsi, Comas, Vicensc, Bonmatí, 2015), thermal concentration (Bonmatí, Campos, Flotats, 2003), and separation/concentration processes (Mondor, Ippersiel, Lamarche, Masse, 2008). Other systems that have not yet been used commercially (despite a high level of scientific interest) include struvite precipitation (magnesium ammonium phosphate) and partial nitrification - autotrophic ANAMMOX denitrification (Daumer, Picard, Saint-Cast, Dabert, 2010; Magri, Béline, Dabert, 2013).

Biological processing technologies for liquid manure are often used to reduce the nutrient load (especially N and P), with or without the production of concentrates usable as organic fertilizers. Other claimed benefits include easier handling of the manure during and following storage. Many treatment technologies modify the slurry characteristics (dry matter, organic content, pH, etc) and thus may have a positive or negative influence on N

emission during subsequent storage and following land application. Some technologies can be a source of emission themselves such as intensive aeration. With the objective to remove excess N from livestock slurries (by nitrification and de-nitrification) the method has been shown to also increase N_2O emissions (Béline, Martinez, Chadwick, Guiziou, Coste, 1999; Chadwick et al., 2011; Loyon, Guiziou, Béline, Peu, 2007) although most of the N is still removed as the harmless di-nitrogen gas. Slurry aeration can lead to a large overall decrease of GHG and NH_3 from the farm when compared to using storage alone (Loyon Guiziou, Béline, 2007). The recovered solid fraction from slurry separation is similar to untreated solid manure and it may likewise result in higher N_2O emissions during subsequent storage (Hansen, Henriksen, Sommer, 2006). On the other hand, Chadwick et al. (2011) report that the stored liquid fraction following separation can lead to lower overall emission of N_2O when compared to unseparated slurry.

2.6. Issues relating to the land-spreading of manure

Land spreading of manure is a major source of NH_3 emissions and in the longer term (weeks or months) of N_2O as well. There may be concurrent emissions of CH_4 but only in very small quantities (Chadwick & Pain, 1997; Rodhe, Abubaker, Ascue, Pell, Nordberg, 2012). The scale of emissions during and following manure spreading is influenced by several factors including, manure composition, the application method, the soil type and weather. Manure spreading can also lead to NO_3^- reaching surface and ground water, especially when nitrogen supplied exceeds crop requirements (Smith, Jackson, Pepper, 2001; Stoddard, Grove, Coyne, Thom, 2005). This can be reduced by good manure management as recommended by the Nitrates Directive (EC, 1996) and specified by codes of good agricultural practice which specify minimum storage periods, closed (non-spreading) periods and N application limits.

Ammonia and odour emissions during landspreading are reduced by the methods that reduce the contact area between the applied manure and the atmosphere (Webb, Pain, Bittman, Morgan, 2010). Thus, emissions from arable land can be cut by the rapid incorporation of applied slurry or by its direct injection into the soil (Table 2). In their review, Webb et al. (2010) reported that rapid incorporation of slurries or solid manures by ploughing within 4 to 6 hours is an effective abatement technique reducing NH_3 emission by up to 90%. Similar incorporation of solid manure has also been reported to reduce subsequent emissions of N_2O (Mkhabela et al., 2008; Webb, Chadwick, Ellis, 2004). During

spreading, NH_3 emissions are less with the trailing shoe (65%) or with an open-slot injection (70-80%) than with trailing hose (35%) when compared to a standard splashplate system (Webb et al. 2010). However, the same authors note a large variation in the span of reported data: trailing hose 0-75%, open-slot injection 23-99% and trailing shoe 38-74%. They also indicated that techniques which reduced NH_3 emissions during and following slurry application may increase emissions of N_2O. Abatement of the later emission is achieved by different strategies including the avoidance of wet and cold weather. However, to reduce emissions of NH_3 the advice is to spread in the cooler months of spring to enable utilization of the slurry nutrients (Table 2). The picture is even more confused when nitrate leaching is considered. Some studies show lower N losses when using an injection technique whilst others suggest an increase or no effect compared to surface spreading (Ball Coelho, Roy, Bruin, 2004; Rotz, 2004).

Table 2. Ammonia volatilization from slurry spread by trailing hose compared to the same when incorporated by plough or harrow within six hours of application and slurry injected into the soil. The effect of incorporation varies according to time lag between application and incorporation (Hansen, Sommer, Hutchings, Sorense, 2008).

Season	Soil surface and Crop	Application technique	NH_3-loss, % of applied TAN	
			Pig	Cattle
Spring	Bare soil	Trailing hose	17.1	32.6
		Trailing hose and incorporation	5.0	9.4
		Injection 3-5 cm bare soil	1.7	3.3
	Cereals	Trailing hose	14.8	28.1
	Grass	Trailing hose	17.1	32.6
		Injection 3-5 cm bare soil	12.8	24.5
Summer	Bare soil	Trailing hose	22.4	42.7
		Trailing hose and incorporation	6.5	12.4
		Injection 3-5 cm bare soil	2.2	4.3
	Grass	Trailing hose	22.3	42.5
		Injection 3-5 cm	16.7	31.9
Autumn	Grass	Trailing hose	21.8	41.6
		Injection 3-5 cm	16.4	31.2

2.7. Assessment, selection and verification of the appropriate BAT technique

2.7.1. Methods for measuring gas emissions from livestock farms

The different existing methods for measuring gas emissions from livestock range from simple static or dynamic chamber to complex micrometeorological methods, tracer technologies or indirect methods based on concentrations measurement and associated dispersion models (Bunton et al., 2007; Parker et al., 2013; Phillips, Lee, Scholtens, Garland, Sneath 2001; Shah, Westerman, Arogo, 2006).

The choice of measurement technique depends on the purpose of measurement, the time and resources and the emitting source itself. However, so far, very few of these methods are standardized or cited as reference methods with a known error that can thus be used to certify emission factors or to estimate the efficiency of BAT technologies in real conditions.

For animal houses under natural ventilation with fluctuating air flow a logical choice might be to use a tracer gas (SF_6, helium or even the carbon dioxide emitted from the livestock) as recommended by some workers (Samer et al., 2012; Schrade et al., 2012) but this method is cumbersome and expensive to be implemented on a large number of buildings (Calvet et al., 2013; Ogink, Mosquera, Calvet, Zhang, 2013). The importance of standardization has been widely recognized, (Hassouna, Robin, Charpiot, Edouard, Méda, 2013; Ogink et al., 2013; Parker et al., 2013,) but the development of reference methods is still lacking. A large part of the problem is the evaluation of the uncertainty around any measurement which reflects the actual variation of the emission parameter as well as the imprecision of the technique itself. Furthermore, there is often the problem of interference caused by the technique on the local environment generating the emission as well illustrated by static and dynamic chambers (Hassouna et al., 2010).

2.7.2. The problem of conflicting objectives

Over many years, research projects have tended to focus on one pollutant and sometimes just one part of the process (ie: housing, manure storage or spreading). Recent publications or reviews have clearly shown that some of these techniques have an impact

(positive or negative) on the same or a different pollutant at some other part of the farming process (Hristov et al., 2013). Abatement strategies are thus not necessarily additive (Eckard et al., 2010) because their cumulative impact at different steps in the system on the total emissions can be significantly different. For example, the effective abatement measures to reduce NH_3 can increase the emissions of N_2O and lead to the increased leaching of N from the soil (Oenema, Oudendag, Velthof, 2007).

Often, the local factors that describe the specific farm will greatly influence the performance of a given mitigation technology. Due to the difficulty of measuring gas emission, the level of reduction of some techniques has been determined under controlled laboratory or pilot-scale conditions. There is thus a question about the actual performance of such techniques when used at the commercial farm level, especially where good maintenance plays an important part. For example the use of the biotrickling filter might be expected to reduce the NH_3 emission from buildings by at least 70% (EC, 2003; Melse et al., 2012). However, some recent studies have shown that the actual NH_3 reduction could in fact be less than 50% at the farm level and, furthermore, that N_2O is also produced (Aguilar, Abaigar, Merino, Estellés, Calvet, 2010; Melse et al., 2012). A degree of swapping between NH_3 and N_2O emissions frequently occurs in solid manure composting systems, deep litter bedding or in slurry stores with a surface crust or a cover for reducing NH_3 losses (Clemens, Trimborn, Weiland, Amon, 2006; Vanderzaag et al., 2008). Even if official documents offer a list of techniques for emission reduction, they do not specify methods for the on-farm verification of the technique performance.

2.7.3. The use of modeling tools to aid the selection of the most appropriate methods

A strictly prescriptive and universal approach to implement a given abatement technology is not necessarily effective nor the best use of limited resources. Thus, how does one then select the most appropriate technology for a given farm? This question has led to the development of computer-based models to aid choice such as by Fabbri, Valli, Bonazzi (2004) with respect to the IPPC framework for Italy and by Gooday et al. (2014) for England and Wales or by Chardon et al. (2012) for France. These programs seek also to integrate the cost and overall effectiveness of a group of techniques to tackle multiple pollutants.

The EU Commission has directly sponsored research in this area including the Bat-Support Tool with the aim to establish a transparent assessment system to classify different proposed techniques in terms of BAT listing with respect to pigs, poultry and cattle types. The proposed system takes into account environmental, economic and ecological aspects as well as animal health and welfare (online tool; http://daten.ktbl.de/batsupport/startSeite.do).

The BATFARM model assesses a given technology or group of technologies based on its mitigation potential on emissions and its resource consumption when used in given different strategies and techniques applied to intensive cattle, pig and poultry farms (Aguilar et al., 2013). The model was set up using a combination of methodologies including known emission factors, empirical equations and process-oriented mechanisms adjusted to several European regions located in Portugal, Spain, France, UK and Ireland. Predictive data is calculated for the whole farm, considering housing, storage, treatment and landspreading stages, and includes: (i) animal feed, energy and water consumption, (ii) farm product output, (iii) manure quantity and composition and (iv), the predicted emissions of CH_4, NH_3 and N_2O. The model also enables comparison between two different farm scenarios defined by users.

2.8. Discussion - the way forward

2.8.1. The current uptake of technology in European livestock farms

Despite the availability of an abundance of methods and technologies over many years (all of which enable at least some reduction of emissions from livestock farming), the available evidence suggests, only a patchy uptake across Europe other than where legislation has enforced certain practices (e.g. the Netherlands). In most cases, the technology that is in use has often been selected as much for good farm management and operation as for any benefits linked to protecting the environment. Thus one can lay claim to the installation of adequate storage, well-ventilated and clean modern buildings along with good manure spreading practice all to the advancement of environmental protection (which it is) but the main reason for uptake would have been benefits linked to good farm operation. This win-win situation is of course not a problem but once the emphasis shifts to meet environmental criteria alone, uptake rapidly falls off and achieving further protection becomes much harder. Thus some success can be reported in the uptake of

those environmental measures that fit in well with farming such as reduced emission spreading methods (Table 3). However, it may be equally expected that additional factors such as odour complaints have also helped the increase in such technology.

A similar situation occurs in the use of manure treatment technology where the headline estimate of 6% of all manure in the EU being treated is largely accounted for by those methods such as separation and sedimentation that aid the farming operation (Foged et al., 2011). Likewise, anaerobic digestion is widely used where financial incentives linked to renewable energy policy can make this a profitable activity even at a modest scale and a similar argument might be made for composting. Once again, this is welcome as far as it goes in protecting the environment, but when there are few direct benefits to the farm (such as in the case of aeration systems to remove excess nitrogen) then justifying the cost to the farmer becomes much harder especially if there is uncertainty around the technology being proposed.

Table 3. Proportions of manures applied by reduced-ammonia emission techniques in the UK (Misselbrook et al., 2012).

Technique	% of all such manure applied
Cattle and pig slurry applied to grassland and arable by trailing shoe	0
Cattle slurry applied to grassland and arable by shallow injection	1
Cattle slurry applied to grassland by trailing hose	3
Cattle slurry applied to arable land by trailing hose	3
Cattle and pig FYM, applied to arable, incorporated within 4 h	3
Cattle and pig slurry, applied to arable, incorporated within 6 h	6
Poultry manure, applied to arable, incorporated within 4 h	8
Pig slurry applied to grassland and arable by shallow injection	11
Pig slurry applied to arable land by trailing hose	15
Cattle FYM, applied to arable, incorporated within 24 h	18
Pig slurry applied to grassland by trailing hose	19
Cattle and pig slurry, applied to arable, incorporated within 24 h	19
Pig FYM, applied to arable, incorporated within 24 h	26
Poultry manure, applied to arable, incorporated within 24 h	46

Thus the challenge might be summarized in two parts: (i) moving nearer a full uptake of those technologies falling into the category of "good farming practice" and which will achieve a measure of environmental protection at little or no additional cost and (ii) the use of specific regulation to ensure the use of additional measures that enable environment

targets to be fully met. Although BAT technology is split between both groups, it is very much the second group that requires much more attention if environmental protection policy is to be completely successful.

2.8.2. Barriers to further uptake of technologies that protect the environment

Inevitably, the reasons for resistance to implementing systems to specifically protect the environment from the livestock farming activity will vary widely depending on the local situation. However, the following account for much of the apprehension:

1. Direct cost to the livestock unit (perceived and/or actual)
2. Fears of excessive or unnecessary action for a given farm
3. Confusion and doubt over adequate performance in meeting environmental objectives

Cost is an obvious problem and it would be fair to say that few farmers will invest in additional technologies that bring them little benefit in the absence of regulatory pressure. Furthermore, there is the wide variation in what is perceived as "affordable" raises social and economic questions that lie outside the context of this review. The notion that some livestock farmers in the Netherlands (for example) are sustaining as much as 30 euro per tonne of manure whereas in other parts of Europe, even 1 euro per ton to be considered too much, reveals a more complicated situation.

Determining whether an individual farm meets environmental standards or requires investment in one or more areas is more of a policy than a scientific issue. It is generally easier to set targets such as the 170 kgN/hectare used in the Nitrates Directive than to use the broad based evaluation approach as in the IPPC/IED Directive. The problem is that the implementation of BAT may not meet any set target or indeed, it may be excessive or unnecessary. The concept of whole farm evaluation may be an alternative strategy but it is currently still too imprecise to implement.

The scientific contribution to improving technology uptake largely falls in the third group of those factors hindering technology uptake, namely the evaluation of the actual performance of proposed systems. Much of the related confusion often comes down to (a) the lack of a clear objective against which a system can be measured and (b) the lack of accurate, consistent and reliable measurement methods that can be used to validate the

claimed abatement in emissions or other environmental impact. Accuracy is particularly relevant when validating those simpler methods making a more modest improvement - indeed it might be asked if it possible to confirm methods cutting emissions by 10% or so if the available instruments only have an accuracy of ± 20%? One might expect then the establishment of BAM (Best Available Measurements) such as pursued in the VERA project (VERA, 2013) in parallel with listing BAT as the way forward.

Lastly, it is equally important to be clear on the farm reference points: a reduction in NH_3 emission of 10% may be claimed, but relating to what original amount? Likewise, what is the desired end point: reduction by a fixed amount or reducing emissions to below a maximum acceptable limit? Simulation modeling clearly has a role but it can only be applied once a strategy and farming system are clearly defined. Thus the idea of a standard farm type or arises – this might align to a typical existing livestock farm in a given region (against which improvements can be proposed) or an ideal farm (against which the shortcomings of existing farms can be measured).

2.9. Conclusions

Livestock farming across Europe remains responsible for a wide range of environmental problems and pressures for improvement are unlikely to subside in the near future. In response to this, a great deal of work has been carried out over the last 40 years or so with many technologies now available. However, there is also a sense of too much information and too little clear instruction on the most appropriate set of techniques to implement at a given farm.

Many methods have been aimed at a specific impact. Thus there are effective measures in the context of diet regime (eg: phased feeding, the use of phytase…), housing (eg: frequent manure removal and/or drying, use of bio-scrubbers, air cleaners….), manure handling (eg: use of covers for stores, physical treatments…) and land spreading (eg: seasonal and targeted applications, reduced emission spreading methods…). However, the sometimes negative interaction of methods can too easily be overlooked with the replacement of one pollutant such as NH_3 with another (such as N_2O) at the same or a different location downstream of the farming process.

Uptake of technologies to protect the environment is variable across Europe but mostly it is those methods perceived to be in keeping with good farming practice, or those

that benefit the farmer in other ways that are most widespread. Thus it is common to find optimized diets, well ventilated and clean housing, good systems for manure removal and storage. Technologies such as reduced emission spreading equipment that fit well into farming are also being used but others implying additional cost with little direct benefit to the farm (such as manure treatment) remain for the most part rare.

Some environmental targets will require many farms to move beyond just good farming practice and to implement one or more technologies carrying a real extra cost that brings little direct benefit to the farm. In most cases, motivation for such measures requires regulation. However, reticence amongst farmers might be considered justified in the light of the uncertainties linked with some of these technologies. Some of hesitation arises from the lack of consistent evaluation of such techniques set against clear objectives.

The scientific challenge is thus one of developing consistent procedures to ensure a fair and trustworthy evaluation of listed techniques (whether currently BAT or not). Implicit in this approach is (a) the concept of Best Available Measurements and (b) evaluation in the context of the whole farm. Modeling is likely to have a growing role but it will only be successful if built on the back of clearly defined strategies.

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2.11. References

Aguilar M., Abaigar A., Merino P., Estellés F. and Calvet S. (2010). Effect of a bioscrubber on NH_3, N_2O, CH_4 and CO_2 emissions from a pig facility in Spain. Proceedings of the AgEng 2010 Conference Towards Environmental Technologies, September 6-8, 2010, Clermont-Ferrand, France.

Aguilar M., Arriaga H., Dupard P., Lalor S., Fragoso R., Pahl O., Abaigar A., Cordovín L., Boyle M., Lanigan G., Loyon L. and Merino P. (2013). BATFARM Software: a support tool in the selection of environmental strategies in livestock operations in the Atlantic Region. Proceedings of the 15th International conference Ramiran, June 3-5, 2013, Versailles, France.

Ball Coehlo B., Roy R. and Bruin A. (2004). Nitrate leaching as affected by liquid swine manure and cover cropping in sandy soil of Southwestern Ontario. Can. J. Soil Sci. 84, 187-197.

Beline F., Martinez J., Chadwick D., Guiziou F. and Coste C.M. (1999). Factors affecting Nitrogen Transformations and Related Nitrous Oxide Emissions from Aerobically Treated Piggery Slurry. J. Agric. Eng. Res. 73, 235-243.

Bioteau T., Burton C., Guiziou F. and Martinez J. (2010). Assessment of manure management practices across the EU 27. Proceedings of the 14th International Conference Ramiran, September 13-15, 2010 Lisboa, Portugal.

Bonmatí A., Campos E. and Flotats X. (2003). Concentration of pig slurry by evaporation: anaerobic digestion as the key process. Water Sci. Technol. 48, 189-194.

Bunton B., O'Shaughnessy P., Fitzsimmons S., Gering J., Hoff S., Lyngbye M., Thorne P., Wasson J. and Werner M. (2007). Monitoring and modeling of emissions from concentrated animal feeding operations: overview of methods. Environ. Health Perspect. 115, 303-307.

Burton C.H. (2007). The potential contribution of separation technologies to the management of livestock manure. Livest. Sci. 112, 208-216.

Burton C.H. (2009). Reconciling the new demands for food protection with environmental needs in the management of livestock wastes. Bioresour. Technol. 100, 5399-5405.

Burton C.H. and Turner C. (2003). Manure management. Treatment strategies for sustainable agriculture. 2nd Edition. Silsoe Research Institute, Wrest Park, Silsoe, Bedford, UK 490p.

Calvet S., Gates R.S., Zhang G., Estellés F., Ogink N.W.M., Pedersen S. and Berckmans D. (2013). Measuring gas emissions from livestock buildings: A review on uncertainty analysis and error sources. Biosyst. Eng. 116, 221-231.

Cerrillo M., Palatsi J., Comas J., Vicensc J. and Bonmatí A. (2015). Struvite precipitation as a technology to be integrated in a manure anaerobic digestion treatment plant – removal efficiency, crystal characterization and agricultural assessment. J Chem Technol Biotechnol 90, 1135-1143.

Chadwick D., Sommer S., Thorman R., Fangueiro D., Cardenas L., Amon B. and Misselbrook T. (2011). Manure management: implications for greenhouse gas emissions. Anim. Feed Sci. Technol. 166–167, 514-531.

Chadwick D.R. and Pain B.F. (1997). Methane fluxes following slurry applications to grasslands soils: laboratory experiments. Agric. Ecosyst. Environ.63, 51-60.

Chardon X., Rigolot C., Baratte C., Espagnol S, Raison C., Martin-Clouaire R., Rellier J.-P., Le Gall A., Dourmad J.Y., Piquemal B., Leterme P., Paillat J.M., Delaby L., Garcia F., Peyraud J.L., Poupa J.C., Morvan T. and Faverdin P. (2012). MELODIE: a whole-farm model to study the dynamics of nutrients in dairy and pig farms with crops. Animal, 6 (10), 1711-1721.

Clark O.G., Moehn S., Edeogu I., Price J. and Leonard J. (2005). Manipulation of dietary protein and nonstarch polysaccharide to control swine manure emissions. J. Environ. Qual. 34, 1461-1466.

Clemens J., Trimborn M., Weiland P. and Amon B. (2006). Mitigation of greenhouse gas emissions by anaerobic digestion of cattle slurry. Agric. Ecosyst. Environ.112, 171-177.

Daumer M.L., Picard S., Saint-Cast P. and Dabert P. (2010). Technical and economical assessment of formic acid to recycle phosphorus from pig slurry by a combined acidification-precipitation process. J. Hazard. Mater.180, 361-365.

de Vries J.W., Aarnink A.J.A., Groot Koerkamp P.W.G. and De Boer I.J.M. (2013). Life cycle assessment of segregating fattening pig urine and feces compared to conventional liquid manure management. Environ. Sci. Technol.47, 1589-1597.

Doreau M., Makkar H.P.S. and Lecomte P. (2013). The contribution of animal production to agricultural sustainability. In Oltjen J.W., Kebreab E., Lapierre H., Energy and Protein metabolism and nutrition in sustainable animal production, Wageningen Acad. Publ., the Netherlands.

Doreau M., Martin C., Eugène M., Popova M. and Morgavi D.P. (2011). Leviers d'action pour réduire la production de méthane entérique par les ruminants.In : Gaz à effet de serre en élevage bovin : le méthane. Dossier. INRA Productions Animales 24, 461-474.

Dourmad J.Y. and Jondreville C. (2007). Impact of nutrition on nitrogen, phosphorus, Cu and Zn in pig manure, and on emissions of ammonia and odours. Livest. Sci. 112, 192-198.

EC (1996). Council Directive 96/61/EC concerning integrated pollution prevention and control. Official Journal L 257, 26-40.

EC (2000). Directive 2000/60/EC of the European parliament and of the council of 23 October 2000 establishing a framework for the community action in the field of water policy. Off. J. Eur. Commiss. L327, 1-72.

EC (2001). Directive 2001/81/EC of the European Parliament and of the Council of 23 October 2001 on National Emission Ceilings for Certain Atmospheric Pollutants. http://eur-lex.europa.eu.

EC (2003). Integrated Pollution Prevention and Control (IPPC) - Reference Document on Best Available Techniques for intensive Rearing of Poultry and pigs. European Commission, 383p

EC (2010). Directive 2010/75/EU of the European Parliament and the Council of 24 November on industrial emissions. Official Journal of the European Union L 334/ 17. 17.12.2010.

Eckard R.J., Grainger C. and de Klein C.A.M. (2010). Options for the abatement of methane and nitrous oxide from ruminant production: a review. Livest. Sci. 130, 47-56.

EEA (2010). European Union emission inventory report 1990–2010 under the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP). Technical report No 7/2010, 90p.

EEC (1991). Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources. Official Journal 1991;L 375 [31/12/1991].

Fabbri C., Valli L. and Bonazzi G. (2004). A method to assess the reduction of Ammonia and Methane emissions by application of Bat in intensive livestock farming. Proceedings of the 11th International Conference Ramiran, 6-9 October 2004, Murcia, Spain.

Fangmeier A., Hadwiger-Fangmeier A., Van der Eerden L. and Jäger H.J. (1994). Effects of atmospheric ammonia on vegetation: a review. Environ. Pollut. 86, 43-82.

Foged H.L., Flotats X., Bonmati A.B., Palatsi J., Magri A. and Schelde K.M. (2011). Inventory of manure processing activities in Europe. Technical Report No. I concerning "Manure Processing Activities in Europe" to the European Commission, Directorate-General Environment., 138p.

Gerber P.J., Steinfeld H., Henderson B., Mottet A., Opio C., Dijkman J., Falcucci A. and Tempio G. (2013). Tackling climate change through livestock - A global assessment of emissions and mitigationopportunities.Food and Agriculture Organization of the United Nations (FAO), Rome.

Gilchrist M., Greko C., Wallinga D., Beran G., Riley D. and Thorne P. (2007). The potential role of concentrated animal feeding operations in infectious disease epidemics and antibiotic resistance. Environ. Health Perspect. 115, 313-316.

Gilhespy S.L., Webb J., Chadwick D.R., Misselbrook T.H., Kay R., Camp V., Retter A.L. and Bason A. (2009). Will additional straw bedding in buildings housing cattle and pigs reduce ammonia emissions? Biosyst. Eng. 102, 180-189.

Gooday R.D., Anthony S.G., Chadwick D.R., Newell-Price P., Harris D., Duethmann D., Fish R., Collins A.L. and Winter M. (2014). Modelling the cost-effectiveness of mitigation methods for multiple pollutants at farm scale. Sci. Total Environ. 468-469, 1198-1209.

Guingand N. and Granier R. (2001). Comparaison caillebotis partiel et caillebotis intégral en engraissement. Effets sur les performances zootechniques et sur l'émission d'ammoniac. Journées de la Rech. Porcine Paris 33, 31-36.

Guingand N., Quiniou N. and Courboulay V. (2010). Comparison of ammonia and greenhouse gas emissions from fattening pigs kept either on partially slatted floor in cold conditions or on fully slatted floor in thermoneutral conditions. Journées de la Rech. Porcine 42, 277-284.

Hansen M.N., Henriksen K. and Sommer S.G. (2006). Observations of production and emission of greenhouse gases and ammonia during storage of solids separated from pig slurry: effects of covering. Atmos. Environ. 40, 4172-4181.

Hansen M.N., Sommer S.G., Hutchings N.J. and Sorense P. (2008). Emissions Factors for Calculation of Ammonia Volatilization by Storage and Application of Animal Manure. DJF Rapport Husdyrbrug 84, Ministry of Food, Agriculture and Fisheries, Danish Institute of Agricultural Sciences, Tjele.

Hartung J. and Phillips V.R. (1994). Control of gaseous emissions from livestock buildings and manure stores. J. Agric. Eng. Res. 57, 173-189.

Hassouna M., Espagnol S., Lagadec S., Loyon L., Guiziou F. and Robin P. (2010). Comparison of different dynamic floating chambers to assess gaseous emissions from stored pig slurry. International Symposium on Air Quality and Manure Management for Agriculture Conference Proceedings, 13-16 September 2010, Dallas, Texas, USA.8p.

Hassouna M., Robin P., Charpiot A., Edouard N. and Méda B. (2013). Infrared photoacoustic spectroscopy in animal houses: effect of non-compensated interferences on ammonia, nitrous oxide and methane air concentrations. Biosyst. Eng. 114, 318-326.

Hoffmann I. (2011). Livestock biodiversity and sustainability. Livest. Sci. 139, 69-79.

Hristov A.N., Oh J., Lee C., Meinen R., Montes F., Ott T., Firkins J., Rotz A., Dell C., Adesogan A., Yang W., Tricarico J., Kebreab E., Waghorn G., Dijkstra J. and Oosting S. (2013). Mitigation of greenhouse gas emissions in livestock production – A review of technical options for non-CO_2 emissions. Edited by Pierre J. Gerber, Benjamin Henderson and Harinder P.S. Makkar. FAO Animal Production and Health Paper No. 177. FAO, Rome, Italy.

Jarret G., Cerisuelo A., Peu P., Martinez J. and Dourmad J.Y. (2012). Impact of pig diets with different fibre contents on the composition of excreta and their gaseous emissions and anaerobic digestion. Agric. Ecosyst. Environ. 160, 51-58.

Jarvie H.P., Neal C., Withers P.J.A., Wescott C. and Acornley R.M. (2005). Nutrient hydrochemistry for a groundwater-dominated catchment: the Hampshire Avon, UK. Sci. Total Environ. 344, 1-3, 143-158.

Lägue C., Marquis A., Godbout S., Legace R., Fonstad T.A., Lemay S.P., Joncas R. and Jaulin L. (2004). Greenhouse Gas Emissions from Swine Operations in Québec and Saskatchewan: Benchmark Assessments. University of Saskatchewan, Saskatchewan, Canada.

Laureni M., Palatsi J., Llovera M. and Bonmatí A. (2013). Influence of pig slurry characteristics on ammonia stripping efficiencies and on the quality of the recovered ammonium-sulfate solution. J Chem Tech Biotech 88(9), 1654 -1662.

Le P.D., Aarnink A.J.A. and Jongbloed A.W. (2009). Odour and ammonia emission from pig manure as affected by dietary crude protein level. Livest. Sci. 121, 267-274.

Loyon L. and Burton C. (2013). Reconciling livestock management to the environment. Applying Best Available Technique (BAT): from the lab to the farm. European BATFARM workshop, Rennes, France - 19 & 20 March 2013. (http://www.batfarm.eu/news-and-activities/2013-workshop-rennes)

Loyon L., Burton C. and Guiziou F. (2009). Intensive livestock farming systems in use across Europe: a review of the current situation relating to IPPC based on recent data gathered by questionnaire. Final Report as the principle deliverable of Task 3 (and covering Tasks 2 relating to the production of the questionnaire) EU Project BAT-Support, 173p.

Loyon L., Guiziou F., Béline F. and Peu P. (2007). Gaseous emissions (NH_3, N_2O, CH_4 and CO_2) from the aerobic treatment of piggery slurry. Comparison with a conventional storage system. Biosyst. Eng. 97, 472-480.

Loyon L., Guiziou F., Picard S. and Saint Cast P. (2007). Farm-scale evaluation of three cover systems (peat, synthetic sheet and polystyrene balls) for reducing ammonia emissions from swine manure stores and subsequent land spreading. International Ammonia Conference in Agriculture.Ede, The Netherlands, NLD, 19-21 March 2007. Gert-Jan Monteny and Eberhard Hartung, Wageningen Academic Publishers, The Netherlands, NLD, pp. 226-227.

Magrí A., Béline F. and Dabert P. (2013). Feasibility and interest of the anammox process as treatment alternative for anaerobic digester supernatants in manure processing: an overview. J. Environ. Manage. 131, 170-184.

Martin C., Morgavi D.P. and Doreau M. (2010). Methane mitigation in ruminants: from microbe to the farm scale. Animal 4, 351-365.

Melse R.W., Ploegaert J.P.M. and Ogink N.W.M. (2012). Biotrickling filter for the treatment of exhaust air from a pig rearing building: ammonia removal performance and its fluctuations. Biosyst. Eng. 113, 242-252.

Misselbrook T.H. and Powell J.M. (2005). Influence of Bedding Material on Ammonia Emissions from Cattle Excreta. Journal of dairy science 88, 4304-4312.

Misselbrook T.H., Gilhespy S.L., Cardenas L., Chambers B.J., Smith K.A., Williams J. and Dragosits U. (2012). Inventory of Ammonia Emissions from UK Agriculture 2011, Defra Contract AC0112, Inventory Submission Report November 2012, 35pp.

Misselbrook T.H., Webb J. and Gilhespy S.L. (2006). Ammonia emissions from outdoor concrete yards used by livestock: quantification and mitigation. Atmos. Environ. 40, 6752-6763.

Misselbrook T.H., Webb J., Chadwick D.R., Ellis S. and Pain B.F. (2001). Gaseous emissions from outdoor concrete yards used by livestock. Atmos. Environ. 35, 5331-5338.

Mkhabela M.S., Madani A., Gordon R., Burton D., Cudmore D., Elmi A. and Hart W. (2008). Gaseous and leaching nitrogen losses from no-tillage and conventional tillage systems following surface application of cattle manure. Soil Till. Res. 98, 187-199.

Mondor M., Ippersiel D., Lamarche F. and Masse L. (2008). Fouling characterization of electrodialysis membranes used for the recovery and concentration of ammonia from swine manure. Bioresour. Technol. 100, 566-571.

Moreau P., Ruiz L., Vertès F., Baratte C., Delaby L., Faverdin P., Gascuel-Odoux C., Piquemal B., Ramat E., Salmon-Monviola J. and Durand P. (2013). CASIMOD'N: an agro-hydrological distributed model of catchment-scale nitrogen dynamics integrating farming system decisions. Agric. Syst. 118, 41-51.

Nahm K.H. (2007). Feed formulations to reduce N excretion and ammonia emission from poultry manure. Bioresour. Technol. 98, 2282-2300.

Ndegwa P.M., Hristov A.N., Arogo J. and Sheffield R.E. (2008). A review of ammonia emission mitigation techniques for concentrated animal feeding operations. Biosyst. Eng. 100, 453-469.

Oenema O., Oudendag D. and Velthof G.L. (2007). Nutrient losses from manure management in the European Union. Livest. Sci. 112, 261-272.

Oenema O., Velthof G., Klimont Z. and Winiwarter W. (2012). Emissions from agriculture and their control potentials. TSAP Report #3, IIASA. 47p.

Ogink N.W.M., Mosquera J., Calvet S. and Zhang G. (2013). Methods for measuring gas emissions from naturally ventilated livestock buildings: developments over the last decade and perspectives for improvement. Biosyst. Eng. 116, 297-308.

Osada T., Takada R. and Shinzato I. (2011). Potential reduction of greenhouse gas emission from swine manure by using a low-protein diet supplemented with synthetic amino acids. Anim. Feed Sci. Technol. 166, 562-574.

Parker D., Ham J., Woodbury B., Cai L., Spiehs M., Rhoades M., Trabue S., Casey K., Todd R. and Cole A. (2013). Standardization of flux chamber and wind tunnel flux measurements for quantifying volatile organic compound and ammonia emissions from area sources at animal feeding operations. Atmos. Environ. 66, 72-83.

Pellerin S., Bamière L., Angers D., Béline F., Benoît M., Butault J.P., Chenu C., Colnenne-David C., De Cara S., Delame N., Doreau M., Dupraz P., Faverdin P., Garcia-Launay F., Hassouna M., Hénault C., Jeuffroy M.H., Klumpp K., Metay A., Moran D., Recous S., Samson E., Savini I. and Pardon L. (2013). How can French agriculture contribute to reducing greenhouse gas emissions? Abatement potential and cost of ten technical measures. Synopsis of the study report, INRA, France, 92 p.

Petersen S.O. and Sommer S.G. (2011). Ammonia and nitrous oxide interactions: roles of manure organic matter management. Anim. Feed Sci. Technol. 166–167, 503-513.

Petersen S.O., Sommer S.G., Béline F., Burton C., Dach J., Dourmad J.Y., Leip A., Misselbrook T., Nicholson F., Poulsen H.D., Provolo G., Sørensen P., Vinnerås B., Weiske

A., Bernal M.P., Böhm R., Juhász C. and Mihelic R. (2007). Recycling of livestock manure in a whole-farm perspective. Livestock Science 112, 180-191.

Philippe F.X. and Nicks B. (2015). Review on greenhouse gas emissions from pig houses: Production of carbon dioxide, methane and nitrous oxide by animals and manure. Agriculture, Ecosystems & Environment 199, 10-25.

Philippe F.X., Cabaraux J.F. and Nicks B. (2011). Ammonia emissions from pig houses: influencing factors and mitigation techniques. Agric. Ecosyst. Environ.141, 245-260.

Philippe F.X., Cabaraux J.F., Laitat M., Stilmant D., Wavreille J. and Nicks B. (2013). Les impacts environnementaux du choix des modalités d'hébergement des porcs charcutiers. University of Liège, Liège, Belgium.

Philippe F.X., Laitat M., Canart B., Vandenheede M. and Nicks B. (2007). Comparison of ammonia and greenhouse gas emissions during the fattening of pigs, kept either on fully slatted floor or on deep litter. Livest. Sci. 111, 144-152.

Philippe F.X., Remience V., Dourmad J.Y., Cabaraux J.F., Vandenheede M. and Nicks B. (2008). Les fibres dans l'alimentation des truies gestantes: effets sur la nutrition, le comportement, les performances et les rejets dans l'environnement. INRA Prod.Anim. 21, 277-290.

Phillips V.R., Lee D.S., Scholtens R., Garland J.A. and Sneath R.W. (2001). A Review of Methods for measuring Emission Rates of Ammonia from Livestock Buildings and Slurry or Manure Stores, Part 2: monitoring Flux Rates, Concentrations and Airflow Rates. J. Agric. Eng. Res. 78, 1-14.

Portejoie S., Martinez J., Guiziou F. and Coste C.M. (2003). Effect of covering pig slurry stores on the ammonia emission processes. Bioresour. Technol. 87, 199-207.

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

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

Rodhe L., Abubaker J., Ascue J., Pell M. and Nordberg Å. (2012). Greenhouse gas emissions from pig slurry during storage and after field application in northern European conditions. Biosyst. Eng. 113, 379-394.

Rotz C.A. (2004). Management to reduce nitrogen losses in animal production. J. Anim. Sci. 82, 119-137.

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

Schrade S., Zeyer K., Gygax L., Emmenegger L., Hartung E. and Keck M. (2012). Ammonia emissions and emission factors of naturally ventilated dairy housing with solid floors and an outdoor exercise area in Switzerland. Atmos. Environ. 47, 183-194.

Seedorf J. and Hartung J. (1999). Survey of ammonia concentrations in livestock buildings. J. Agric. Sci., Cambridge 133, 433-437.

Shah S.B., Westerman P.W. and Arogo J. (2006). Measuring ammonia concentrations and emissions from agricultural land and liquid surfaces: a review. J Air Waste Manage. 56, 945-960.

Smith K.A., Jackson D.R. and Pepper T.J. (2001). Nutrient losses by surface run-off following the application of organic manures to arable land. 1. Nitrogen. Environ. Pollut. 112, 41-51.

Sommer S.G., Christensen B.T., Nielson N.E. and Schjørring J.K. (1993). Ammonia volatilization during storage of cattle and pig slurry: effect of surface cover. J. Agric. Sci., Camb. 121, 63-71.

Sommer S.G., Østergård H.S., Løfstrøm P., Andersen H.V. and Jensen L.S. (2009). Validation of model calculation of ammonia deposition in the neighbourhood of a poultry farm using measured NH_3 concentrations and N deposition. Atmos. Environ. 43, 915-920.

Sommer S.G., Petersen S.O. and Sogaard H.T. (2000). Greenhouse gas emission from stored livestock slurry. J. Environ. Qual. 29, 744-751.

Sommer S.G., Webb J., Kupper T. and Groenestein K. (2010). Gaseous emissions during the management of litter-based manures. Proceedings of the 14th International Conference Ramiran, September 13-15, 2010 Lisboa, Portugal.

Sommer S.G., Zhang G.Q., Bannink A., Chadwick D., Misselbrook T., Harrison R., Hutchings N.J., Menzi H., Monteny G.J., Ni J.Q., Oenema O. and Webb J. (2006). Algorithms Determining Ammonia Emission from Buildings Housing Cattle and Pigs and from Manure Stores. Adv. Agron. 89, 261-335.

Steinfeld H., Gerber P., Wassenaar T., Castel V., Rosales M. and de Haan C. (2006). Livestock's long shadow - Environmental issues and options. FAO, Rome, Italy

Stoddard C.S., Grove J.H., Coyne M.S. and Thom W.O. (2005). Fertilizer, Tillage, and Dairy Manure Contributions to Nitrate and Herbicide Leaching. J. Environ. Qual. 34, 1354-1362.

Swierstra D., Braam C.R. and Smits M.C.J. (2001). Grooved floor system for cattle housing: ammonia emission reduction and good slip resistance. Appl. Eng. Agric. 17, 85-90.

UN (1997). Kyoto protocol to the United Nations framework convention on climate change. NY, USA. 1997.

UNECE (1999). Protocol to the 1979 convention on long-range transboundary air pollution to abate acidification, eutrophication and ground-level ozone (Gothenburg protocol).

UNECE (2014). Guidance document on preventing and abating ammonia emissions from agricultural. ECE/EB.AIR/120. 7 February 2014, UNECE, Geneva, 100p.

Van der Stelt B., Temminghoff E.J.M., Van Vliet P.C.J. and Van Riemsdijk W.H. (2007). Volatilization of ammonia from manure as affected by manure additives, temperature and mixing. Bioresour. Technol. 98, 3449-3455.

Vanderzaag A.C., Gordon R.J., Glass V.M. and Jamieson R.C. (2008). Floating covers to reduce gas emissions from liquid manure stores: a review. Appl. Eng. Agric. 24, 657-671.

Veldkamp T., Star L., van der Klis J.D. and van Harn J. (2012). Reduction of ammonia emission from poultry houses by nutrition. Report 490. Wageningen UR Livestock Research, Lelystad, The Netherlands.

VERA (2013). Verification of Environmental Technologies for agricultural Production. http://www.veracert.eu/en/technology-manufacturers/test-protocols/

Webb J., Chadwick D. and Ellis S. (2004). Emissions of ammonia and nitrous oxide following incorporation into the soil of farmyard manures stored at different densities. Nutr. Cycl. Agroecosyst. 70, 67-76.

Webb J., Pain B., Bittman S. and Morgan J. (2010). The impacts of manure application methods on emissions of ammonia, nitrous oxide and on crop response: a review. Agric. Ecosyst. Environ.137, 39-46.

Yamulki S. (2006). Effect of straw addition on nitrous oxide and methane emissions from stored farmyard manures. Agric. Ecosyst. Environ.112, 140-145.

A whole farm model for Ammonia and Greenhouse Gas Emissions from Livestock Operations.

Part 1: Model description.

To be submitted as:

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A whole farm model for Ammonia and Greenhouse Gas Emissions from Livestock Operations. Part 1: Model description.

Abstract

A model has been developed to assess the emissions of nitrous oxide (N_2O), methane (CH_4) and ammonia (NH_3) as a consequence of different strategies and techniques implemented on intensive cattle, pig and poultry farms. Designed with methodological rigour and as a user-friendly tool, the model calculates mass balances throughout housing, storage, treatment and landspreading stages, on a cumulative monthly and annual basis in order to estimate manure evolution (mass, dry and organic matter, N, P, K, Cu and Zn contents), related emissions, feed consumptions and animal production. The model was set up using a combination of methodologies including emission factors, empirical equations and process-oriented mechanisms. Different mitigation strategies, farmer practices and climatic conditions which have significant effects on gaseous emissions have been considered (e.g. low protein rations, different types of flooring within housing, covers and additives in storages, anaerobic digestion, composting, slurry injectors, etc). Results produced by the model aims to identify the key stages giving rise to particular farm emissions, which would most benefit from implementing environmental techniques and predicts farm emission variation under different scenarios. Results must be interpreted as indicative of the relative emission reduction achieved due to implementation of mitigation practices rather than absolute values. Further research and validation is needed to extend its use.

Keywords

Nitrous Oxide, Methane, BAT, Pollution, Environment, Manure Management.

Nomenclature

$ABATE_{DM}$	Ammonia emission reduction due to manure dry matter content at landspreading (%)
$ABATE_{INCORP}$	Ammonia emission reduction due to manure incorporation at landspreading (%)
$ABATE_{METHOD}$	Ammonia emission reduction due to application method at landspreading (%)
A_c	Annual cost of capital (euros per year)
A_e	Energy/fuel cost (euros per year)
A_l	Labour cost (euros per year)
A_r	Reparations/maintenance cost (euros per year)
$B0$	Maximum methane producing capacity (m^3 CH_4 per kg VS)
C	Total capital investment cost (euros)
E_{CH4Ent}	Enteric methane emission (kg)
E_{CH4Man}	Manure management methane emission (kg)
EF_{N2O}	Nitrous oxide emission factor (kg N_2O-N per kg N excreted)
$EF_{N2OCover}$	Nitrous oxide emission factor due to covers at slurry storage (kg N_2O-N per kg TN)
$EF_{N2OCrust}$	Nitrous oxide emission factor due to natural crust at slurry storage (kg N_2O-N per kg TN)
$EF_{N2ODirect}$	Direct nitrous oxide emission factor during grazing (% of N excreted)
$EF_{N2OIndirect}$	Indirect nitrous oxide emission factor during grazing (% of NH_3-N)
$EF_{N2OLeaching}$	Nitrous oxide emission factor during grazing from N leaching (% of N excreted)
EF_{NH3}	Ammonia emission factor (kg NH_3-N per kg N excreted)
$EF_{NH3grazing}$	Ammonia emission factor during grazing (kg NH_3-N per kg N excreted)
$EF_{NH3land}$	Ammonia emission factor at landspreading (kg NH_3-N per kg TAN applied)
E_{N2O}	Nitrous oxide emission (kg)
E_{NH3}	Ammonia emission (kg)
MCF	Methane conversion factor (%)
n	The life of investment (years)
N_{ex}	Nitrogen excreted (kg)
$N_{exgrazing}$	Nitrogen excreted during grazing (kg)
r	The interest rate (decimal of 1)
R_{TAN}	Total ammoniacal N application rate at landspreading (kg TAN per ha)
TAN_{ABATE}	Total ammoniacal N abated at landspreading (kg TAN per ha)
T_c	Total cost (euros per year)
VF_{Dur}	Variation factor of gas emission due to duration at solid storage
VF_{InTemp}	Ammonia emission variation factor due to indoor temperature at housing
VF_{Ndil}	Ammonia emission variation factor due to nitrogen dilution in the slurry
VF_{Stall}	Ammonia emission variation factor due to type of facility and floor (dairy cows)
VF_{Temp}	Ammonia emission variation factor due to slurry temperature
VF_{Type}	Variation factor of gas emission due to manure type at solid storage
VF_{Vent}	Ammonia emission variation factor due to ventilation rates in buildings
VF_{XAdd}	Emission variation factor of gas X due to additives at slurry storage
VF_{Xbed}	Emission variation factor of gas X due to type of bedding material
VF_{XCover}	Emission variation factor of gas X due to covers at slurry storage
VF_{XCrust}	Emission variation factor of gas X due to natural crust at slurry storage
VF_{Xdrink}	Emission variation factor of gas X due to type of drinkers
VF_{Xfloor}	Emission variation factor of gas X due to type of floor at housing
VF_{Xfreq}	Emission variation factor of gas X due to frequency of slurry removal at housing
VF_{XOther}	Emission variation factor of gas X due to other mitigation strategies at housing
VF_{XRem}	Emission variation factor of gas X due to manure removal system at housing
VF_{Xthick}	Emission variation factor of gas X due to bedding thickness
VS_{ex}	Organic matter excreted (kg)

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3.1. Introduction

Environmental impacts on natural resources have led to a complex set of new and changing policies in order to minimize those damages, such as: EU Nitrates Directive (91/676/CEE), EU Water Framework Directive (2000/60/CE), National Emission Ceilings Directive (2001/81/CE), the proposed Soil Framework Directive (2004/35/CE), the Kyoto protocol under the United Nations Framework Convention on Climate Change, and the Industrial Emissions Directive (IED) (2010/75/EU). One of the defined sectors under the IED is intensive livestock farming with more than 40000 poultry, 2000 fattening pig or 750 sow places. These facilities must have a permit to operate based on the implementation of Best Available Techniques (BAT) in the whole farm production process.

However, despite the abundance of possible techniques proposed to reduce gas emissions (EC, 2003; EC, 2013), in practice there is a low level of BAT implementation, especially when farms have to move beyond just good farming practices or "win-win strategies". The diversity of farming systems, management options and socioeconomic conditions, their perceived high cost, in combination with a lack of the demonstrable benefits of these techniques in commercial farms, may be some of the reasons for this (Loyon et al., 2016).

Essentially, any valid strategy to model a possible abatement technique must consider the context of the whole farm. In addition, a specific environmental technique (or a combination of several of them) may have different effects depending on which point(s) along the animal production process it acts and the potential for interactions with the current situation. Computer-based models and decision support tools can assist producers and technicians with the identification of the most suitable environmental strategy for a particular farm situation (Karmakar, Nketia, Laguë, Agnew, 2010).

Different models have been developed to estimate gaseous emissions from agriculture. The dynamic ammonia model DYNAMO was used to calculate the Swiss NH_3 emission inventory and abatement potential (Menzi, Rüttimann, Reidy, 2003; Reidy, Rhim, Menzi, 2008). Based on expert consultations and new literature, the algorithms in the DYNAMO model were reviewed to develop a new model AGRAMMON (Menzi, Bonjour, Zaucker, Leuenberger, Reidy, 2009). The models used for NH_3 emission inventory determination in Denmark and the Netherlands, are described respectively in

Hutchings, Sommer, Andersen, and Asman (2001) and Velthof et al. (2012). The NARSES model (Webb & Misselbrook, 2004; Webb et al., 2006) estimates NH_3 emission from UK agriculture and identifies the most cost-effective means of reducing it. In relation to this, FARMSCOPER (Gooday et al., 2014; Zhang, Collins, Gooday, 2012) was developed to predict diffuse agricultural pollution from representative UK farm systems associated with multiple pollutants and to determine the cost of implementation and the effectiveness of one or more mitigation methods in reducing the emissions of these pollutants. GAS-EM (Gaseous Emissions) has been used to calculate the German agricultural emission inventory (Dämmgen, Lüttich, Döhler, Eurich-Menden, Osterburg, 2003) and as a policy advice tool. FASSET (Farm Assessment Tool) is a whole-farm dynamic model, which can be used to evaluate consequences of changes in regulations, management, prices and subsidies on a range of indicators for sustainability at the farm level (Berntsen, Petersen, Olesen, Hutchings, 1999; Berntsen, Petersen, Jacobsen, Olesen, Hutchings, 2003).

In Italy, Fabbri, Valli, and Bonazzi (2004) describes a method for calculation of CH_4 and NH_3 emissions considering BAT applied in pigs and poultry farms. Used in the Lombardy region, VALORE (Valorisation of Effluents) is a user-friendly software developed to cope with different livestock (i.e. cattle, swine, poultry, sheep, goats and horses) and to suggest and analyze alternative manure management options at the farm and territorial scale (Acutis et al., 2014; Provolo, Grimaldi, Riva, 2012).

Developed in France, the model MELODIE aims to evaluate the environmental impact of production strategies in integrated dairy, swine and crop farms by simulating nutrients flow over several decades at a farm scale (Chardon et al, 2007). The model MOLDAVI (Meda, Robin, Aubert, Dourmad, Hassouna, 2012) enables simulation of broiler managment systems taking into account animal performance, environmental and economical issues.

The Global Livestock Environmental Assessment Model (GLEAM) was developed by the FAO (Food and Agriculture Organization of the United Nations) to evaluate the environmental impact of the livestock sector and to assess intervention scenarios. It provides disaggregated and spatially explicit estimations of livestock production and GHG (Greenhouse Gas) emissions based on Tier 2 methodologies (http://www.fao.org/gleam).

However, there is a lack of support and knowledge transfer models designed for farm-scale operations that combine a variety of methodologies and uses inputs well known

by the farmers rather than requiring additional data collection. Besides, for the farmer to be able to reduce farm emission, animal husbandry has to be considered in the entire process chain, from animal to manure export from the farm (EC, 2015). Our model was developed precisely for this purpose: designed with methodological rigour and as a user-friendly tool that can be used by farmers and technicians to assess the emissions of N_2O, CH_4 and NH_3 resulting from different strategies and techniques implemented on intensive cattle, pig and poultry farms. This model provides guidance on how farm management and the establishement of different strategies in each production step can regulate gas emissions from a particular farm.

3.2. Model Formulation

Funded by the BATFARM Interreg-Atlantic Area Project (2009-1/071), the model has been developed specifically by authors from Portugal, Spain, France, Scotland and Ireland but it could be adapted to other climatic conditions and farm practices. Particularly, modifiable default values have been included to develop a versatile and user-friendly model thus, different default values can been set for zootechnical data, climatic information and emission factors depending on the country selected (Portugal, Spain, France, Scotland or Ireland), we refer to them as "regionalizable default values". A total of 77 climatic areas have been defined for the five countries with modifiable default values of temperature, relative humidity, precipitation and wind speed obtained from historical data series (20 years or more). Model database, accounting for more than 4000 parameters, can be classified as primary and intermediate sources of data. Primary data or basic input data are taken from literature, databases and expert consultation or has to be entered. Intermediate data are the output of model calculations which are the basis for further calculations; in certain cases these can be also modified.

The model calculates whole farm emissions allowing for the simulation of the effect of diverse abatement techniques (Table 1) by comparing different scenarios.

Table 1. Abatement techniques considered by the model at different stages of livestock production.

ABATEMENT TECHNIQUES					
NUTRITIONAL MANAGEMENT		√	√	√	√
INSTALLATIONS	Type of floor/bedding charact.	√	√	√	√
	Frequent manure removal	√	√	√	√
	Ventilation/Temperature	√	√	√	√
	Other	√	√	√	√
STORAGE (SLURRY)	Covers	√	---	---	√
	Additives	√	---	---	√
TREATMENT	Separation	√	---	---	√
	Aerobic	√	---	---	√
	Anaerobic	√	√	√	√
	Composting	√	√	√	√
LANDSPREADING	Equipment	√	---	---	√
	Incorporation	√	√	√	√

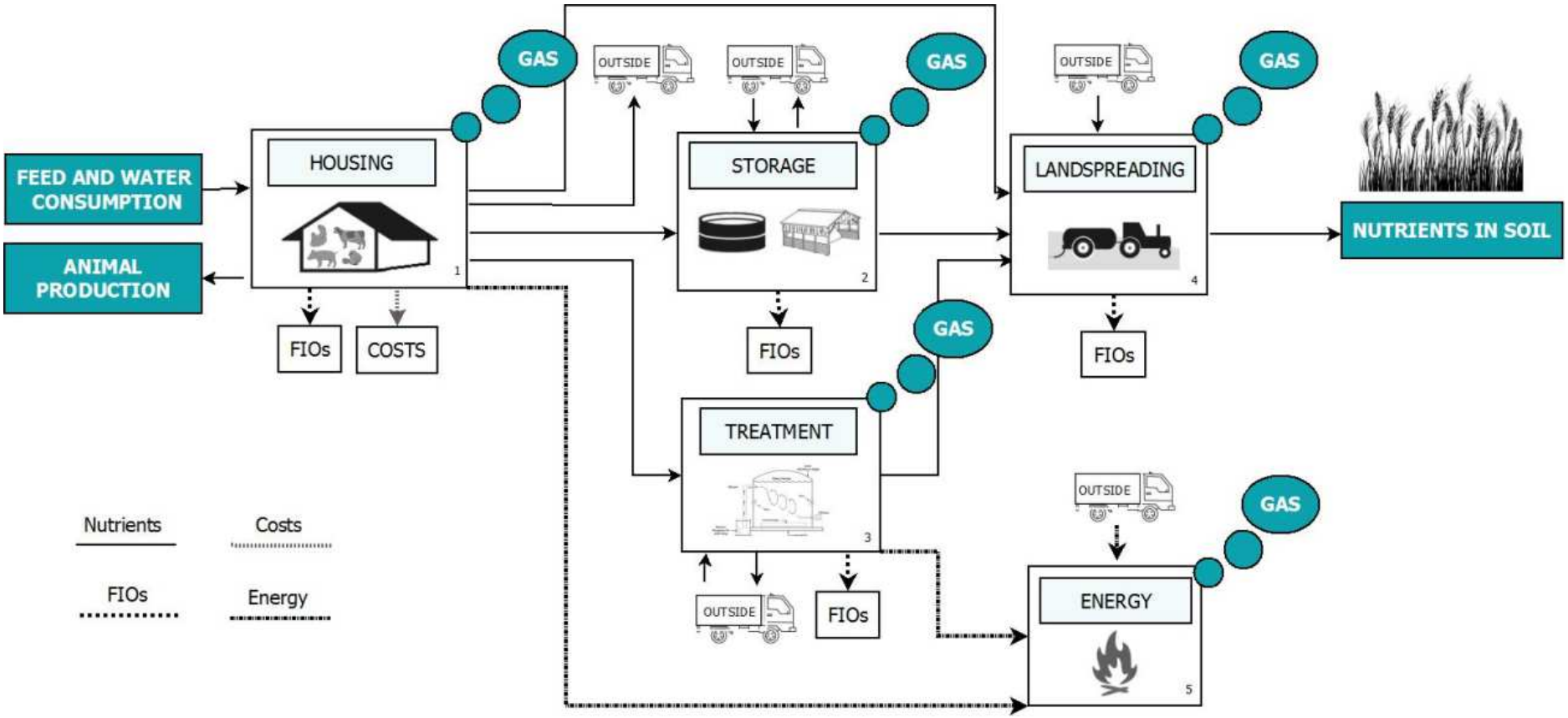
√: Techniques considered in the model

The model was set up using a combination of methodologies including emission factors, empirical equations and process-oriented mechanisms. It comprises 8 main subroutines:

- 4 Housing subroutines, one for each type of animal considered: swine, broilers, laying hens and dairy cattle.
- 2 Outside storage subroutines: one for liquid and one for solid manures.
- 1 Treatment subroutine.
- 1 Landspreading subroutine.

The general structure of the model and the interconnection between the different subroutines is shown in Figure 1.

Fig. 1. General model structure and subroutines. Numbers indicates subroutines sequence.



A mass balance is carried out in each subroutine in order to estimate emissions, consumptions and animal production. Outputs from one stage are linked as inputs to the following stage. These calculations are performed on a cumulative monthly and annual basis.

An assessment on faecal indicator organisms (FIOs) is provided for each subroutine, using a 3-tier qualitative scale. With regard to information related to the economical feasibility of implementing the selected techniques, users have the option of introducing parameters to calculate the cost of the housing abatement techniques selected according to the EC (2003) equation. An additional subroutine evaluating the farm direct energy consumption is also available.

The following sections provide further detailed descriptions of the approach considered for the different stages of livestock production methods. Supplementary data (equations and default values) can be found in the Appendixes.

3.2.1. Housing subroutines

Animal excretion at housing is calculated considering data related to nutritional management and animal performance. Gaseous emissions (*E*) are calculated with an emission factor (*EF*), weighted by the product of several variation factors *(VF*) according to building design and manure management. Outputs in the form of gaseous emissions, animal production, feed and water consumption and manure are transferred to the next subroutine (Figure 2).

Gaseous emissions at housing are calculated using Equation 1 based on Rigolot, Espagnol, Pomar, Hassouna, et al. (2010), with some modifications: additional *VF* have been identified and their effects on *EF* have been reviewed and adapted to reflect a wider range of abatement techniques and other species different from swine. Different *EF* have been searched in literature for the various animals and can have different values for each of the 5 countries considered by the model (Table 2).

$$E_x = EF \times \prod_{i=1}^{n} VF_i \quad (1)$$

Fig. 2. Schematic representation of housing subroutines.

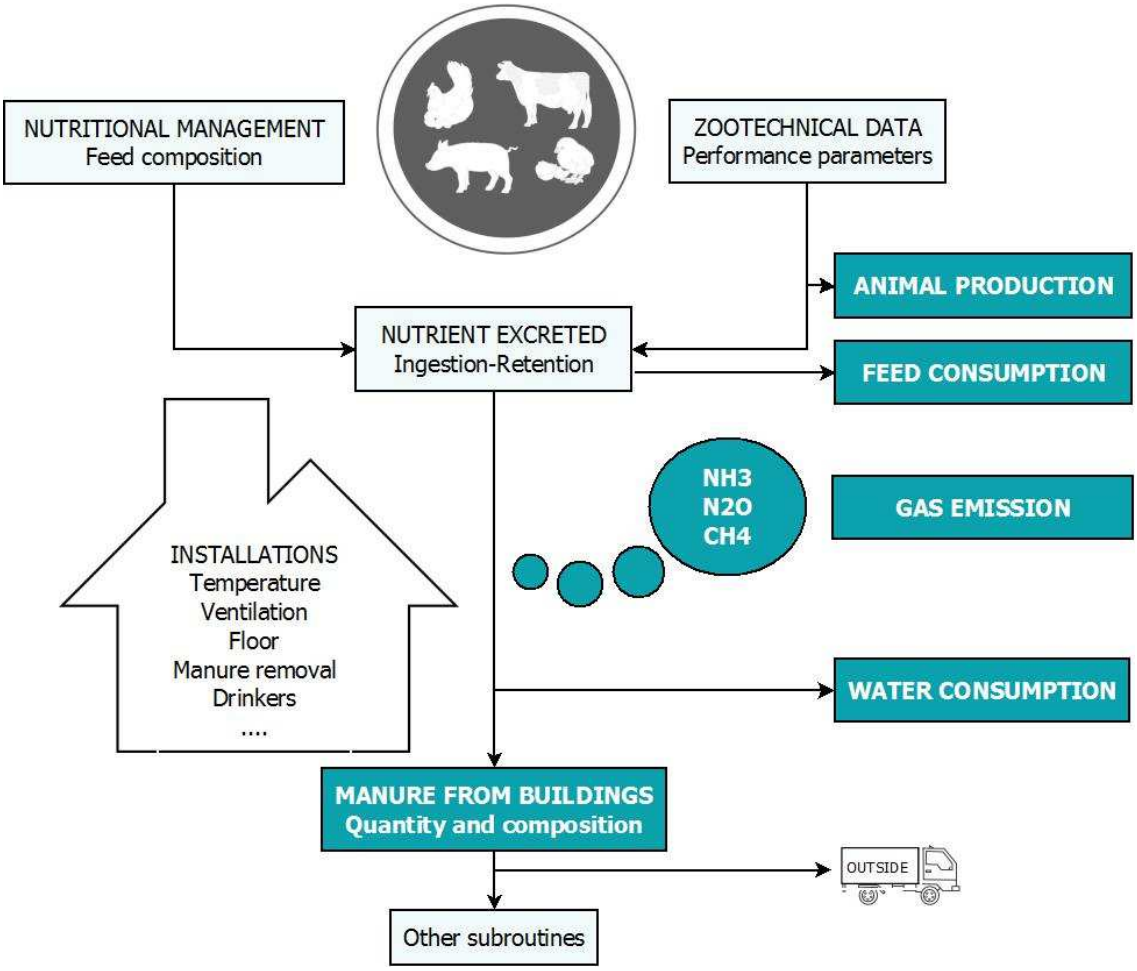


Table 2. Regionalizable default values for EF utilised in the housing subroutines*.

HOUSING SUBROUTINE		EF_{NH3} (kg NH_3-N kg $N_{excreted}^{-1}$)	Source	EF_{N2O} (kg N_2O-N kg $N_{excreted}^{-1}$)	Source
	*Prefattening* *Fattening* *Gestating sows & gilts* *Lactating sows*	0.24	Dourmad, Guingand, Latimier, and Sève (1999)	0.002	IPCC (2006)
	*Rearing chickens* *Laying hens*	0.6	Corpen (2006); Fournel, Pelletier, Godbout, Lagacé, and Feddes (2012); Patterson & Adrizal (2005)	0.00212	EC (2013)
	*Broilers*	0.30	Corpen (2006); Patterson & Adrizal (2005)	0.03	EC (2013)
	*Cows* *Calves* *Heifers*	0.14	Hristov et al. (2011)	0 – 0.01	IPCC (2006)

*Methane emissions follow Tier 2 methodology (IPPC, 2006).

3.2.1.1. Swine housing subroutine

Model Input data

Table 3 shows data required for simulating housing at swine facilities which will vary depending on the animal housed (gestating sows, farrowing sows, prefattened and fattened pigs).

Table 3. Required input parameters for the swine housing subroutine, together with the regionalizable default values (Source: INTIA).

MODEL INPUTS	VALUE	MODEL INPUTS	VALUE
Country and climatic region	Data to select	Number of piglets, piglets per sow and year	23
Type of facility (type of animal housed and produced)	Data to select	Age weaned piglets, days	21
Type of floor	Data to select	Number litters, litter per sow per year	2.25
Number of sows housed	Data to enter	First service, months	2
Number of places pre-fattening	Data to enter	Initial weight pre-fattening / fattening pigs, kg	5 / 22
Number of places fattening pigs	Data to enter	Final weight pre-fattening / fattening pigs, kg	22 / 110
Type of feeding fattening pigs	Data to select	Mortality pre-fattening / fattening pigs, %	3.5 / 4
Feed composition	Default values	Feed conversion rate pre-fattening / fattening pigs	1.65/2.8
Slurry removal frequency	Data to select	Daily weight gain pre-fattening / fattening pigs, g	350 / 716
Type of drinker	Data to select	Period between batches pre-fattening / fattening pigs, days	7 / 7
Other Best Management Practices	Data to select	Maximum ventilation rate, (gestating / lactating / prefattening / fattening), $m^3/h/kg$	0.9 / 0.95 / 1.28 / 0.9
Feed intake sows (gestating / lactating / gilts), kg/day	2.72 / 4.6 / 2.72	Slurry removal and destination	Data to enter
Gilt replacement, %	45		

Calculations

Animal production at facilities is calculated according to Equations A1 to A13 (Appendix A). The number of gestating, lactating and replacement sows and the length of each of these periods over the year per sow are calculated. This is based on the number of sows housed and the default values for the age of the weaned piglets, the number of litters

per sow per year, the gilt replacement rate and the period until the first service (Equations A1 to A6, Appendix A). Piglets produced at the facility are estimated by the number of sows and the number of weaned piglets throughout the year (Equation A7, Appendix A). Prefattened and fattened pig population is estimated according to allocated number of places, pig productivity and the mortality index at both stages (Equation A9 and A12, Appendix A). This model assumes that pigs are sold out of farm when animal productivity exceeds farm capacity for prefattening and/or fattening (Equation A8 and A10, Appendix A).

Animal nutrient balance (N, P, Cu and Zn) at pig facilities is calculated as:

$$Nutrient\ excreted = Nutrient\ intake - Nutrient\ retention \quad (2)$$

The exception being K excreted, which is directly calculated based on Corpen (2003) considering the protein content of the feed (Table A1, Appendix A).

Nutrient consumption at the facilities is estimated based on feed consumption for each stage of the pigs life (Equation A14 and A15, Appendix A) together with crude protein (CP), P, Cu and Zn levels of the corresponding concentrates. Default values of concentrates composition are those collected by INTIA after extensive sampling in different pig production systems in Navarre and compared with values reported across different European countries (EC, 2013). The important aspect of pig CP nutrition considered by this model is the effect on NH_3 and N_2O emmisions at the facilities. The standard CP content of concentrates offered to gilt and lactating sows is set at 16.5%, whereas gestating sows are fed with 13.5% CP. Piglet nutrition, which is split into pre-starter and starter phases, assumes that the concentrates used average 20.5 and 18.5% CP, respectively. In relation to fattening pigs, the default CP content of feeds provided to the growing and finishing phases are 17.0 and 16.0%, respectively. Nonetheless, the model is shaped to consider different nutritional strategies for fattening pigs. Firstly, growing and finishing pigs may be fed with biphase, triphase and multiphase rations. If any of these options are applied on-farm, the model reduces CP concentration to 16.0 and 14.5% for growers and finishers, respectively. Additionally in the model feeding fattening pigs with amino acids and phytases is allowed, if pigs are supplemented with amino acids then the model assumes that ration CP content is 15.0% for growers and 13.5% for finishers. In relation to P nutrition, the content of P may be reduced from 0.55% in default concentrates to 0.4% in phytases enriched concentrates. Nutrient retention values are taken

from Corpen (2003), Guillou, Dourmad, and Noblet (1993) and INRA (1984) (Table A2, Appendix A).

Calculation of the amount of organic matter and dry matter excreted by the animals are calculated from feed intake and feed digestibility coefficients according to Le Goff and Noblet (2001) in Rigolot, Espagnol, Pomar and Dourmad (2010) (Equation A16 and A17, Appendix A).

Water consumption at pig housings is the combination of drinking and water wasted by the animals, cleaning water and the water consumed due to technical BAT implemented (i.e. wet scrubbers increases farm water consumption). The model considers an average sow drinks 6 l per feed kg (Abaigar, Íñigo, Cordovín, 2005) and 260 l of washing water is used per litter (Latimier, Gallard, Corlouër, 1996). For prefattening and fattening pigs, the water consumed is calculated considering feeding dilution, together with the indoor temperatures at the facilities (Collin, Van Milgen, Le Dividich, 2001; Massabie, 2001) (Figure A1 and A2, Table A3, Appendix A) and the protein content of the feed based on Albar and Granier (1996) and Shaw, Beaulieu, and Patience (2006) (Table A4, Appendix A). The wasted water is related to the type of drinkers used and has been established according to Chosson, Granier, Maigne, Bouby, and Mongin (1998) and Massabie (2001) (Table A5, Appendix A). Finally, the amount of cleaning water per animal produced is based on Latimier, Gallard, and Corlouër (1996) (Table A6, Appendix A).

Regarding slurry production at facilities, different approaches are considered by the model. For sows, default slurry production values are used (16.1 l per gestating sow per day, 20.2 l per farrowing sow per day and 14 l per gilt per day) from Latimier et al. (1996) and Levasseur (1998). On the contrary, slurry production in fattening and prefattening pigs is defined as the addition of animals' excreta, wasted water and cleaning water. The model considers that 60% of the drinking water is excreted by the animals in accordance to Fiedler (1982).

Gaseous losses (NH_3, N_2O and CH_4) from pig facilities are affected by various aspects of building design and/or slurry management such as the type of floor, slurry removal frequency or the installed BAT (i.e. wet scrubbers, fogging coolers, etc). The model considers these parameters together with climatic and ventilation conditions at the housing stage to calculate the variation factors (*VF*) used in the gas emission equations.

The ammonia emission equation (E_{NH3}, kg NH_3) is based on Rigolot, Espagnol, Pomar, Hassouna, et al. (2010):

$$E_{NH3} = 17 \times 14^{-1} \times EF_{NH3} \times N_{ex} \times VF_{Ndil} \times VF_{Temp} \times VF_{Vent} \times VF_{NH3floor} \times VF_{NH3freq} \times VF_{NH3Other} \quad (3)$$

There are a number of variation factors (VF) which may modulate the default EF_{NH3} (Table 2) expressed in kg NH_3-N per kg N excreted (N_{ex}). VF_{Ndil} represents the effect of N dilution in the slurry on NH_3 emissions, and it is estimated according to Equation A18 (Appendix A) (VF_{Ndil}=1 when slurry N content is 0.51 mol l^{-1}). The effect of slurry temperature on the volatilization rate (VF_{Temp}) is estimated using Equation A19 (Appendix A) and takes into account the average indoor temperatures (Table A7, Appendix A) and Granier, Guingand, and Massabie (1996) investigations (VF_{Temp}=1 when slurry temperature is 22ºC). The air ventilation rate (VF_{Vent}) is calculated according to Equation A20 (Appendix A) and considers the average ventilation rates recorded at swine farms in the region (Table A8, Appendix A) (VF_{Vent}=1, when ventilation rate is 0.6 m^3 h^{-1} kg^{-1} LW). $VF_{NH3floor}$ will be 1 when 100% of the floor surface is fully slatted using concrete slats (Equation A21, Table A9, Appendix A). The model considers that when the slurry removal frequency exceeds one month there will be no reduction on NH_3 emission ($VF_{NH3freq}$=1), more frequent slurry removal will achieved NH_3 abatement (Equation A22, Table A10, Appendix A). Finally the effect of other mitigation techniques on NH_3 emission ($VF_{NH3Other}$) is shown in Table A11 and Equation A23 (Appendix A).

Nitrous oxide emission (E_{N2O}, kg) is estimated according to the following equation:

$$E_{N2O} = 44 \times 28^{-1} \times EF_{N2O} \times N_{ex} \times VF_{N2Ofloor} \times VF_{N2Ofreq} \times VF_{N2OOther} \quad (4)$$

The model considers that overall 0.2% of excreted N is lost as N_2O-N (EF_{N2O}, Table 2). Additionally, in accordance with the VF values reported in Tables A9, A10 and A11 for N_2O losses (Appendix A), the effect of the type of floor ($VF_{N2Ofloor}$), frequency of slurry removal ($VF_{N2Ofreq}$) and other techniques implemented in animal houses ($VF_{N2OOther}$) are included in the estimation. The procedure to calculate these variation factors is similar to the ones described previously for the NH_3 (Equations A21, A22 and A23, Appendix A).

Concerning CH_4 losses at pig facilities, the model accounts for animal CH_4 emission and manure derived CH_4 loss. Methane production at pig level (E_{CH4Ent}, kg) is estimated using Tier 2 methods (Equation A24, Appendix A) reported by IPCC (2006). This formula integrates the gross energy (GE, MJ) consumed by pigs and the CH_4 conversion factor for

each animal stage (Ym, %) (Table A12, Appendix A). Gross energy consumption is calculated based on the digestible energy content in the feed, feed consumption and organic matter digestibility coefficient calculated according to Le Goff and Noblet (2001) (Equation A25, Appendix A). Emission of CH_4 from manure management (E_{CH4Man}, kg) is calculated according to Equation 5, considering Tier 2 methodology (IPCC, 2006), which is modified by a number of *VF* that modulate CH_4 losses.

$$E_{CH4Man} = VS_{ex} \times Bo \times 0.67 \times (MCF_{>1} \times 100^{-1}) \times VF_{CH4floor} \times VF_{CH4freq} \times VF_{CH4Other} \quad (5)$$

VS_{ex} is the amount of organic matter excreted by each animal stage (kg). *Bo* is the maximum CH_4 producing capacity attributed to pig slurry. The IPCC (2006) guideline assumes that *Bo* value is 0.45 m^3 CH_4 kg VS^{-1} for all types of swine. *MCF>1* represents the CH_4 conversion factor (%) for deep pits after a slurry accumulation period longer than 1 month (IPCC, 2006). These values are affected by the average monthly temperatures of the simulated region. When the slurry removal frequency is less than one month, $VF_{CH4freq}$ is calculated considering the IPCC (2006) values for <1 month storage and the mean monthly temperature in each region (Equations A26 and A22, Appendix A). The procedure to calculate the effect of the type of floor ($VF_{CH4floor}$) and other BAT implemented in animal houses ($VF_{CH4Other}$) is similar to the ones described previously for the NH_3 and N_2O (Equations A21 and A23, Tables A9 and A11, Appendix A).

3.2.1.2. Laying hens housing subroutine

Model Input data

The model can simulate emissions from a farm with or without rearing chickens. Up to two different installations for laying hens and one of rearing chicken can be simulated in the model. Table 4 lists the input data required.

Calculations

Poultry production. Quantity of eggs produced in the farm is calculated considering the number of laying days per year and other zootechnical data (mortality, population and laying rates) (Equations B1, B2 and B3, Appendix B).

Poultry nutrient balance (N, P, K, Cu, and Zn) is calculated according to Equation 2. Feed consumption is calculated considering the total live weight, the quantity of eggs

produced in the farm and the corresponding feed conversion rate (FCR) (Equation B5 and B6, Appendix B). Three types of feeds, with different composition, are considered during the production cycle. Moreover, various feed strategies can be simulated: standard (16-17% CP; 0.5-0.7% phosphorus), adjusted (15-17% CP; 0.45-0.7% phosphorus), with synthetic amino acids (14-16% CP) and/or with phytases (0.4-0.6% phosphorus). Nutrient retention is determined taking into account the nutrient content per kg of egg produced and bird weight increase (Table B1, Appendix B). The model utilises the quantities of organic matter excreted per bird per day shown in Table B2 (Appendix B).

Table 4. Required input parameters for the laying hen housing subroutine, together with the regionalizable default values (Source: INTIA).

MODEL INPUTS	VALUE	MODEL INPUTS	VALUE
Country and climatic region	Data to select	Number of rearing weeks	17
Type of facility (with/without rearing)	Data to select	Average final weight rearing chicken (kg)	1.52
Number of weeks per laying cycle	56	FCR (kg feed per kg reared chicken)	6
Mean laying rate (%)	90	Type of feeding	Data to select
Mean egg weight (g)	54	Feed composition	Default values
FCR (kg feed per kg egg)	2.2	Number of animals	Data to enter
Number of weeks before laying	3	Manure management	Data to select
Laying rate after pre-laying period (%)	5	Manure dry matter content (%)	Default values
Mortality during laying (%)	10	Other Best Management Practices	Data to enter
Number of weeks between batches	3	Building surface (m^2)	Data to enter
Average weight of culled hens (kg)	2	Manure removal and destination	Data to enter

Water consumption is the addition of water consumed by the animals, defined as 1.9 l kg^{-1} feed (EC, 2013), cleaning water, 0.045 and 0.01 m^3 m^{-2} and laying cycle for pits and for belts systems respectively (EC, 2013) and the water consumption due to the abatement techniques implemented.

Regarding manure production at facilities, a variety of manure removal systems and frequencies are offered by the model, including manure belt with/without drying, mid deep and deep pit systems (Table B3, Appendix B). Their effect on gaseous emissions is taken from EC (2003) and MARM (2010b). The manure produced is calculated considering the quantity of dry matter removed from the buildings and the dry matter content of the manure according to the system selected (Equations E13, E10 and Table E2, Appendix E).

Gaseous losses (NH_3, N_2O and CH_4)

Ammonia emission (E_{NH3}, kg) is calculated according to Equation 6.

$$E_{NH3} = 17 \times 14^{-1} \times EF_{NH3} \times N_{ex} \times VF_{NH3Rem} \times VF_{Vent} \times VF_{NH3Other} \quad (6)$$

Where EF_{NH3} is 0.6 kg NH_3-N per kg nitrogen excreted (N_{ex}) in deep pit systems reported by Corpen (2006), Fournel et al. (2012) and Patterson and Adrizal (2005) (Table 2). This potential NH_3 emission is moderated by certain variation factors (VF) according to the manure removal system selected (VF_{NH3Rem}), the ventilation rates in the buildings (VF_{Vent}) and other mitigation strategies in the houses ($VF_{NH3Other}$) that can be defined. VF_{Vent} simulates the effect of ventilations rates, and therefore indoor temperatures, on NH_3 emissions using equations and limits based on Fabbri, Valli, Guarino, Costa and Mazzotta (2007) and Alberdi, Arriaga, Calvet, Estellés, and Merino (2016) (Table B4, Appendix B). Average monthly ventilation rates for each installation will vary according to the stage (laying hens or rearing chicken) and type of manure removal system (Table B5, Appendix B). VF_{NH3Rem} and $VF_{NH3Other}$ are calculated according to Equations B7 and A23, respectively (Appendices B and A).

Nitrous oxide emission (E_{N2O}, kg) is calculated based on 2.12 g of N_2O-N emitted per kg of N excreted, this value is calculated from the volatilization data for laying hens cage systems from EC (2013) (Table 2). The emission is also influenced by the manure removal system selected (VF_{N2ORem}) and other mitigation strategies implemented in animal houses ($VF_{N2OOther}$) (Equation 7).

$$E_{N2O} = 44 \times 28^{-1} \times EF_{N2O} \times N_{ex} \times VF_{N2ORem} \times VF_{N2OOther} \quad (7)$$

Rates of CH_4 emission from laying hens is due to the manure management approach (E_{CH4Man}, kg CH_4) and it is calculated according to Equation 8, considering Tier 2 methods (IPCC, 2006).

$$E_{CH4Man} = VS_{ex} \times Bo \times 0.67 \times (MCF \times 100^{-1}) \times VF_{CH4Rem} \times VF_{CH4Other} \quad (8)$$

Where VS_{ex} is the organic matter excreted (kg), Bo is the maximum CH_4 producing capacity (0.39 m^3 CH_4 kg^{-1} VS from IPCC (2006)), MCF is the methane conversion factor (1.5% from IPCC (2006)). The manure removal system selected (VF_{CH4Rem}) and other mitigation strategies in poultry houses ($VF_{CH4Other}$) are also considered.

3.2.1.3. Broilers housing subroutine

Model Input data

The model distinguishes among four types of broilers based on the final weight (from 1-2.4 kg). Depending on the category selected, different default values are available (Table 5).

Table 5. Required input parameters for the broilers housing subroutine, together with the regionalizable default values (Source: INTIA).

MODEL INPUTS	VALUE			
Broiler Type	Light chicken 1	Light chicken 2	Standard lightweight	Standard heavyweight
Final weight (kg)	1	1.6	1.8	2.4
Cycles per year	7	7	6	5.5
Animals per m²	28	23	20	15
% Mortality	3	3	4	5
FCR (kg feed per kg body weight)	1.7	1.7	1.8	1.9
Growing weeks per cycle	Calculated by the model*			
Surface	Data to enter			
Type of feeding	Data to select			
Bedding material	Data to select			
Type of drinkers	Data to select			
Other Best Management Practices	Data to enter			
Manure removal and destination	Data to enter			

*Modifiable

Calculations

The model calculates poultry production (kg of live weight produced) based on zootechnical data (Equation C1, Appendix C).

Animal nutrient balance (N, P, Cu, and Zn) is calculated according Equation 2, with the exception of K. In this case the model assumes 13.7 g of K excreted per kg of live weight produced. For the rest of nutrients, consumption is calculated based on feed intake and composition. Feed consumption is calculated from the total live weight produced at the farm and the feed conversion rate (FCR). The model considers up to four types of feeds, with different composition, during the production cycle. Moreover, each feed composition will depend on the type of feeding selected: standard (20.1-22.1% CP; 0.68-

0.76% phosphorus), adjusted (18-21% CP; 0.60-0.70% phosphorus), with synthetic amino acids (17-19% CP) and/or with phytases (0.50-0.60% phosphorus). The nutrient retention is determined taking into account the quantity of live weight produced (Corpen, 2013) (Table C1, Appendix C).

An adaptation of the quantities of organic matter excreted per animal per day suggested by MARM (2008) has been carried out in order to reflect the four broilers categories considered by the model (Table C2, Appendix C).

Total water consumption is the summation of the water consumed by the broilers, set at 1.8 l kg^{-1} feed (EC, 2013), plus cleaning water usage (0.0065 m^3 m^{-2}) and production cycle (EC, 2013) and water consumption due to mitigation strategies implemented.

Regarding manure production at facilities, the type of bedding material (straw, rice husks or sawdust) needs to be indicated. This value will be used to adjust the reference quantity of manure produced using the method of Corpen (2006) (described in the Appendix C).

Gaseous losses (NH_3, N_2O and CH_4)

Ammonia emission (E_{NH3}, kg) is calculated according to Equation 9.

$$E_{NH3} = 17 \times 14^{-1} \times EF_{NH3} \times N_{ex} \times VF_{NH3drink} \times VF_{NH3thick} \times VF_{NH3bed} \times VF_{InTemp} \times VF_{NH3Other} \quad (9)$$

Where EF_{NH3} is 0.30 kg NH_3-N volatilized per kg nitrogen excreted (N_{ex}) calculated based on Corpen (2006) and Patterson and Adrizal (2005) (Table 2). The potential NH_3 emission is moderated by certain variation factors: type of drinkers ($VF_{NH3drink}$), bedding thickness ($VF_{NH3thick}$), type of bedding material (VF_{NH3bed}), indoor temperature (VF_{InTemp}) and other mitigation strategies implemented in the houses ($VF_{NH3Other}$). $VF_{NH3drink}$ is calculated to reflect the percentage reduction in NH_3 emissions associated to the type of drinker (Equation C2, Table C5, Appendix C). Four different types of drinkers are considered (nipple drinkers, drinkers with a drip-cup, water troughs and round drinkers). A 40% reduction in NH_3 emission is applied to the default value if drinkers with a drip-cup are selected in comparison to standard nipple drinkers, according to Da Borso and Chiumenti (1999). On the contrary, NH_3 emission increases by 30% when either water throughs or round drinkers are used, this is based on the results obtained by Da Borso and Chiumenti (1999) and Nicholson, Chambers, and Walker (2004). Gas emission might vary depending on the type (VF_{NH3bed}) and amount of bedding ($VF_{NH3thick}$). The model calculates the

bedding thickness, evulating the quantity and the density of the material used. If the thickness is over 4.5 cm then an NH_3 emission reduction of 27% is applied (Al Homidan, Robertson, Petchey, 1997) (Equation C3, Appendix C). Per default, the model considers that the type of material has no effect on emission (Elwinger & Svensson, 1996; Nicholson et al., 2004), however this can be modified (Equation C4, Appendix C). VF_{InTemp} simulates the indoor temperature effect on NH_3 emissions using the equations based on Calvet, Cambra-López, Estellés and Torres (2011) (Tables C6 and C7, Appendix C). Other mitigation techniques can be simulated at housing ($VF_{NH3Other}$) (Equation A23, Appendix A), e.g. fogging coolers, with a 26% reduction in NH_3 emissions (Institut de L'Elevage et al., 2010). Additionally, up to 2 new techniques can be simulated as long as gas emission reduction percentages are known.

Nitrous oxide emission (E_{N2O}, kg) is calculated based on an EF_{N2O} of 30 g of N_2O-N emitted per kg of N excreted (N_{ex}). This emission factor has been calculated from the information given in EC (2013) (Table 2). This value is modulated considering certain variation factors: type of drinkers ($VF_{N2Odrink}$), bedding thickness ($VF_{N2Othick}$), type of bedding material (VF_{N2Obed}) and other mitigation techniques implemented in houses ($VF_{N2OOther}$) (Equation 10).

$$E_{N2O} = 44 \times 28^{-1} \times EF_{N2O} \times N_{ex} \times VF_{N2Odrink} \times VF_{N2Othick} \times VF_{N2Obed} \times VF_{N2OOther} \quad (10)$$

The CH_4 emission from broilers is due to manure management (E_{CH4Man}, kg CH_4) and it is calculated according to Equation 11, considering Tier 2 methods (IPCC, 2006) and certain variation factors.

$$E_{CH4Man} = VS_{ex} \times Bo \times 0.67 \times (MCF \times 100^{-1}) \times VF_{CH4drink} \times VF_{CH4thick} \times VF_{CH4bed} \times VF_{CH4Other} \quad (11)$$

Where VS_{ex} is the organic matter excreted (kg), Bo is the maximum CH_4 producing capacity (0.36 m^3 CH_4 kg VS^{-1}, IPCC (2006)), MCF is the methane conversion factor (1.5%, (IPCC, 2006)). The type of drinkers ($VF_{CH4drink}$), bedding thickness ($VF_{CH4thick}$), the type of bedding material (VF_{CH4bed}) and other mitigation techniques implemented in houses ($VF_{CH4Other}$) can also affect CH_4 emissions.

3.2.1.4. Dairy housing subroutine

Model Input data

Table 6 summarizes the dataset required for the housing simulation model at dairy facilities. Eight types of facilities are considered for adult cows whereas six stall systems are allocated to young cattle (tie-stall systems are excluded): 6 freestall systems (with cubicles; with cubicles and sloped concrete floor; with deep litter and mechanical removal; with deep litter and slatted floor; with deep litter and yard; and with deep litter whole farm) and 2 tie-stall systems (with a gutter system, and slatted floor).

Table 6. Required input parameters for the dairy cattle housing subroutine, together with the regionalizable default values (Source: INTIA).

MODEL INPUTS	VALUE	MODEL INPUTS	VALUE
Country and climatic region	Data to select	Livestock management (grazing, confinement)	Data to select
Type of facility	Data to select	Grazing period milking cows (% of the day)	Data to enter
Type of floor	Data to select	Grazing area (ha)	Data to enter
Number of adult cows	Data to enter	Manure removal system	Data to select
Number of calves (< 1 year)	Data to enter	Manure removal frequency	Data to select
Number of heifers (< 2 years)	Data to enter	Type of milking parlor	Data to select
Milk yield (l per cow and year)	Data to enter	Wash water management	Data to enter
Milking/Dry period	Data to enter	FIO assessment	Data to select
Type of bedding	Data to select	Other Best Management Practices	Data to select
Amount of bedding (t per year)	Data to enter	Weight calves at birth, kg	40
Milking cow nutrition	Data to enter	Weight calves 1 year, kg	325
Calf nutrition	Data to enter	Weight calves 2 year, kg	475
Heifer nutrition	Data to enter	Manure removal and destination	Data to enter

Calculations

Animal production. Milk yield at farm level is calculated taking into account the mean annual milk production of the cows and the mean number of milking cows in the herd (Equation D1, Appendix D). The number of milking cows is estimated according to Equations D2 and D3 (Appendix D).

Animal nutrient balance. Nutrient balance (N, P, K, Cu, Zn) at dairy facilities is calculated according to Equation 2. Nutrient intake by dairy cattle is estimated monthly based on the default values reported by Corpen (1999) for milking or dry cows and young cattle (Equations D4 to D7, Appendix D). Nutrient intake in the dairy herds is related to grass silage/hay, maize silage or grazing based rations. The ingestion level of milking cows, with the default values allocated to herds producing 6000 kg per cow and year, is corrected depending on the production level of the herd (increase of 5 to 12 % when milk yield arises every 1000 kg per cow per year). In addition, users are allowed to introduce the daily dry matter intake (DMI) and nutrient content of the total mixed rations (TMR) offered to the milking cows. The model assumes that milking cows are fed with TMR whose CP, P, K, Cu and Zn concentrations are 17.0%, 0.4%, 1.0%, 10 mg kg^{-1} and 45 mg kg^{-1}, respectively. These values may be modified within concentration limits established from data collected by regional extension services at commercial dairy farms. Two nutritional strategies may fit the nutrional intake of milking herds without compromising milk production: the computerised individual feeding system (CIFS) and phase-feeding system (PFS). The model considers that 2.0 kg concentrate per cow per day is saved through the individual feeding system (expert criteria). This value is close to the amount which is estimated to be saved in previous studies using CIFS (Grant & Bodman, 1995; Khalili, Mäntysaari, Sariola, Kangasniemi, 2006). The model considers the following nutrient concentration of the concentrate when CIFS is applied at dairy farms: 20.1% CP, 0.53% P, 0.10% K, 63 mg kg^{-1} Zn and 7.50 mg kg^{-1} Cu (Arriaga, 2010). Dividing cows into different feeding groups may contribute to matching feed nutrient concentrations with nutrient requirements (Jonker, Kohn, High, 2002). The model considers that splitting a TMR consuming milking herd (milk yield > 8,000 kg per cow per year) according to the lactation phase allows a reduction of 1.25 kg DM per cow per day during the late lactation stage (Arriaga, 2010). Nutrients saved by CIFS and PFS systems are estimated according to Equation D8 and D9 (Appendix D).

Nutrient retention (N, P, K, Cu, Zn) values in milk and meat produced are taken from Corpen (1999). The model sets the default weights at birth and for cattle at 1 year and 2 years old at 40, 325 and 475 kg, respectively. Nutrient retention for adult and young cattle is provided by Equations D10 to D12.

The organic matter (OM) excretion allocated for each cattle category is obtained from the Spanish National Methodology for Estimating Gaseous Losses from Livestock

Sector (MAGRAMA, 2013). According to this guidebook, the model estimates that on average an adult cow (milking and dry) excrete 4.7 kg OM per cow per day; calves excrete 1.1 kg OM per animal per day and heifers produce 2.4 kg OM per animal per day.

Water consumption. As Equation D13 shows, the model estimates the water balance of dairy facilities taking into account the following parameters: animal consumption, water management at milking parlors, water used for barn cleaning and water consumption/abatement through the technical solutions implemented at the farm for reducing the gaseous losses. Cattle water intake assumes that 90, 45, 25 and 35 l per animal per day is drunk by milking cows, dry cows, calves and heifers, respectively (Hernández-Benedí, 1984; NRC, 2001; Ward & McKague, 2007). Up to two milking parlors may be considered in the model and their water consumption, which integrates either wash water or grey water for cleaning the equipment and the parlor, is estimated according to values reported by Institut de l´Élevage et al (2001) (Table D1, Appendix D). The model lets users indicate the percentage of wash water re-utilized as grey water for cleaning the installation. The amount of water used for barn cleaning is assumed to be equivalent to the slurry produced by cattle (Rotz, 2004). This author stated that compared to traditional scraped or slatted floor systems, the amount of manure produced in barn systems is roughly doubled through the inclusion of the flushing water.

Manure production. Slurry and/or solid manure production related to each cattle stage is obtained from data compiled by the Institut de l´Élevage et al (2001) (Table D2 to D4, Appendix D). In addition to the default slurry production, the model assumes that the wastewater from milking parlors and the hose-water used for flushing activities are conveyed to the slurry pit. Similarly, if any technological practice (e.g. milk tank heat recovery, milk pre-cooling systems) produces variations in water consumption then the model considers that slurry production is also altered. Concerning solid manure production, default amounts include some bedding material (sand, straw, sawdust or wood shavings, rice hulls) as Tables D5 and D6 show (Appendix D). The model permits the introduction of extra bedding material, which is added to the overall solid manure calculation. Equation D14 indicates how slurry production is calculated by the model. Table D7 (Appendix D) lists the nutrient composition of the different bedding materials.

Gaseous losses. Two main types of gaseous sources are described for dairy cattle farms: CH_4 losses from enteric fermentation and NH_3, CH_4 and N_2O emissions from manure management. Enteric CH_4 loss from milking herds confined within dairy barns is a

parameter which depends on the DMI (Ellis et al., 2007). Enteric CH_4 losses from other confined herds or grazing cattle are estimated using default values (MARM, 2008; Vermorel et al., 2008), the model assumes that 255, 57 and 119 g CH_4 per animal per day are emitted by dry cows, calves and heifers, respectively.

Ammonia emission from slurry (E_{NH3}, kg) is calculated by adapting the equation developed by Rigolot, Espagnol, Pomar, Hassouna, et al. (2010) to dairy housings:

$$E_{NH3} = 17 \times 14^{-1} \times EF_{NH3} \times N_{ex}\ x\ VF_{stall}\ x\ VF_{Temp} \times VF_{NH3freq} \times VF_{NH3Rem} x\ VF_{NH3Other} \qquad (12)$$

As previously described for the rest of the livestock sector, different *VF* will moderate the NH_3 volatilization rate at dairy facilities. The mean NH_3 emission from slurry-based systems is affected by the type of the facility (freestall *vs* tie-stall, concrete floor *vs* slatted floor) ($VF_{NH3Stall}$), the slurry temperature (VF_{Temp}), the frequency of manure removal ($VF_{NH3freq}$), the type of manure removal system (VF_{NH3Rem}) and other mitigation techniques implemented ($VF_{NH3Other}$). A default value of 0.14 is set as the mean NH_3 emission factor (EF_{NH3}, kg NH_3-N per kg N excreted) for free-stall barns with concrete floor, cubicles and daily slurry removal (Hristov et al., 2011). The model estimates that NH_3 emission from solid manure is assumed to be 50% of the mean emission from slurry based systems (Ross, Scholefield, Jarvis, 2002). In relation to the type of facility, the model considers that NH_3 emission is reduced by 35% in tie-stalls in comparison to free-stall systems (Monteny & Erisman, 1998). It also estimates that NH_3 emission is reduced by 36% in slatted floor systems compared to solid concrete floors (Pereira et al., 2011). The effect of slurry temperature on NH_3 losses is calculated according to Equation A19, which has previously been described for swine facilities. Mean slurry temperature is fixed at 15.8ºC (from Hristov et al., 2011) and the temperature inside the stall is estimated to be 2ºC higher than the atmospheric temperature throughout the year (INTIA database). Equation A22 is used to estimate the effect of manure removal frequency at dairy facilities. According to the available literature, daily NH_3 losses may increase by 40, 42, and 60% if slurry is removed fortnightly, monthly or with a frequency greater than one month, respectively (Hartung & Phillips, 1994). No data are currently available in the literature in relation to solid manure removal frequency. The slurry removal system will also regulate NH_3 losses from the barns. In this sense, the model considers mechanical scrapping as the default system and estimates that by using hose water flushing systems on solid or slatted floors up to 17 or 30% of emitted NH_3 respectively may be abated (Kroodsma, Huis In 't Veld, Scholtens, 1993; Ogink & Kroodsma, 1996). In accordance with these authors, the

model also estimates that NH_3 emission may be reduced by 20% if conveyors are installed at the farm. Finally, the variation factor estimated from abatement technologies is calculated according to Equation A23 reported in the swine section. Currently, the model assumes that NH_3 losses are reduced by 15% if curved slat mats with valves are installed (Teagasc pers. comm.).

Greenhouse gas losses (E_{N2O} and E_{CH4Man}, kg) from dairy barns are estimated via the following equations:

$$E_{N2O} = 44 \times 28^{-1} \times EF_{N2O} \times N_{ex}\ \ VF_{N2Ofreq} \times VF_{N2OOther} \quad (13)$$

$$E_{CH4Man} = VS_{ex} \times Bo \times 0{,}67 \times (MCF \times 100^{-1}) \times VF_{CH4Other} \quad (14)$$

In accordance with IPCC (2006), no N_2O emission is allocated to slurry-based facilities with concrete solid floors. Nitrous oxide emission is estimated to be 0.002 kg N_2O-N per kg N excreted if slurry is stored in deep pits under the slatted floors. For deep-litter based system, the model estimates that N_2O emission rate is 0.01 kg N_2O-N per kg N. In relation to CH_4 losses, according to IPCC (2006), maximum CH_4 producing capacity is 0.24 m^3 CH_4 per kg VS for dairy cattle (*Bo*). No data is currently available for emission rate variation of both gases with the use of curved slat mats with valves.

3.2.1.5. Nutrients in the manure removed from buildings

Nitrogen content in the outgoing manure is calculated as the N excreted minus the N volatilized at housing. Values such as total ammoniacal nitrogen (TAN) in respect to total nitrogen (TN), pH and density are taken from an extensive database of manure analysis (Tables E1, E2 and E3, Appendix E). Organic matter content in the removed manure is calculated assuming that the degradation rate of OM follows the variation of degradable C. Final DM is calculated deducting the OM and the N volatilized from the DM excreted. In poultry models, a ratio between OM and DM obtained from manure analysis is used to calculate the final dry matter content in the manure. When a bedding material is used, nutrients provided by it are also included in the calculations. The model does not consider P, K, Cu and Zn losses during manure management. Detailed methods and equations can be found in Appendix E.

3.2.1.6. Grazing

As previously described for confined cattle, N, P, K, Cu, Zn intake by grazing animals is estimated monthly based on the default values reported by Corpen (1999) and using Equations D4 to D7 (Appendix D). Nutrient retention by grazing cattle is assumed to be the same as confined animals.

Outdoors NH_3 emission from grazing animals (E_{NH3}, kg) is calculated according to the next equation:

$$E_{NH3} = 17 \times 14^{-1} \times EF_{NH3grazing} \times N_{exgrazing} \quad (15)$$

Nitrogen excreted during grazing ($N_{exgrazing}$) is calculated monthly for all stages (milking cows, dry cows, heifers and calves). Nitrogen excretion from milking cows is balanced with the number of hours that cows spend grazing outdoors. The model considers that dry cows, heifers and calves (50% of total calf herd at dairy farm) spend all of the day grazing. Ammonia emission factors ($EF_{NH3grazing}$) from grassland, have been collected from the RAINS model (Amman, Cofala, Klimont, Schöpp, 2004) and split at national level (Table F1, Appendix F).

Nitrous oxide emission from grassland (E_{N2O}, kg) is calculated according to the next equation:

$$E_{N2O} = 44 \times 28^{-1} \times (N_{exgrazing} \times (EF_{N2ODirect} + EF_{N2OLeaching})) + 14 \times 17^{-1}(E_{NH3}\ x\ EF_{N2OIndirect}) \quad (16)$$

Direct N_2O emission from N excretion ($EF_{N2ODirect}$), indirect N_2O emission from volatilized NH_3 ($EF_{N2OIndirect}$) and N_2O emission from N leaching ($EF_{N2OLeaching}$) are included in the calculations. Table F1 (Appendix F) shows the different emission factors included in the model. No CH_4 emission from manure management in grazing animals is considered by the model.

3.2.2. Storage subroutines

There are two storage subroutines in the model, one for liquid and one for solid manure. The quantity of material stored is calculated monthly considering the manure management indicated by the user (manure entrance from buildings, manure imported from outside and the quantity removed). Calculations are performed on a monthly basis and iteratively, adding the remaining manure from each month to next month's calculations

(Figure 3). The model assumes that once a year the storage is emptied. Nutrient inputs and gaseous losses are used for nutrient balance. Ammonia, nitrous oxide, carbon dioxide and methane emissions are calculated, including water balance (precipitation-evaporation).

3.2.2.1. Liquid manure storage

Model Input data

There are two types of liquid storage available in the model: single tanks or lagoons, with or without natural crusting. Simultaneously, up to 3 different storage types for liquids can be simulated in a farm. Table 7 shows all inputs parameters of this subroutine.

Calculations

Ammonia emissions (E_{NH3}, kg) are calculated according to Rigolot, Espagnol, Robin *et al.* (2010) (Equation 17). The model considers the influence of storage surface (m^2), storage time (days), temperature (VF_{Temp}), nitrogen dilution (VF_{Ndil}) and the effects of natural crust ($VF_{NH3Crust}$), covers ($VF_{NH3Cover}$) and additives (VF_{NH3Add}) (Equations G1 to G6 in Appendix G).

$$E_{NH3} = 1.57 \times 1000^{-1} \times VF_{Temp} \times VF_{Ndil} \times VF_{NH3Crust} \times VF_{NH3Cover} \times VF_{NH3Add} \times Surface \times Storage\ time \quad (17)$$

The effect of natural crusts has been taken from IPCC (2006) and MAGRAMA (2014) (Table G1, Appendix G). Previous works on effect of covers and additives on NH_3, N_2O and CH_4 emissions reported by Bicudo et al. (2004) and Ndegwaa, Hristovb, Arogoc and Sheffieldd (2008) respectively have been utilised in the model (Tables G2 and G3, Appendix G).

Total nitrogen quantity in stored slurry (*Total-N*, kg) is used for the N_2O emissions calculation (E_{N2O}, kg) along with the emission factor (kg N_2O-N kg TN^{-1}) which may vary in accordance to the existence of natural crusting ($EF_{N2OCrust}$) and the use of covers ($EF_{N2OCovers}$) (Equation 18). The effect of additives (VF_{N2OAdd}) is also considered.

$$E_{N2O} = 44 \times 28^{-1} \times Total\text{-}N \times (EF_{N2OCrust} + EF_{N2OCover}) \times VF_{N2OAdd} \quad (18)$$

Fig. 3. Schematic representation of storage subroutines.

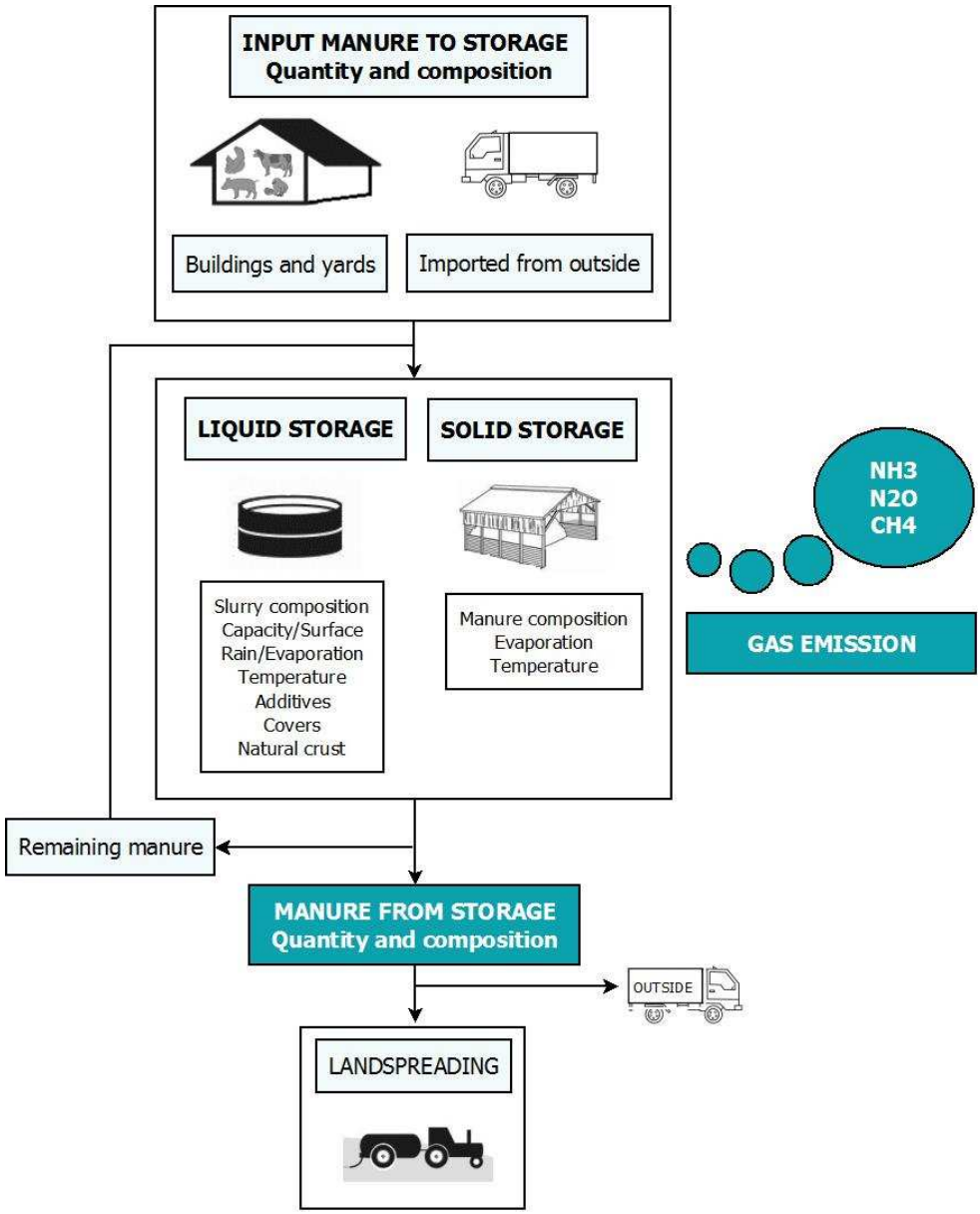


Table 7. Required input parameters for liquid storage subroutine.

MODEL INPUTS	VALUE	MODEL INPUTS	VALUE
Number of lagoons or tanks	1 to 3	Imported slurry from outside the farm	Data to select
Type of storage	Data to select	Type of cover	Data to select
Capacity of the storage (m^3)	Data to enter	Type of additive and months of application	Data to select
Surface of the storage (m^2)	Calculated by the model*	Slurry removal and destination	Data to select
Slurry input from housing	Calculated by the model		

*Modifiable

Methane emissions (E_{CH4Man}, kg) are based on IPCC (2006) Tier 2 methodology (Equation 19).

$$E_{CH4Man} = VS \times Bo \times MCF \times VF_{CH4Cover} \times VF_{CH4Add} \quad (19)$$

Where *VS* and *Bo* are the organic matter content (kg) and the methanogenic potential (m^3 CH_4 kg VS^{-1}) of the slurry, respectively. The methane conversion factor (*MCF*) is taken from IPCC (2006). $VF_{CH4Cover}$ and VF_{CH4Add}, accounts for the influence of covers and additives on CH_4 emission, respectively.

Water entry at storage is calculated considering precipitation and storage surface area. Evaporation is driven by climatic parameters (wind, relative humidity and temperature) and storage surface (Rowher, 1931) (Equation G7). In the presence of covers, evaporation is multiplied by a correction coefficient with a default value of 0.5 (which can be modified) and rainfall accumulation in the slurry can be considered null (in the case of impermeable covers).

The model does not consider P, K, Cu and Zn losses during storage. The nitrogen losses considered are NH_3 and N_2O emissions. Equations G9 to G13 (Appendix G) presents the calculation for the final manure quantity and composition.

3.2.2.2. Solid manure storage

Model Input data

The storage infrastructure is assumed to be covered by a roof (insulated regarding solar radiation but otherwise open to the atmosphere). Table 8 lists solid storage inputs parameters.

Table 8. Required input parameters for solid storage subroutine.

MODEL INPUTS	VALUE
Manure input from housing	Calculated by the model
Imported manure from outside the farm	Data to select
Manure removal and destination	Data to select

Calculations

The solid manure storage model has been adapted from Rigolot, Espagnol, Pomar, Hassouna, et al. (2010). If there is manure entry, it is assumed that the farmer will pile up the new layer of fresh manure on the first day of the month and in cases where a certain quantity is removed, this will be done the last day of the month. Only the manure stacked in the last four months is prone to gaseous emissions. The remaining manure is inert to gaseous emissions.

Gas emissions, $X_{Emitted}$ in kg (X=NH_3, N_2O, CH_4, CO_2) are based on the approach applied by Rigolot, Espagnol, Pomar, Hassouna, et al. (2010) for solid manure composting. The emissions factors (EF) correspond to the maximum emission measured by Paillat, Robin, Hassouna and Leterme (2005) in experiments with no turning, outside temperature of about 25ºC and over a two month period. EF is moderated by accounting for the manure type (VF_{Type}), temperature (VF_{Temp}) and duration (VF_{Dur}) (Equation 20) (Table H1, Appendix H). VF_{Temp} has been calculated by linearization of the values given by Rigolot, Espagnol, Pomar, Hassouna, et al. (2010).

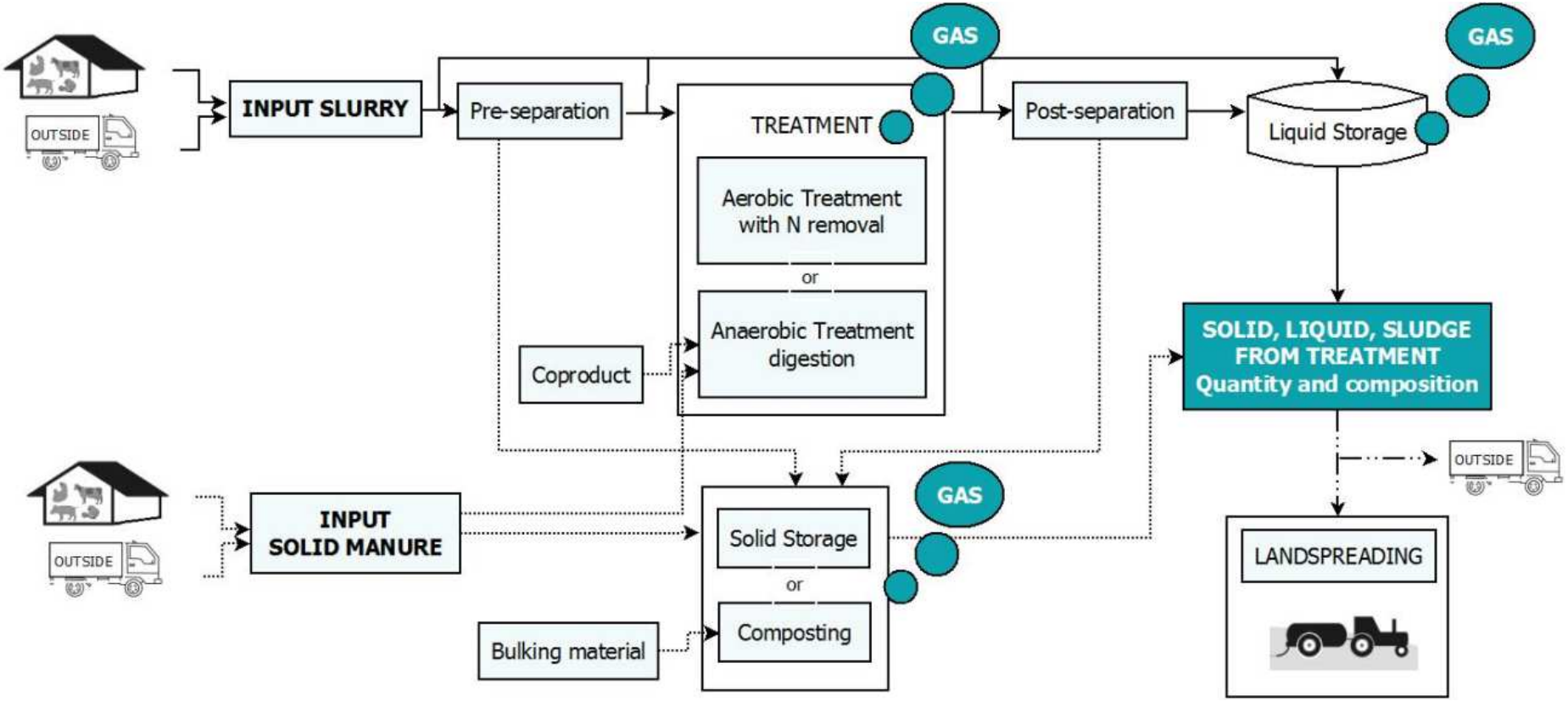
$$X_{Emitted} = EF \times VF_{Type} \times VF_{Temp} \times VF_{Dur} \quad (20)$$

Final manure quantity and composition equations follow the principles mentioned for liquid storage, (Appendix H, Table H1) under no rain entrance circumstances.

3.2.3. Treatment subroutine

Four major treatment techniques are considered: (1) Aerobic with Nitrification/Denitrification, (2) Anaerobic Digestion, (3) Phase Separation including separation before aerobic (pre-separation) or after aerobic/anaerobic (post-separation) and (4) Composting. Separation through gravity settling was also considered for liquid slurries after aerobic treatment. These treatment techniques can be combined in the model giving various whole treatment systems (Figure 4). Products from out of the farm can be incorporated in the treatment (manures, coproducts for anaerobic digestion and bulking agent used with laying hen compost). Default values for these imported manures from out of the farm are based on Levasseur et al. (2011).

Fig. 4. Schematic representation of treatment subroutines.



Model Input data

Depending on the treatment selected, model input data will vary (Table 9).

Table 9. Required input parameters for treatment subroutine, including default values (Source: IRSTEA)

MODEL INPUTS	VALUE	MODEL INPUTS	VALUE
Manure input from housing	Calculated by the model	**Anaerobic treatment**	
Imported material from outside	Data to select	Biogas leakage (%)	1.5
Type of treatment	Data to select	Use of biogas	Data to select
Separation		Type of treatment (Temperature)	Data to select
Type of separator	Data to select	**Liquid storage**	
Solid phase management	Data to select	Type of storage	Data to select
% of initial DM to Solid fraction	65-40	Storage average duration (days)	40
% of DM in the solid fraction	32-36	Storage surface (m²)	Calculated by the model*
Gravity Settling		Storage depth (m)	6
% Matter to Supernatant	66.5	**Solid storage/Composting**	
Aerobic treatment		Duration (days)	Data to select
Duration of the treatment (days)	40	Turning number (composting)	Data to select
Reactor surface (m²)	Calculated by the model*	Products removal and destination	Data to select
Reactor depth (m)	6		

*Modifiable

Calculations

The model calculates the composition of the different fractions (solid, liquid and sludge) and the gas emissions (NH_3, N_2O, CH_4, CO_2) resulting from the different treatment processes on a mass balance approach.

The emissions of NH_3, N_2O, CH_4 and CO_2 are calculated montly with daily emission factors related to the mass or volume of the treated manure or to the surface of slurry in contact with air and with time (number of days) (Equation 21).

$$E_{i\,m} = EF_{i\,m} \times V_i \times T_i \qquad (21)$$

With E_{im}, the emission of gas m (m = NH_3, N_2O, CH_4, CO_2) during the process i, EF_{im} the emission factor for gas m and process i, V_i the volume (or mass or surface) of the

treated slurry of process *i* and T_i the number of days of process *i* in one year. The mass of gas emissions (NH_3, N_2O, CH_4 and CO_2) are included in the mass balance. The default gas emission for the N mass balance is emitted as N_2. In each treatment process the calculation of *Bo x VS* variation (m^3 CH_4) follows the variation of degradable C. For treatment with surfaces open to the air, the calculation includes rain and evaporation based on Rowher's formula for a large surface (Rowher, 1931).

The composition of the different fractions (solid, liquid, sludge) and the gas emissions (NH_3, N_2O, CH_4 and CO_2) resulting from the different treatment processes are calculated with the following assumptions for each treatment type.

3.2.3.1. Separation by Screw Press, Centrifuge Decanter and Gravity settling

A key parameter for these processes is the percentage of Total Matter in the solid and the liquid fractions. The mass splitting parameters (x_{ijk}) are used for an annual amount of slurry production such as:

$$X_{ijk} = x_{ijk} \times S_{ik} \tag{22}$$

with X_{ijk} the mass of the element *k* (with *k* = total mass, DM, TKN, NH_4+, P, K, Cu, Zn) in the coproduct, *j* (with *j*= biological sludges, supernatant, liquid fractions, solid fractions...) coming from the process *i* (with *i* = separation, aeration, decantation...), x_{ijk} the fraction (%) of the element *k* in the input S_{ik} which is split into the co-products *j* (Table 10), S_{ik} the mass of element *k* in the input of the process *i*.

Table 10. Partition (x_{ijk}) of matter and nutrients of slurry between products issued from mechanical separation and gravity settling (as a percentage of the total amounts entering the treatment) (from Béline, Daumer, Guiziou, 2004 and Loyon, Guiziou, Beline, Peu, 2007).

		TM	DM	Nt	NH_4^+	P	K	Cu	Zn	VS	Ct
Centrifuge Decanter	**liquid phase**	91.5	35.0	70.0	91.5	20.0	91.5	67.0	50.0	35.0	35.0
	solid phase	8.5	65.0	30.0	8.5	80.0	8.5	33.0	50.0	65.0	65.0
Screw Press	**liquid phase**	95.3	60.0	91.5	95.3	77.3	95.3	92.0	89.5	60.0	60.0
	solid phase	4.7	40.0	8.5	4.7	22.7	4.7	8.0	10.5	40.0	40.0
Gravity settling	**supernatant**	66.5	20.0	20.0	57.0	30.0	67.0	15.0	17.5	20.0	20.0
	sludge	33.5	80.0	80.0	43.0	70.0	33.0	85.0	82.5	80.0	80.0

No gas emissions are considered for Screw Press or Centrifuge Decanter separation. Gas emissions associated with gravity settling are included in the storage subroutine after the treatment step.

3.2.3.2. Aerobic Treatment

Aerobic treatment by nitrification/denitrifcation described by Béline et al. (2004) and Loyon et al. (2007) allows removal of about 60–70% of the N content as gas (mainly N_2). The nitrogen reduction by nitrification/denitrification is fixed at 63% of the N entering the treatment (Béline et al. 2004; Loyon et al., 2007). There is no change for P, K, Cu and Zn while NH_4^+ is reduced by 95% and TN by 63% according to Béline et al. (2004) and Loyon et al. (2007). Gas emissions are calculated following emissions factors (E_{im}) derived from Loyon et al. (2007). Treatment duration is set at 40 days with a reactor depth of 6 m.

3.2.3.3. Anaerobic Digestion

Two cases are considered depending on the final use of the biogas produced (i) energy cogeneration/combustion or (ii) biofuel/network injection. Anaerobic digestion converts the majority of the organic N into ammonium, and thereby the concentration of ammonium in digested slurry is increased compared to undigested slurry. Mineralization of organic N to NH_3, which is important during the digestion, is assumed to be equivalent to C mineralization (Vedrenne, 2007). There is no change for P, K, Cu and Zn. Gas emissions are due to biogas leakage and partial burning of gas in the case of cogeneration. The produced biogas is mainly composed of CH_4 and CO_2, but a small amount of other gases is also produced. The model is set with a maximum value for biomethanogenic production of 70%. The amount of organic matter (VS) converted into biogas during the process is expressed as the *Bo* fraction associated with the % of CH_4 in the biogas from the main manure and co-product. Gas leakage is assumed to be 1.5% (IPCC, 2006; De Vries, Groenestein, De Boer, 2012). Ammonia and nitrous oxide emissions are negligible during anaerobic digestion (EC, 2003; IPCC, 2006).

3.2.3.4. Composting

Manure characteristics and gaseous emissions are calculated in one step, assuming that the process continues up to the final use of the product. Carbon biodegradability, N

availability, moisture content and dry matter density are the main factors involved in the composting process (El Kader, Robin, Paillat and Leterme, 2007; Paillat et al., 2005). The variation of nitrogen content (TN and NH_4^+), Carbon content (Total C, VS) is due to gaseous emissions which are calculated considering the approach of Rigolot, Espagnol, Pomar, Hassouna, et al. (2010) already explained in the solid storage model (Equation 20), but in this case also considering the number of turnings (VF_{Tur}) (Table 11).

Table 11. Emission Factor coefficients (EF), Variation Factors (VF) and Total N and water losses. Adapted from Rigolot et al. (2010) considering Moscatelli et al. (2008) and experts opinions.

		Emissions				Losses	
		NH_3-N	**N_2O-N**	**CO_2-C**	**CH_4-C**	**Total N**	**H_2O**
EF		0.45	0.03	0.45	0.015	0.50	0.75
VF_{Type}							
C:N	**DM (%)**						
<15	**<25**	0.4	1	0.6	1	1	0.8
	25 to 35	1	0.3	0.9	0.03	1	0.8
	36 to 74	0.7	0.1	0.8	0.02	0.8	0.5
	>75	0.09	0.01	0.1	0	0.1	0.04
15 to 25	**<25**	0.4	0.5	1	0.24	1	0.7
	25 to 35	0.8	0.3	1	0.04	0.9	1
	36 to 74	0.5	0.1	0.8	0.02	0.5	0.5
	>75	0.09	0.01	0.1	0	0.1	0.04
>25	**<25**	0.3	0.5	1	0.1	0.7	0.6
	25 to 35	0.5	0.3	1	0.02	0.6	0.9
	36 to 74	0.2	0.1	0.8	0.02	0.2	0.6
	>75	0.09	0.01	0.1	0	0.1	0.04
VF_{Tur}	**1**	1.1	1.2	1.2	1	1.1	1.2
	2	1.2	1.3	1.3	1	1.2	1.3
VF_{Dur}	**< 2 months**	1	1	1	1	1	1
	> 2 months	1.1	1.4	1.4	1.4	1.4	1.4
VF_{Temp}		*	1	**	**	**	**

*VF_{Temp}= (0.2-15*0.14/18)/375*temperature^2+0.14/18*temperature+(33.1-210*0.14/18)/48
**VF_{Temp}=(0.2-15*0.34/18)/375*temperature^2+0.34/18*temperature+(33.1-210*0.34/18)/48

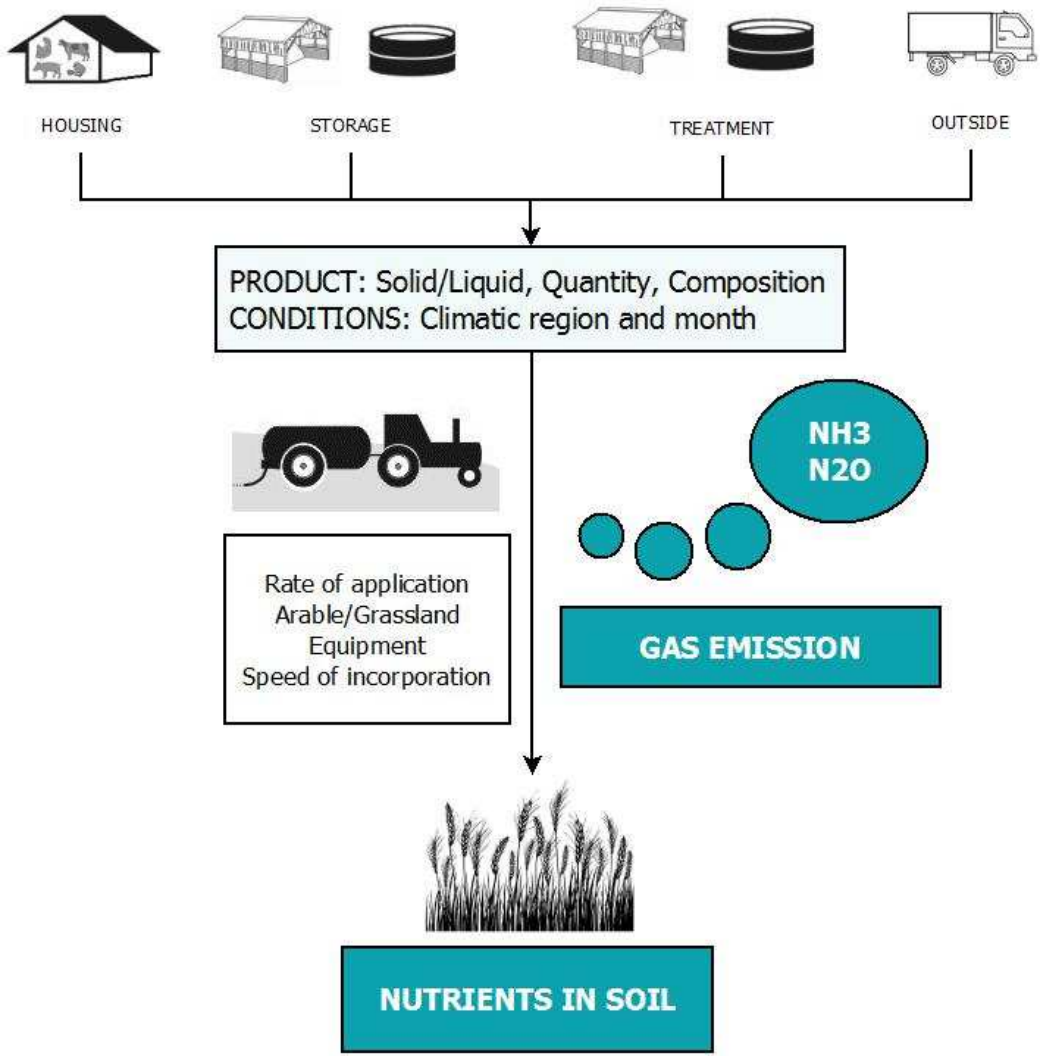
3.2.3.5. Storage after Treatment

Emissions due to the storage of the liquid and solid products resulting from treatment follows similar procedures to those already described for the storage subroutine, but a yearly storage time is used instead of monthly.

3.2.4. Landspreading subroutine

Landspreading, the last subroutine, is fed with the material sent from previous stages (housing, storage and/or treatment) and also considers imported material from outside the farm. Gas emissions (NH_3 and N_2O) and nutrients in soil after application (N, P, K, Cu, Zn) are also calculated, the overview of this subroutine is given in Figure 5.

Fig. 5. Schematic representation of landspreading subroutine.



Model Input data

For each month and product to be landspread, the model considers the following parameters: the manure application rate (t ha^{-1}), the type of land (arable/grassland), the application method (broadcast, trailing hose, trailing shoe, shallow injection and deep injection) and the speed of incorporation (no incorporation/immediate plough, immediate shallow, <4 hours, 4-24 hours, >24 hours) (Table 12).

Table 12. Required input parameters for landspreading subroutine.

MODEL INPUTS	VALUE	MODEL INPUTS	VALUE
Manure input from housing	Calculated by the model	Land	Arable/Grassland
Imported material from outside	Data to select	Method	Selected by the user
Rate (t ha-1)	Data to enter	Incorporation	Selected by the user

Calculations

A default emission factor for NH_3 loss ($EF_{NH3land}$, kg NH_3-N per kg TAN applied) is included for each climatic region, manure type, and land type. Ammonia emission factors vary between month and regions due to varying weather conditions (principally wind speed and temperature) and in some regions, this was estimated using the ALFAM model (Søgaard et al., 2002). Application method ($ABATE_{METHOD}$, %), manure incorporation ($ABATE_{INCORP}$, %), and manure dry matter content ($ABATE_{DM}$, %) will also affect the extent of NH_3 loss during application. Ammonia emissions have been shown to rise with increasing dry matter content (Misselbrook, Smith, Jackson and Gilhespy, 2004). The model assumes as a default that for every 1% reduction in slurry dry matter content this will reduce emissions of NH_3-N as a percentage of TAN applied by 0.6% (Bittman, Dedina, Howard, Oenema and Sutton, 2014). Default dry matter contents for liquid manures used by the model to calculate $ABATE_{DM}$ are shown in Table I1 (Appendix I). Emission abatement factors due to the application method ($ABATE_{METHOD}$) and manure incorporation ($ABATE_{INCORP}$), have been taken from EC (2013), Lalor (2014), MAGRAMA (2014), and Bittman et al. (2015) (Table I2, Appendix I). Ammonia loss (E_{NH3}, kg NH_3 ha^{-1}) is then calculated from the TAN application rate (R_{TAN}, kg TAN, ha^{-1}) according to the manure composition and application rate (Equation 23).

$$E_{NH3} = 17 \times 14^{-1} \times EF_{NH3land} \times R_{TAN} \times [1 - (ABATE_{DM} + ABATE_{METHOD} + ABATE_{INCORP}) \times 100^{-1}] \quad (23)$$

The total ammoniacal N abated (TAN_{ABATE}, kg TAN ha^{-1}) or saved due the technique used rather than the default scenario (broadcast or splashplate application with no incorporation and default slurry DM content) is calculated by Equation 24.

$$TAN_{ABATE} = EF_{NH3land} \times R_{TAN} \times (ABATE_{DM} + ABATE_{METHOD} + ABATE_{INCORP}) \times 100^{-1} \quad (24)$$

The emissions of N_2O following landspreading are divided into: direct emissions from N applied in the manure, indirect emissions from redeposited E_{NH3} and emissions from chemical N fertiliser that are offset due to substitution with TAN_{ABATE}. Detailed equations of N_2O emissions can be found in Appendix I.

3.2.5. Additional subroutines

3.2.5.1. Energy balance

The energy subroutine derives the energy (KWh $year^{-1}$) and CO_2 equivalent (t CO_2 eq $year^{-1}$) farm balances evaluating: the electricity and combustion consumptions, the energy produced by biogas (when there is anaerobic digestion with combustion/co-generation) and optionally, the effect of housing abatement techniques selected (Equations J1 and J2, Appendix J).

Total farm electricity consumption and the type and quantity of combustible consumed have to be indicated. At this point, different types can be selected (fossil fuels and biofuels) (Table J1, Appendix J). The heating values and emission factors of the different combustibles are taken from IPCC (2006), MARM (2011) and pellet analyses. Direct CO_2 combustion emission for the biomass products has been considered as zero, according to the 2009/28/CE Directive. Biomass from lignocellulosic compounds presents a CO_2 neutral balance as it closes crop growth carbon cycle (Sanz, López, Callejo, Parras, 2010). Nevertheless, N_2O and CH_4 produced during biomass combustion have been included in the CO_2 equivalents calculations due to their greater warming potential values. The model allocates 28 and 265 CO_2 equivalents are produced per unit of CH_4 and N_2O, respectively (Myhre et al., 2013). Also, total electricity consumption is transformed into CO_2 equivalents using 0.65 kg CO_2 per KWh. All these values can be modified by users.

Anaerobic digestion can be associated either with co-generation or combustion. A heating value of 23 MJ per m^3 of biogas produced is utilised to calculate the total energy

produced (Nielsen, 2004). The different percentage of heat used are applied depending on the biogas exploitation (co-generation/combustion plants) according to IRSTEA database, but can be adjusted by users (Figure J1, Appendix J). As in the case of biomass products, the energy model considers direct CO_2 combustion emission from biogas equal to zero (although this CO_2 emission is calculated in the treatment subroutine). N_2O and CH_4 emissions produced during anaerobic digestion (due to biogas leakage and to non-combusted biogas in the reactor) have been imported from the treatment model to be converted into CO_2 equivalents.

3.2.5.2. Evaluation of Faecal Indicator Organisms

Evaluation of the management effects on faecal indicator organisms (FIO) is based on a qualitative approach due to the fact that FIO counts describe the presence of live organisms in response to general biological activity. This qualitative evaluation also means that the effect of different farming processes cannot be compared or summated. Consequently, each individual activity is evaluated on a 'stand-alone' basis because in all but the most extreme cases the effect of each process step can be outweighed or negated by the following step. Poor land-application, for example, cannot be compensated for by good housing. The FIO assessments adopted in the model (Table K1, Appendix K) range from:

- "negligible effect" (normal practice), to
- "some effect" (good practice likely to lead to FIO reduction in the region of 1-2 orders of magnitude), and
- "high effect" (good practice likely to lead to FIO reduction by 3 orders of magnitude or above).

Final evaluation of the overall farm is undertaken by the user: even though consistent good practice will reduce the number of FIOs escaping into the natural environment, most of the environmental burden arises in the final stage, i.e. during land application of manures and slurries, and good practice at this stage is paramount if FIO pollution prevention is a key target. Here, it is important to distinguish between viability of FIOs in the field and escape of FIOs into the aquatic environment i.e. creation of environmental pollution: different pathogenic types of organisms are affected differently by environmental conditions (e.g. sun radiation, moisture levels, etc) and some organisms are viable but not

cultivable. Thus, the model assumes that reducing escape of FIOs from the field has a greater pollution reducing effect than kill-off in the field.

In housing, good practice centres around removal of manures so that re-infection of the herd is kept low, and drying of manures (poultry) and long term storage without addition of fresh slurry (grazing dairy cattle) occurs to reduce micro-organisms' viability (Burton & Turner, 2003; FEC, 2003; Reinoso & Becares, 2008; Vanotti, Millner, Hunt, Ellison, 2005). There is little published information on the effect of dietary modifications on FIO, with data on feed additives and phase feeding being inconclusive or conflicting.

During storage, most FIOs decline in number, dependant on various environmental factors. FIO decline is most pronounced in storage without 're-infection' due to addition of fresh material (Côte, Villeneuve, Lessard, Quessy, 2006; Hutchison, Walters, Avery, Synge, Moore, 2004; SEPA, Personal Communication; Sneath, Beline, Hilhorst, Peu, 2006) and over longer time periods (SEPA, Personal Communication). The use of additives in (liquid) storage has no effect on FIO survival and growth (Turner & Williams, 1999), except for pH altering.

For treatment, good practice centres around mesophilic and thermophilic biological treatment, solids separation, and incineration of solids manures (normally restricted to poultry manure) leading to complete removal of FIOs (Bendixen & Bennetzen, 1995; Bendixen, 1999; Burton & Turner, 2003; FEC, 2003; Martens & Böhm, 2009). The overriding factor is temperature, followed by time. For all thermophilic (and mesophilic) treatment, there is a large risk of reinfection, which must be carefully managed for example by pile separation, sequential or two-stage operation. Solids separation has limited effect on FIO counts especially for coarse solids separation as would be applied in slurry treatment (Reinoso & Becares, 2008; Vanotti et al., 2005).

Land application. The model assumes that bad practices such as application on water logged soil, frozen ground or in excess volumes is not undertaken. Measures to reduce FIO pollution should be aimed at retaining any FIOs in the soil, i.e. reducing the risk of run-off after application (Moriarty, Mackenzie, Karki, Sinton, 2011; Nicholson, Groves, Chambers, 2005; Van Kessel, Pachepsky, Shelton, Karns, 2007). There is some evidence that exposure to oxygen or sun light (UV) can reduce FIO survivability whereas soil incorporation increases survivability. However, this affects different organisms in different and sometimes juxtapositional ways, and survival will reduce naturally over time.

Finally, whilst there is evidence indicating that more natural behaviour during grazing reduces stress-induced shedding of FIOs in cattle, there is also significant pollution potential for surface waters. Poaching, i.e. the grass sward being cut-up through cattle moving on wet soils, can be a problem, particularly around watering points, leaving the soil open to the elements and prone to FIO water run-off (Kay et al., 2007; SEPA, 2012).

3.2.5.3. Cost of the housing abatement techniques

The model brings the option of introducing parameters to calculate the cost of the housing abatement techniques selected according to EC (2003) equations.

$$A_c = C \times [r \times (1+r)^n] \times [(1+r)^n - 1]^{-1} \quad (25)$$

Equation 25 calculates the annual cost of capital (A_c, euros per year), where C *is* the total capital investment cost (euros), n the life of the investment (years) and r the interest rate (decimal of 1). Total cost (T_c) will be then calculated adding to annual cost of capital (A_c), additional variable costs: labour (A_l), energy/fuel (A_e) and reparations/maintenance (A_r), all expressed in euros per year (Equation 26).

$$T_c = A_c + A_l + A_e + A_r \quad (26)$$

3.2.6. Whole farm emission and scenario comparison

The model calculates the whole farm emission of ammonia, nitrous oxide and methane, by adding the individual emission of the gas produced in each subroutine, results are shown on annual and monthly basis.

Two different scenarios already simulated for a particular farm can be compared. Relevant parameters variation (X_{AB}, %) is calculated as the percentage that scenario A is minor (negative value) or major (positive value) respect to B (Equation 27), with X_A the value of parameter X in scenario A, whereas X_B is the value of parameter X in scenario B.

$$X_{AB} = (X_A - X_B) \times X_B^{-1} \times 100 \quad (27)$$

For each parameter, if situation A is more favourable than situation B, a smile will appear in the score column. Relevant parameters able to be selected for comparison are listed in Annexe L.

3.3. Discussion and conclusion

This model predicts gaseous emissions and nutrient flows from existing farms and allows simulation of the effect of several farm management strategies on emissions and consumptions.

However, there are other features missing in the initial version of this model. For example the current version lacks crop nutrient balance after landspreading and it does not consider litter based systems in swine and laying hen farms. The current version does not yet consider aqueous streams that pollute the soil, surface and ground water. It is expected that future developments will overcome these limitations and will also incorporate more types of environmental strategies enriched with new knowledge as it becomes available.

The level of model development differs among the main subroutines, with the housing modules being more elaborated than the rest. The landspreading subroutine could be completed with other existing models which contemplates the effect of a wider range of variables that affects on air, soil and water pollution. It would be also be desirable to deepen the additional subroutines methods (cost calculator and energy balance), as they are important aspects of BAT selection. In particular, the model could be completed with cost-effectiveness calculations, considering farm emissions and the cost-benefit throughout the whole production process.

As stated before, modifiable default values have been included to develop a versatile and user-friendly model, which can be adjusted to state-specific requirements without the need to alter source code. Regardless, the availability and quality of data varies considerably within and between parameters. In the current version of the model most of regionalizable default values (zootechnical data and emission factors) have the same value for the different countries. Therefore, an important challenge to be faced would be to revise and update the model's database, especially for the regionalizable values to improve estimations at regional scale.

Input data describing the system or any modifications introduced on model database, must be consistent with each other to produce reliable predictions. Especially considering that *VF* in emissions calculations, are not necessarily independent (Rigolot, Espagnol, Pomar, Hassouna, et al., 2010).

Model validation would be required to broaden it's use, not only for scenario comparisons at a particular farm, but also for the calculation of absolute farm emissions e.g. to be reported to the European Pollutant Release and Transfer Register (E-PRTR). This aspect will be addressed in the future, conducting detailed sensitivity analysis, testing model results with empirical data and comparing it with other similar tools.

To improve model construction based on empirical data, it is necessary to count on detailed farm studies describing not only environmental and measuring conditions, but also management aspects and facility characteristics that in some occasions are overlooked.

Even with these limitations, the model considers that the effects of any management change are transferred throughout the entire system identifying the resulting net farm emission. In some cases this approach helps us to identify synergies (reduction of both NH_3 and N_2O) and trade-offs (compromise between N_2O and CH_4 emissions) avoiding potentially ill-advised practices based on a preoccupation with one individual gas.

Another important aspect is that the model predicts the emissions of each gas (NH_3, N_2O, CH_4) troughout the year and by farm stages (housing, storage, treatment and landspreading). Moreover, the model also provides emissions for the different animals housed. All this information can be useful to identify where and when major emissions are produced and therefore which mitigation strategies could achieve optimum results in a particular farm.

The model also predicts the nature of the manure produced and its composition, which can help to determine opportunities and constraints to their use (treatments, agronomic use, export out of the farm), and possible associated environmental impacts.

In conclusion, the derived shortcomings of our model means that the results obtained must be interpreted as indicative of the relative emission reduction achieved in particular farms due to mitigation practices implemented rather than absolute emission values. Further research and validation is needed to extend its use. Despite these limitations, the model can be used to assist in decision making regarding selection of appropriate mitigation practices that are best suited to the region and farm circumstances. In any case, professional advice with overall farm vision, including economic aspects, should be sought on the interpretation of the results provided by the model and to evaluate the plausibility of the different proposals to each particular farm situation.

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3.5. References

Abaigar A., Íñigo J.A. and Cordovín L. (2005). El empleo de sistemas de abrevamiento eficientes. Navarra Agraria, 152, 55-64.

Acutis M., Alfieri L., Giussani A., Provolo G., Di Guardo A., Colombini S., Bertoncini G., Castelnuovo M., Sali G., Moschini M., Sanna M., Perego A., Carozzi M., Chiodini M .E. and Fumagalli M. (2014). ValorE: An integrated and GIS-based decision support system for livestock manure management in the Lombardy region (northern Italy). Land Use Policy 41, 149–162.

Al Homidan A., Robertson J.F. and Petchey, A.M. (1997). Effect of temperature, litter and light intensity on ammonia and dust production and broiler performance. British Poultry Science, 38, S5-S6.

Albar J. and Granier R. (1996). Incidence du taux azoté de l'aliment sur la consommation d'eau, la production de lisier et les rejets azotés en engraissement. Journées Recherche Porcine en France, 28, 257 - 266.

Alberdi O., Arriaga H., Calvet S., Estellés F. and Merino P. (2016). Ammonia and greenhouse gas emissions from an enriched cage laying hen facility. Biosystems Engineering, 144, 1-12.

Amann M., Cofala J., Klimont Z. and Schöpp, W.(2004). RAINS Review 2004 – Modelling of Emission Control Potentials and Cost. IIASA, Austria

Arriaga H. (2010). Cattle nutrition as a strategy to mitigate gaseous nitrogen losses from dairy farming. PhD-thesis. ISBN: 978-84-457-3082-9

Béline F., Daumer M.L. and Guiziou F. (2004). Biological aerobic treatment of pig slurry in France: nutrients removal efficiency and separation performances. Transactions of the ASAE. 47, 857-864.

Bendixen H.J. (1999). Hygienic safety: results of scientific investigations in Denmark (sanitation requirements in Danish Biogas Plants). In proceedings of the IEA workshop: Hygienic and environmental aspects of anaerobic digestion: legislation and experiences in Europe. Universität Hohenheim, Stuttgart, Germany, 27-47.

Bendixen H.J. and Bennetzen O. (1995). Smitstofreduktion i biomasse: rapport vedrørende det veterinære forsøgsprogram i biogasfællesanlæg. Published by the Danish Veterinary Service, Friederiksberg, Denmark.

Berntsen J., Petersen B.M., Jacobsen B.H., Olesen J.E. and Hutchings N.J. (2003). Evaluating nitrogen taxation scenarios using the dynamic whole farm simulation model FASSET. Agricultural Systems 76, 817–839.

Berntsen J., Petersen B.M., Olesen J.E. and Hutchings N. (1999). FASSET - An Integrated Economic and Environmental Farm Simulation Model. International Symposium Modelling Cropping Systems, Lleida (Spain), 45-56.

Bicudo J., Clanton C., Schmidt D., Powrs W., Jacobson L. and Tengman C. (2004). Geotextile covers to reduce odor and gas emissions from swine manure storage. Applied Engineering in Agriculture, 20 (1), 65-75.

Bittman S., Dedina M., Amon B., Menzi H., Webb J., Groenestein K., Misselbrook T., Hutchings N., Dohler H., van der Hoek K., Gyldenkærne S., Valli L., Pallière C., Howard C., Oenema O. and Sutton M. (2015). Framework Code for Good Agricultural

Practice for Reducing Ammonia Emissions. European Commission on behalf of the Task Force on Reactive Nitrogen of the UNECE Convention on Long-range Transboundary Air Pollution, 32 pp.

Bittman S., Dedina M., Howard C.M., Oenema O. and Sutton M.A. (2014). Options for Ammonia Mitigation: Guidance from the UNECE Task Force on Reactive Nitrogen. Centre for Ecology and Hydrology, Edinburgh, UK, 83 pp.

Burton C.H. and Turner C. (2003). Manure management — treatment strategies for sustainable agriculture. 2nd Edition. Silsoe Research Institute, Wrest Park, Silsoe, Bedford (UK), 490 pp.

Calvet S., Cambra-López M., Estellés F. and Torres A.G. (2011). Characterization of gas emission from a Mediterranean broiler farm. Poultry Science, 90, 534-542.

Chardon X., Rigolot C., Baratte C., Le Gall A., Espagnol S., Martin-Clouaire R., Rellier, J.P., Raison C., Poupa J.C. and Faverdin P. (2007). MELODIE : a whole-farm model to study the dynamics of nutrients in integrated dairy and pig farms. Proceedings of the International Congress on Modelling and Simulation, Modelling and Simulation Society of Australia and New-Zealand, 1638-1645.

Chosson C., Granier R., Maigne A., Bouby A. and Mongin J.P. (1998). Reduction du volumen de lisier produit par un porc a l`engrais. TechniPorc , 11, 27-41.

Collin A., Van Milgen J. and Le Dividich J. (2001). Modelling the effect of high, constant temperature on food intake in young growing pigs. Animal Science, 72, 519-527.

Corpen (1999). Estimation des flux d'azote, de phosphore et de potassium associés aux vaches laitières et à leur système fourrager Influence de l'alimentation et du niveau de production, 18 pp.

Corpen (2003). Estimation des rejets d`azote-phosphore-potassium-cuivre et zinc des porcs. Influence de la conduite alimentaire et du mode de logement des animaux sur la nature et la gestion des déjections produites, 41 pp.

Corpen (2006). Estimation des rejets d`azote-phosphore-potassium-calcium-cuivre et zinc par les élevages avicoles. Influence de la conduite alimentaire et du mode de logement des animaux sur la nature et la gestion des déjections produites, 55 pp.

Corpen (2013). Estimation des rejets d`azote-phosphore-potassium-calcium-cuivre et zinc par les élevages avicoles. Mise à jour des références CORPEN-Volailles de 2006, 63 pp.

Côte C., Villeneuve L., Lessard L. and Quessy S. (2006). Fate of pathogenic and non-pathogenic microorganisms during storage of liquid hog manure in Quebec. Livestock Science, 102(3), 204-210.

Da Borso F. and Chiumenti R. (1999). Poultry housing and manure management systems: recent developments in Italy as regards ammonia emissions. Proceedings of the 8th International Conference of the FAO ESCORENA Network on the Recycling of Agricultural, Municipal and Industrial Residues in Agriculture, Rennes (France), 15-21.

Dämmgen U., Lüttich M., Döhler H., Eurich-Menden B. and Osterburg B. (2003). GAS-EM – a procedure to calculate gaseous emissions from agriculture. Landbauforschung Völkenrode 52, 19-42.

De Vries J.W., Groenestein C. M., De Boer I.J. (2012). Environmental consequences of processing manure to produce mineral fertilizer and bio-energy. Journal of Environmental Management 102, 173-183.

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

EC (2003). Reference Document on Best Available Techniques for intensive Rearing of Poultry and pigs.Integrated Pollution Prevention and Control (IPPC), July 2003, 341 pp.

EC (2013). Best Available Techniques (BAT) Reference Document for the Intensive Rearing of Poultry and Pigs. Integrated Pollution Prevention and Control (IPPC), Draft 2-August 2013, 839 pp.

EC (2015). Best Available Techniques (BAT) Reference Document for the Intensive Rearing of Poultry or Pigs. Final Draft - August 2015, European Commision, 911 pp.

El Kader N.A., Robin P., Paillat J.M. and Leterme P. (2007). Turning, compacting and the addition of water as factors affecting gaseous emissions in farm manure composting. Bioresource Technology 98, 2619-2628.

Ellis J.L., Kebreab E., Odongo N.E., McBride B.W., Okine E.K. and France J. (2007). Prediction of methane production from dairy and beef cattle. Journal of Dairy Science 90, 3456-3467.

Elwinger and Svensson (1996). Effect of Dietary Protein Content, Litter and Drinker Type on Ammonia Emission from Broilers Houses. Journal of Agricultural Engineering Research 64, 197-208.

Fabbri C., Valli L. and Bonazzi G. (2004). A method to assess the reduction of Ammonia and Methane emissions by application of Bat in intensive livestock farming. Proceedings of the 11th International Conference of the FAO ESCORENA Network on the Recycling of Agricultural, Municipal and Industrial Residues in Agriculture, Murcia (Spain), 301-304.

Fabbri C., Valli L., Guarino M., Costa A. and Mazzotta V. (2007). Ammonia, methane, nitrous oxide and particulate matter emissions from two different buildings for laying hens. Biosystems Engineering, 97, 441-455.

FEC Services (2003). Anaerobic digestion, storage, oligolysis, lime, heat and aerobic treatment of livestock manures. FEC Services Ltd., UK, 110 pp.

Fiedler E. (1982). Versuchs und Erfahrungsbericht. Landesanstalt für Schweinzucht Forchheim, 72-75.

Fournel S., Pelletier F., Godbout S., Lagacé R. and Feddes J.J.R. (2012). Odour emissions, hedonic tones and ammonia emissions from three cage layer housing systems. Biosystems engineering, 112, 181-191.

Gooday R.D., Anthony S.G., Chadwick D.R., Newell-Price P., Harris D., Duethmann D., Fish R., Collins A.L. and Winter M. (2014). Modelling the cost-effectiveness of mitigation methods for multiple pollutants at farm scale. Science of the Total Environment 468-469, 1198-1209.

Granier R., Guingand N. and Massabie P. (1996). Influence du niveau d'hygrométrie, de la température et du taux de renouvellement de l'air sur l'évolution des teneurs en ammoniac. Journé es de la Recherche Porcine en France 28, 209-216.

Grant R.J. and Bodman G.R. (1995). Guidelines for using computerized concentrate feeders for dairy herds. Historical Materials from University of Nebraska-Lincoln Extension. Paper 491.

Guillou D., Dourmand J.Y. and Noblet J., (1993). Influence de l`alimentation, du stade physiologique et des performances sur les rejets azotés du porc à l`engrais de la truie et du porcelet. Journées Recherche Porcine en France, 25, 307-314.

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

Hernández-Benedí J.M. (1984). Manual de Nutrición y Alimentación de Ganado. Ministerio de Agricultura, 487 pp.

Hristov A.N., Hanigan M., Cole A., Todd R., McAllister T.A., Ndegwa P.M. and Rotz A. (2011). Review: Ammonia emissions from dairy farmsand beef feedlots. Canadian Journal of Animal Science 91, 1-35.

Hutchings N.J., Sommer S.G., Andersen J.M. and Asman W.A.H., (2001). A detailed ammonia emission inventory for Denmark. Atmospheric Environment, 35, 1959-1968

Hutchison M.L., Walters L.D., Avery S.M., Synge B.A. and Moore, A. (2004). Levels of zoonotic agents in British livestock manures. Letters in Applied Microbiology, 39(2), 207-214.

INRA (1984). L`alimentation des animaux monogastriques : porc, lapin, volailles. INRA, Paris (France), 282 pp.

Institut de L'Elevage, Ifip and Itavi (2010). Guide des Bonnes Pratiques Environnementales d'Élevage, 303 pp. http://www.rmtelevagesenvironnement.org/pdf/bpe_avi_porcs_bovins.pdf

Institut de L'Elevage, Itavi, ITCF and ITP (2001). Fertiliser avec les engrais de ferme, 104 pp.

IPCC (2006). IPCC guidelines for national greenhouse gas inventories, prepared by the national greenhouse gas inventories programme. Eds H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, K. Tanabe, Hayama, Japan: IGES, 87 pp.

Jonker J.S., Kohn R.A. and High J. (2002). Dairy herd management practices that impact nitrogen utilization efficiency. Journal of Dairy Science 85, 1218-1226.

Karmakar S., Nketia M., Laguë C. and Agnew J. (2010). Development of expert system modelling based decision support system for swine manure management. Computers and Electronics in Agriculture 71, 88-95.

Kay D, Aitken M, Crowther J, Dickson I, Edwards A.C., Francis C., Hopkins M., Jeffrey W., Kay C., McDonald A.T., McDonald D., Stapleton C.M., Watkins J., Wilkinson J. and Wyer M.D. (2007). Reducing fluxes of faecal indicator compliance parameters to bathing waters from diffuse agricultural sources: the Brighouse Bay study, Scotland. Environmental Pollution, 147(1), 138-149.

Khalili H., Mäntysaari P, Sariola J. and Kangasniemi R. (2006). Effect of concentrate feeding strategy on the performance of dairy cows fed total mixed rations. Agricultural and Food Science 15, 268-279.

Kroodsma W., Huis In 't Veld J.W.H. and Scholtens R.(1993). Ammonia emissions and its reduction from cubicle houses by flushing. Livestock Production Science 35, 293-302.

Lalor S.T.J. (2014). Cattle slurry on grassland - application methods and nitrogen use efficiency. Wageningen University, The Netherlands. 184 pp.

Latimier P., Gallard F. and Corlouër A. (1996). Actualisation des volumes et des quantités d'azote, de phosphore et de potasse rejetés dans le lisier par un élevage naisseur-engraisseur. Journées Recherche Porcine en France, 28, 241-248.

Le Goff G. and Noblet J. (2001). Utilisation digestive compare´ e de l'e´ nergie des aliments chez le porc en croissance et la truie adulte. Journées de la Recherche Porcine, Paris, France. Editions ITP-INRA 33, 211-220.

Levasseur P. (1998). Composition et volume de lisier produit par le porc. Données bibliographiques. TechniPorc, 21, 17-24.

Levasseur P., Aubert C., Berger S., Charpiot A., Damiano A., Meier V., Quideau P. (2011). Développement d'un calculateur pour déterminer l'intérêt technico-économique de

la méthanisation dans les différents systèmes de productions animales : Méthasim. Innovations Agronomiques 17, 241-253.

Loyon L., Burton C.H., Misselbrook T., Webb J., Philippe F.X., Aguilar M., Doreau M., Hassouna M., Veldkamp T., Dourmad J.Y., Bonmati A., Grimm E. and Sommer S.G. (2016). Best available technology for European livestock farms: availability, effectiveness and uptake. Journal of Environmental Management 166, 1-11.

Loyon L., Guiziou F., Beline F., Peu P. (2007). Gaseous Emissions (NH_3, N_2O, CH_4 and CO_2) from the aerobic treatment of piggery slurry - Comparison with a conventional storage system. Biosystems Engineering, 97(4), 472-480

MAGRAMA (2014). Evaluación de Técnicas de Reducción de Emisiones en Ganadería. Sectores de Porcino y Avicultura de Carne y Puesta. Ministerio de Agricultura, Alimentación y Medio Ambiente, Madrid, 114 pp.

MARM (2008). Metodología para la estimación de las emisiones a la atmósfera del sector ganadero para el inventario nacional de emisiones. Ministerio de Medio Ambiente y Medio Rural y Marino, España, 221 pp.

MARM (2010). Guía de Mejores Técnicas Disponibles del Sector de la Avicultura de Puesta. Ministerio de Medio Ambiente y Medio Rural y Marino, España, 86 pp. http://www.magrama.gob.es/es/ganaderia/publicaciones.

MARM (2011). Inventario de emisiones de gases de efecto invernadero de España e información adicional 1990-2009. Secretaría de Estado de Cambio Climático, Dirección General de Calidad y Evaluación Ambiental, D.G. Oficina Española de Cambio Climático, Madrid, 706 pp.

Martens W. and Böhm R. (2009). Overview of the ability of different treatment methods for liquid and solid manure to inactivate pathogens. Bioresource Technology, 100(22), 5374-5378.

Massabie P. (2001). L'abreuvement des porcs.Techniporc, 24(6), 9-14.

Meda B., Robin P., Aubert C., Dourmad J-Y and Hassouna M. (2012). MOLDAVI: A model to predict environmental and economic performances of broiler farming systems. Producing and reproducing farming systems. New modes of organisation for sustainable

food systems of tomorrow. 10th European IFSA Symposium, Aarhus (Denmark), unpaginated.

Menzi H., Bonjour C., Zaucker F., Leuenberger C. and Reidy B. (2009). Agrammon: A New Internet Based Model for the Differentiated Estimation of Ammonia Emissions from Individual Farms or at the Area-Wide Scale. Ecology and Future-Bulgarian Journal of Ecological Science, 8(1), 28-32.

Menzi H., Rüttimann L. and Reidy B. (2003). DYNAMO: a new calculation model for dynamic emission inventories for ammonia. International Symposium on Gaseous and Odour Emission from Animal Production Facilities, Horsens (Denmark), 378-381.

Misselbrook T.H., Smith K.A., Jackson D.R. and Gilhespy S.L. (2004). Ammonia emissions from irrigation of dilute pig slurries. Biosystems Engineering, 89 (4), 473-484.

Monteny G.J. and Erisman J.W. (1998). Ammonia emissions of dairy cow buildings: a review of measurement techniques, influencing factors and possibilities for reduction. Netherlands Journal of Agricultural Science 46, 225-247.

Moriarty E.M., Mackenzie M.L., Karki N. and Sinton LW. (2011). Survival of Escherichia coli, Enterococci, and Campylobacter spp. in Sheep Feces on Pastures. Applied Environmental Microbiology, 77(5), 1797-803.

Myhre G., Shindell D., Bréon F.M., Collins W., Fuglestvedt J., Huang J., Koch D., Lamarque J.F., Lee D., Mendoza B., Nakajima T., Robock A., Stephens G., Takemura T. and Zhang H. (2013). Anthropogenic and Natural Radiative Forcing. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 82 pp.

Ndegwaa P., Hristovb A., Arogoc J. and Sheffieldd R. (2008). A review of ammonia emission mitigation techniques for concentrated animal feeding operations. Biosystems Engineering, 100, 453-469.

Nicholson F.A., Chambers B.J. and Walker A.W. (2004). Ammonia emissions from broiler litter and laying hen manure management systems. Biosystems Engineering, 89(2), 175-185.

Nicholson F.A., Groves S.J. and Chambers B.J. (2005). Pathogen survival during livestock manure storage and following land application. Bioresource Technology, 96(2), 135-143.

Nielsen H. (2004). Heat and power production from pig manure.The Institute for Product Development.
http://www.lcafood.dk/processes/energyconversion/heatandpowerfrommanure.htm

NRC. (2001). Nutrient Requirements of Dairy Cattle. Seventh revised edition. National Academy Press. Washington D.C. 408 pp.

Ogink N.W.M. and Kroodsma W. (1996). Reduction of ammonia emission from a cow cubicle house by flushing with water or a formalin solution.Journal of Agricultural Engineering Research 63, 197-204.

Paillat J.M., Robin P., Hassouna M. and Leterme P. (2005). Effet du compostage d'effluents porcins sur les émissions gazeuses et les teneurs en éléments polluants. Rapport final convention ADEME-INRA 0375C0077, GIS Porcherie Verte, UMR SAS, Rennes, France.

Patterson P.H. and Adrizal (2005). Management strategies to reduce air emissions: emphasis-dust and ammonia. The Journal of Applied Poultry Research, 14, 638-650.

Pereira J., Fangueiro D., Misselbrook T.H., Chadwick D.R., Coutinho J. and Trindade H.(2011). Ammonia and greenhouse gas emission from slatted and solid floor in dairy cattle house: a scale model study. BiosystemEngeneering 109, 148-157.

Provolo G, Grimaldi D. and Riva E. (2012). Environmental assessment of livestock farms: a comparison of different methods to estimate emissions to air. International Conference RAGUSA SHWA 2012, Ragusa (Italy), 148-155.

Reidy B., Rhim B. and Menzi H. (2008). 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, 3266-3276.

Reinoso R. and Becares E. (2008). The occurrence of intestinal parasites in swine slurry and their removal in activated sludge plants. Bioresource Technology 99 (14), 6661-6665.

Rigolot C., Espagnol S., Pomar C. and Dourmad J.Y. (2010). Modelling of manure production by pigs and NH_3, N_2O and CH_4 emissions. Part I: animal excretion and enteric CH_4, effect of feeding and performance. Animal 4 (8), 1401-1412.

Rigolot C., Espagnol S., Pomar P., Hassouna M., Béline F., Paillat J. M. and Dourmad J.Y. (2010). Modelling of manure production by pigs and NH_3, N_2O and CH_4 emissions. Part II: effect of animal housing, manure storage and treatment practices. Animal 4 (8), 1413-1424.

Rohwer C. (1931). Evaporation from Free Water Surface. USDA Tech. Null. 217, 1-96.

Ross C.A., Scholefield D. and Jarvis S.C. (2002). A model of ammonia volatilisation from a dairy farm: An examination of abatement strategies. Nutrient Cycling in Agroecosystems 64, 273-281.

Rotz C.A. (2004). Management to reduce nitrogen losses in animal production. Journal of Animal Science 82, 119-137.

Sanz A., López M.T., Callejo J.A. and Parras, T. (2010). Balance de emisiones de GEI procedentes de la producción y el uso de pelets de Cynara cardunculus con fines térmicos. Consejería de Agricultura y Pesca, Junta de Andalucía, 42 pp.

SEPA (2012). Improving land quality. Scottish Environment Protection Agency, Stirling, UK. http://www.sepa.org.uk/land/agriculture/beef.aspx

SEPA (Personal Communication). Scottish Environment Protection Agency, Stirling, UK.

Shaw M.I., Beaulieu A.D. and Patience J.F. (2006). Effect of diet composition on water consumption in growing pigs. Journal of animal science 84, 3123-3132.

Sneath R.W., Beline F., Hilhorst M.A. and Peu P. (2006). Monitoring GHG from manure stores on organic and conventional dairy farms. Agriculture, Ecosystems and Environment, 112(2-3),122-128.

Søgaard H.T., Sommer S.G., Hutchings N.J., Huijsmans J.F.M., Bussink D.W. and Nicholson F. (2002). Ammonia volatilization from field-applied animal slurry - the ALFAM model. Atmospheric Environment 36, 3309-3319.

Turner C. and Williams S.M. (1999). Laboratory scale inactivation of African swine fever virus, and swine vesicular disease virus in pig slurry. Journal of Applied Microbiology, 87, 447-453.

Van Kessel J.S., Pachepsky Y.A., Shelton D.R. and Karns J.S. (2007). Survival of Escherichia coli in cowpats in pasture and in laboratory conditions. Journal of Applied Microbiology 103 (4), 1122-1127.

Vanotti M.B., Millner P.D., Hunt P.G. and Ellison A.Q. (2005). Removal of pathogen and indicator microorganisms from liquid swine manure in multi-step biological and chemical treatment. Bioresource Technology 96 (2), 209-214.

Vedrenne F. (2007). Etude des processus de dégradation anaérobie et de production de méthane au cours du stockage des lisiers. Thèse de doctorat de l'ENSA de Rennes, 232 pp.

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

Vermorel M., Jouany J.P., Eugène M., Sauvant D., Noblet J. and Dourmad J.Y. (2008). Evaluation quantitative des émissions de méthaneentérique par les animauxd'élevage. INRA Productions Animales 21, 403-418.

Ward D. and Mckague K. (2007). Water requirements of livestock. Agricultural Engineering Animal Science 7, 23.

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

Webb J., Ryan M., Anthony S.G., Brewer A., Laws J., Aller M.F. and Misselbrook T.H. (2006). Cost-effective means of reducing ammonia emissions from UK agriculture using the NARSES model. Atmospheric Environment 40, 7222–7233.

Zhang Y., Collins A.L. and Gooday R.D. (2012). Application of the FARMSCOPER tool for assessing agricultural diffuse pollution mitigation methods across the Hampshire Avon Demonstration Test Catchment, UK. Environmental science & policy 24, 120-131.

APPENDIX A: Swine

Nomenclature

DAYS	Number of days in each phase per sow and year	$MEAT_{PRE}$	kg of meat produced in pre-fattening per year
DAYSG	Number of gestating days per sow and year	MOR_F	% Mortality in fattening
DAYSL	Number of lactating days per sow and year	MOR_{PRE}	% Mortality in pre-fattening
DAYSR	Number of replacement days per sow and year	N	Number of animals produced (kg)
DW_F	Average weight of dead animals during fattening (kg)	NFAT	Number of fattening pigs produced per year
DWG_F	Fattening: Daily Weight Gain (g/animal and day)	NGILTS	Number of gilts
DWG_{PRE}	Pre-fattening: Daily Weight Gain (g/animal and day)	NLAC	Number of lactating sows
DW_{PRE}	Average weight of dead animals during pre-fattening (kg)	NP	Number of piglets per housed sow and year
FCAP	Fattening capacity (number of animals/year)	NPRE	Number of Pre-fattening piglets produced per year
FCD	Feed consumption per sow and day (kg)	$NPRE_F$	Number of prefattened pigs to be fattened per year. If NPRE=0 then $NPRE_F$=FCAP. If FCAP> or =NPRE, then $NPRE_F$=NPRE. If FCAP<NPRE, then $NPRE_F$=FCAP
FCR	Feed conversion rate (kg feed/ kg live weight)	$NPRE_S$	Number of prefattened pigs to be sold per year. If NPRE=0 then $NPRE_S$=0, if not, $NPRE_S$=NPRE-$NPRE_F$
FI	kg of feed intake pre-fattening and fattening	NSOWS	Number of sows housed (including gilts)
FIS	kg of feed intake sows	NWP	Number of weaned piglets per year
FSER	First service (months)	NWP_{PRE}	Number of weaned piglets to be pre-fattened per year. If NWP=0 then NW_{PPRE}=PRECAP. If NWP < or = PRECAP, then NWP_{PRE} = NWP. If NWP>PRECAP, then NWP_{PRE}=PRECAP
FW	Final weight (kg)	PB_F	Empty time between batches in fattening (days)
FW_F	Final weight fattening (kg)	PB_{PRE}	Period between batches pre-fattening (days)
FW_{PRE}	Final weight pre-fattening piglets (kg)	PL_F	Number of fattening places
IW	Initial weight (kg)	PL_{PRE}	Number of pre-fattening places
IW_F	Initial weight fattening (kg)	PRECAP	Pre-fattening capacity (number of animals/year)
IW_{PRE}	Initial weight pre-fattening (kg)	REP	% of replacement gilts
L_{SOW}	Number of litters per sow and year	W_{AGE}	Aged of the weaned piglets (days)
$MEAT_F$	kg of meat produced in fattening per year		

References

Aguilar M., Abaigar A., Merino P., Estellés F. and Calvet S. (2010). Estudio de un lavador de aire en una nave de gestación porcina. Navarra Agraria, 183, 41-46.

Albar J. and Granier R. (1996). Incidence du taux azoté de l'aliment sur la consommation d'eau, la production de lisier et les rejets azotés en engraissement. Journées Rech. Porcine en France, 28, 257 - 266.

Chosson C., Granier R., Maigne A., Bouby A. and Mongin J.P. (1998). Reduction du volumen de lisier produit par un porc a l`engrais. TechniPorc , 11, 27-41.

Collin A., Van Milgen J. and Le Dividich J. (2001). Modelling the effect of high, constant temperature on food intake in young growing pigs. Animal Science, 72, 519-527.

Corpen (2003). Estimation des rejets d`azote-phosphore-potassium-cuivre et zinc des porcs. Influence de la conduite alimentaire et du mode de logement des animaux sur la nature et la gestion des déjections produites, 41 pp.

EC (2003). Reference Document on Best Available Techniques for intensive Rearing of Poultry and pigs.Integrated Pollution Prevention and Control (IPPC), July 2003, 341 pp.

Granier R., Guingand N. and Massabie P. (1996). Influence du niveau d'hygrométrie, de la température et du taux de renouvellement de l'air sur l'évolution des teneurs en ammoniac. Journé es de la Recherche Porcine en France 28, 209-216.

Guillou D., Dourmand J.Y. and Noblet J., (1993). Influence de l`alimentation, du stade physiologique et des performances sur les rejets azotés du porc à l`engrais de la truie et du porcelet. Journées Rech. Porcine en France, 25, 307-314.

INRA (1984). L`alimentation des animaux monogastriques : porc, lapin, volailles. INRA, Paris (France), 282 pp.

Institut de L'Elevage, Ifip and Itavi (2010). Guide des Bonnes Pratiques Environnementales d'Élevage, 303 pp. http://www.rmtelevagesenvironnement.org/pdf/bpe_avi_porcs_bovins.pdf

Latimier P., Gallard F. and Corlouër A. (1996). Actualisation des volumes et des quantités d'azote, de phosphore et de potasse rejetés dans le lisier par un élevage naisseur-engraisseur. Journées Recherche Porcine en France, 28, 241-248.

MAGRAMA (2014). Evaluación de Técnicas de Reducción de Emisiones en Ganadería. Sectores de Porcino y Avicultura de Carne y Puesta. Ministerio de Agricultura, Alimentación y Medio Ambiente, Madrid, 114 pp.

MARM (2008). Metodología para la estimación de las emisiones a la atmósfera del sector ganadero para el inventario nacional de emisiones. Ministerio de Medio Ambiente y Medio Rural y Marino, España, 221 pp.

MARM (2010a). Guía de Mejores Técnicas Disponibles del Sector Porcino. Ministerio de Medio Ambiente y Medio Rural y Marino, 129 pp. http://www.magrama.gob.es/es/ganaderia/publicaciones.

MARM (2010b). Guía de Mejores Técnicas Disponibles del Sector de la Avicultura de Puesta. Ministerio de Medio Ambiente y Medio Rural y Marino, España, 86 pp. http://www.magrama.gob.es/es/ganaderia/publicaciones.

Massabie P. (2001). L'abreuvement des porcs.Techniporc, 24(6), 9-14.

Massabie P. (2001). L'abreuvement des porcs.Techniporc, 24(6), 9-14.

Rigolot C., Espagnol S., Pomar C. and Dourmad J.Y. (2010a). Modelling of manure production by pigs and NH_3, N_2O and CH_4 emissions. Part I: animal excretion and enteric CH_4, effect of feeding and performance. Animal 4 (8), 1401-1412.

Rigolot C., Espagnol S., Pomar P., Hassouna M., Béline F., Paillat J. M. and Dourmad J.Y. (2010b). Modelling of manure production by pigs and NH_3, N_2O and CH_4 emissions. Part II: effect of animal housing, manure storage and treatment practices. Animal 4 (8), 1413-1424.

Shaw M.I., Beaulieu A.D. and Patience J.F. (2006). "Effect of diet composition on water consumption in growing pigs". Journal of animal science, 84, 3123-3132.

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Animal production

Equation A1: Number of lactating days per sow and year (DAYSL)

$DAYSL = (W_{AGE} + 7) * L_{SOW}$

Equation A2: Number of gestating days per sow and year (DAYSG)

DAYSG = 365-DAYSL

Equation A3: Number of replacement days per sow and year (DAYSR)

DAYSR = ((REP/100)/12)*FSER*365)

Equation A4: Number of gilts (NGILTS)

NGILTS = (NSOWS*(REP/100))/FSER

Equation A5: Number of lactating sows (NLAC)

NLAC = NSOWS*((W_{AGE}*L_{SOW})/365)

Equation A6: Number of gestating sows (NGES)

NGES = NSOWS-NLAC

Equation A7: Number of weaned piglets per year (NWP)

NWP = NSOWS * NP

Equation A8: Pre-fattening capacity (PRECAP, number of animals/year)

PRECAP = (365/(((((FW_{PRE}-IW_{PRE})*1000)/DWG_{PRE})+PB_{PRE}))*PL_{PRE}

Equation A9: Number of Pre-fattening piglets produced per year (NPRE)

NPRE = NWP_{PRE}-(NWP_{PRE}*(MOR_{PRE}/100))

Equation A10: Fattening capacity (FCAP, number of animals/year)

FCAP = (365/ (((FW_F-IW_F)*1000/DWG_F) +PB_F))*PL_F)

Equation A11: kg of meat produced in pre-fattening per year ($MEAT_{PRE}$)

$MEAT_{PRE}$ = (($NPRE_F$*IW_F) + ($NPRE_S$*FW_{PRE}) + (NWP_{PRE}*(MOR_{PRE}/100)* DW_{PRE}))-(NWP_{PRE}*IW_{PRE}))

Equation A12: Number of fattening pigs produced per year (NFAT)

NFAT = $NPRE_F$-($NPRE_F$ * MOR_F/100)

Equation A13: kg of meat produced in fattening per year ($MEAT_F$)

$MEAT_F$ = ((FW_F*NFAT) + ($NPRE_F$*(MOR_F/100)* DW_F))-($NPRE_F$*IW_F)

Nutrient balance

Equation A14: kg of feed intake pre-fattening and fattening (FI)

FI = (FW-IW) * FCR * N

N: Number of animals produced (kg) (NPRE Equation A9, NFAT Equation A12)

Equation A15: kg of feed intake sows (FIS)

FIS = (FCD*DAYS)*NSOWS

DAYS: Number of days in each phase per sow and year (DAYSL Equation A1, DAYSG Equation A2, DAYSR Equation A3)

Table A1. K excretion rates. CORPEN, (2003)

	Reference crude protein in the feed (%)	K excreted (g/sow day or g/kg LW*)	
		Protein ≥ Reference	Protein < Reference
Gestation	16.5	19.64	17.28
Lactation	16.5	36.04	31.71
Gilts	16.5	19.64	17.28
Prefattening (prestarter feed)	21	13.96*	13.27*
Prefattening (starter feed)	19		
Fattening (growing feed)	17.5	22.37*	19.46*
Fattening (finishing feed)	17.5		

Table A2. N, P, Cu and Zn retention rates.

	Sows			Prefattening	Fattening	Source
	Gestation	Lactation	Gilts			
	g/sow day*			g/kg live weight		
N	75*	63*	80*	25	25	Guillou et al. (1993) Corpen (2003)
P	2.28	11.9	2.08	5.3	5.3	INRA (1984) Corpen (2003)
Cu	0.22	0.22	0.22	0.0011	0.0011	Corpen (2003)
Zn	4.36	4.36	4.36	0.0218	0.0218	Corpen (2003)

* N retention in sows in % of N consumed

Equation A16: kg DM or OM excretion ($X_{Excretion}$)

$X_{Excretion}$ = Feed * X_{Feed} * (1-dC_X)

Feed: Feed intake (kg) (FI Equation A14, FIS Equation A15)

X_{Feed}: DM or OM feed content (kg/kg)

dC_X: Digestibility coefficient (Equation A17)

Equation A17: Digestibility coefficients for DM or OM (dC_X)

dC_{OMSow} = ((((19.14*ED)-(0.28*F*10)-(1.63*10*C))/(MS*10))+0.674)/((MS-C)/MS)

dC_{OMPig} = ((((14.69*ED)-(0.5*F*10)-(1.54*10*C))/(MS*10))+0.744)/((MS-C)/MS))

Dc_{DMSow} = (((21.57*ED)-(0.26*F*10)-(1.21*C*10))/(MS*10))+0.63)

Dc_{DMPig} = (((17.94*ED)-(0.49*F*10)-(1.09*C*10))/(MS*10))+0.709)

ED: Digestible Energy feed (MJ/kg)

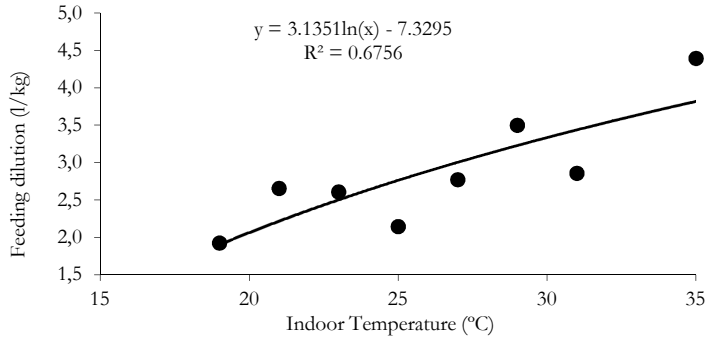
F: Neutral Detergent Fibre feed (%)

C: Mineral Matter feed (%)

MS: Dry Matter Gestating feed (%)

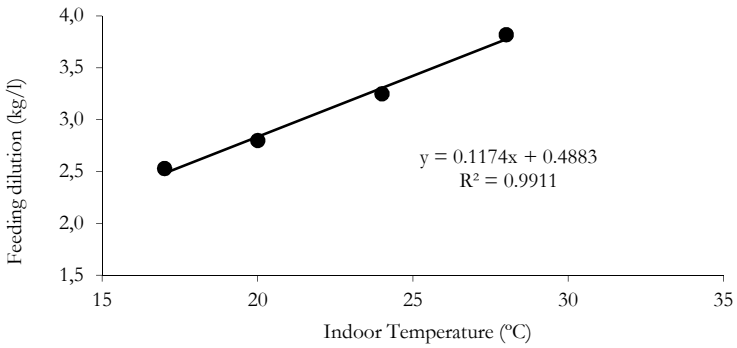
Water consumption

Fig. A1. Relation between indoor temperature and feeding dilution (nipple drinker in a cup or biting nipple). Prefattening. Adapted from Collin et al. (2001).



Note: In the case of wet feeder, the dilution considered is the 73% of the value obtained for the other drinkers (INTIA).

Fig. A2. Relation between indoor temperature and feeding dilution (nipple drinker in a cup or biting nipple). Fattening. Adapted from Massabie (2001).



Note: In the case of wet feeder, the dilution considered is the 73% of the value obtained for the other drinkers (INTIA).

Table A3. Reference values for feeding dilution in fattening pigs with liquid feeding. Regionalizable default values. INTIA.

Feeding Dilution (l/kg)											
Jan	Feb	Mar	April	May	June	July	Aug	Sept	Oct	Nov	Dec
2.5	2.5	2.6	2.6	2.7	2.9	3	3.1	2.9	2.7	2.6	2.5

Table A4. Protein effect on drinking water. Prefattening and fattening pigs.

	Variation of drinking water	Source
CP>22%	+5% per unit of CP over 22%	Shaw, Beaulieu and Patience (2006)
CP<14%	-2.7% per unit of CP under 14%	Albar & Granier (1996)

Table A5. Wasted water by animals. Prefattening and fattening pigs. Chosson et al. (1998) and Massabie (2001).

Type of drinker	Wasted water (l)	
	Prefattening	Fattening
Nipple drinker in a cup	0.1*Drinking water (l)	0.1*Drinking water (l)
Wet feeder	0	0
Biting nipple	0.3* Drinking water (l)	0.25*Drinking water (l)
Liquid feeding		0

Table A6. Cleanning water. Prefattening and fattening pigs. Latimier et al. (1996).

	Cleaning water (l/per animal)
Prefattening pig sold	14
Prefattening pig to fattening	19
Fattening pig	30

Table A7. Indoor temperature at swine housing. Regionalizable default values. INTIA Database

	Indoor Temperature (ºC)											
	Jan	Feb	Mar	April	May	June	July	Aug	Sept	Oct	Nov	Dec
Gestating	20	20	21	22	23	24	26	26	23	22	21	20
Lactating	21	21	21.5	22	23	24	26	26	24	23	21	21
Prefattening	26	26	26	26.5	26.5	27	27	27	26.5	26.5	26	26
Fattening	22	22	23	24	24	25	27	26	25	23	22	22

Gas Emission

Equation A18: Variation Factor associated to N dilution in the slurry (VF_{Ndil})

$VF_{Ndil}=1+1.27*(S_{TAN}-0.51)$

S_{TAN}: Total Ammoniacal N concentration in the slurry (mol/l). Calculated from urinary N according to Rigolot et al. (2010a).

Equation A19: Variation Factor associated to slurry temperature (VF_{Temp})

$VF_{Temp}=1+0.053*(Temp_{Effluent}-0.22)$

$Temp_{Effluent}$: Slurry temperature (ºC) estimated from ambient temperature (Table A7) using a relationship derived from the study of Granier et al. (1996).

Equation A20: Variation Factor associated to ventilation rate (VF_{Vent})

$VF_{Vent}=1+0.636*(Vent_{rate}-0.6)$

$Vent_{rate}$: Ventilation rate (m3/h/kg LW) (Table A8)

Table A8. Reference ventilation rates at swine housing. Regionalizable default values. INTIA Database.

	Ventilation Rate (m3/h/kg LW)											
	Jan	Feb	Mar	April	May	June	July	Aug	Sept	Oct	Nov	Dec
Gestating	0.2	0.2	0.2	0.6	0.6	0.9	0.9	0.9	0.9	0.6	0.2	0.2
Lactating	0.23	0.23	0.23	0.64	0.64	0.95	0.95	0.95	0.95	0.64	0.23	0.23
Prefattening	0.43	0.43	0.43	0.85	0.85	1.28	1.28	1.28	1.28	0.85	0.43	0.43
Fattening	0.3	0.3	0.3	0.6	0.6	0.9	0.9	0.9	0.9	0.6	0.3	0.3

Equation A21: Variation Factor of gas X associated to type of floor (VF_{Xfloor})

$$VF_{Xfloor} = 1 - \left[\sum_{i=1}^{3}\left(\frac{RF_{Xfloor_i}}{100} * \frac{S_{floor_i}}{100}\right)\right]$$

RF_{Xfloor}: Gas X emission reduction associated to the type of floor (Table A9)

S_{floor} : Percentage of each type of floor (%)

Please note that up to 3 different types of floor can be described for each type of animal.

Equation A22: Variation Factor of gas X associated to frequent slurry removal (VF_{Xfreq})

$$VF_{Xfreq} = 1 - \left[\sum_{i=1}^{3}\left(\frac{RF_{Xfreq_i}}{100} * \frac{S_{freq_i}}{100}\right)\right]$$

RF_{Xfreq}: Emission reduction of gas X associated to the slurry removal frequency (Table A10, Equation A26)

S_{freq}: Percentage of surface with that slurry management (%)

Please note that up to 3 different types of slurry frequencies can be described for each type of animal.

Equation A23: Variation Factor of gas X associated to other mitigation techniques ($VF_{NH3Other}$)

$$VF_{XOther} = \prod_{i=1}^{3}\left[1 - \left(\frac{RF_{XOther_i}}{100} * \frac{S_{Other_i}}{100}\right)\right]$$

RF_{XOther}: Emission reduction of gas X associated to other mitigation techniques (Table A11)

S_{Other}: Percentage of surface with that technique (%)

Please note that up to 3 different types of techniques can be described for each type of animal.

Table A9. Effect of type of floor on swine emissions at housing. Regionalizable default values.

Type of animals housed	Type of floor	% Emission reduction comparing to the reference system: fully slatted floor of concrete slats			Source
		$RF_{NH3floor}$	$RF_{N2Ofloor}$	$RF_{CH4floor}$	
Gestating sows	Partly slatted floor and reduced manure pit	50	--	28	MARM (2010a) MAGRAMA (2014)
Farrowing sows	Partly slatted floor and reduced manure pit	34	--	28	EC (2003) MARM (2010a)
	Metallic Fully-slatted floor	15	--	--	Rigolot et al.(2010b)
	Plastic Fully-slatted floor	10	--	--	Rigolot et al.(2010b)
Prefattening	Partly slatted floor and reduced manure pit both sides	30	--	34	EC (2003) MARM (2010a)
	Metallic Fully-slatted floor	15	--	--	Rigolot et al.(2010b)
	Plastic Fully-slatted floor	10	--	--	Rigolot et al.(2010b)
Fattening	Partly slatted floor and reduced manure pit both sides	32.5	--	34	MARM (2010a)

-- No data available, no effect considered in calculations for the moment.

Table A10. Emission reduction at swine housing due to frequent slurry removal. Regionalizable default values.

Removal Frequency	% Emission reduction comparing to the reference system: more than one month		
	$RF_{NH3freq}$	$RF_{N2Ofreq}$	$RF_{CH4freq}$
Daily	35	41-83*	78.2-91.5 depending on the climatic region (Equation A26)
Weekly	20	41-83*	
2 Weeks	10	--	
Monthly	0	--	

Source: Rigolot et al. (2010b); IPPC (2006); MAGRAMA (2014)

*83 Gestating and Farrowing sows; 41 Prefattening and Fattening

-- No data available, no effect considered in calculations for the moment

Table A11. Other mitigation techniques at swine housing. Regionalizable default values.

Technique	% Emission reduction comparing to the reference system[1]			G[2]	F[3]	PF[4]	FT[5]	Source
	$RF_{NH3Other}$	$RF_{N2OOther}$	$RF_{CH4Other}$					
Wet scrubbers	74	-74	0	√				Aguilar et al. (2010) Institut de L'Elevage et al. (2010)
	-65	-74	0				√	
Fogging coolers	26	0	0	√				Institut de L'Elevage et al. (2010)
V shape pits	32	43	65		√			MARM (2010a) MAGRAMA (2014)
	60	27	65			√		
	20	--	50				√	

[1]Deep pit of rectangular section under fully-slatted floor with concrete slats and frequency of slurry removing over 1 month

[2] G: Gestating sows; [3] F: Farrowing sows; [4] PF: Prefattening;[5] FT: Fattening

-- No data available, no effect considered in calculations for the moment

Equation A24: Enteric Methane (E_{CH4Ent}, kg)

$$E_{CH4Ent} = (GE * Ym * 100^{-1}) * 55.65^{-1}$$

GE: Gross energy consumption (MJ) (Equation A25)

Ym: CH_4 conversion factor for each category of animal (%) (Table A12)

Equation A25: Gross energy consumption (GE, MJ)

$$GE = \frac{DE * FC}{dC_{OM}}$$

DE: Digestible energy content in the feed (MJ/kg)

FC: Feed consumption (kg) (Equation A14 and A15)

dC_{OM}: Organic matter digestibility coeficient (Equation A17)

Table A12. Methane conversion factor (Ym). Regionalizable default values. MARM (2008).

Animal Category	Ym (%)
Gilts	0.65
Farrowing sows	0.9
Gestating sows	1.05
Prefattening	0.28
Fattening	0.48

Equation A26. CH_4 Emission reduction associated to the slurry removal frequency ($RF_{CH4freq}$)

$$RF_{CH4freq} = \left[1 - \left(\frac{MCF_{<1}}{MCF_{>1}}\right)\right] * 100$$

MCF<1: Methane conversion factor (%) for pit storage below animal confinements below 1 month of duration and average monthly temperatures of the region in question (IPCC, 2006).

MCF>1: Methane conversion factor (%) for pit storage below animal confinements longer than 1 month and average monthly temperatures of the region in question (IPCC, 2006).

When frequency is over one month ($RF_{CH4freq}$ = 0).

APPENDIX B: Laying hen

Nomenclature

B_W	Number of weeks between batches	L_{RATE}	Mean laying rate (%)
EGG_W	Average egg weight (g)	MOR_L	% Mortality during laying
FCR_{LH}	Feed conversion rate (kg feed/kg egg)	NBL_{DAYS}	Number of days before laying per year
FCR_{RC}	Feed conversion rate (kg feed/kg rearing chicken)	NBL_W	Number of weeks before laying
$FEED_{LH}$	Feed consumption laying hens (kg)	NEGGS	Number of eggs produced per year
$FEED_{RC}$	Feed consumption rearing chicken (kg)	NL_{DAYS}	Number of laying days per year
FW_{RC}	Average final weight of rearing chicken (laying period start) (kg)	NLH	Number of laying hens
IL_{RATE}	Laying rate after pre-laying period (%)	NL_W	Number of weeks per laying cycle
IW_{RC}	Average initial weight of rearing chicken (kg)	NRC	Number of rearing chicken

References

Alberdi O., Arriaga H., Calvet S., Estellés F. and Merino P. (2016). Ammonia and greenhouse gas emissions from an enriched cage laying hen facility. Biosystems Engineering, 144, 1-12.

CORPEN (1996). "Estimation des Rejets d'azote par les elevages avicoles", 9 pp.

Corpen (2013). Estimation des rejets d`azote-phosphore-potassium-calcium-cuivre et zinc par les élevages avicoles. Mise à jour des références CORPEN-Volailles de 2006, 63 pp.

EC (2003). Reference Document on Best Available Techniques for intensive Rearing of Poultry and pigs.Integrated Pollution Prevention and Control (IPPC), July 2003, 341 pp.

Fabbri C., Valli L., Guarino M., Costa A. and Mazzotta V. (2007). Ammonia, methane, nitrous oxide and particulate matter emissions from two different buildings for laying hens. Biosystems Engineering, 97, 441-455.

MARM (2008). Metodología para la estimación de las emisiones a la atmósfera del sector ganadero para el inventario nacional de emisiones. Ministerio de Medio Ambiente y Medio Rural y Marino, España, 221 pp.

MARM (2010). Guía de Mejores Técnicas Disponibles del Sector de la Avicultura de Puesta. Ministerio de Medio Ambiente y Medio Rural y Marino, España, 86 pp. http://www.magrama.gob.es/es/ganaderia/publicaciones.

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Animal production

Equation B1. Number of laying days per year (NL_{DAYS})

NL_{DAYS}= ((NL_W*52)/(NBL_W+ NL_W+B_W))*7

Equation B2. Number of days before laying per year (NBL_{DAYS})

NBL_{DAYS}= ((NBL_W*52)/(NBL_W+NL_W+B_W))*7

Equation B3. Number of eggs produced per year (NEGGS)

NEGGS= (((1+(1-(MOR_L/100)))/2)*(NLH)*(L_{RATE}/100)*NL_{DAYS})+((NLH)*(IL_{RATE}/100)*NBL_{DAYS})

Equation B4. Number of rearing chicken (NRC)

NRC= NLH*(52/(NBL_W+NL_W+B_W))

Nutrient balance

Equation B5. kg Feed consumption laying hens ($FEED_{LH}$)

$FEED_{LH}$= NEGGS*(EGG_W/1000)*FCR_{LH}

Equation B6. kg Feed consumption rearing chicken ($FEED_{RC}$)

$FEED_{RC}$=NRC*(FW_{RC}-IW_{RC})*FCR_{RC})

Table B1. Nutrient retention. CORPEN (1996)* and CORPEN (2013).

	Crude Protein (g)	Phosphorus (g P)	Potassium (g K)	Zinc (mg)	Copper (mg)
1 kg of fresh eggs	115	2.15	1.2	12.6	0.9
1 kg of LW in laying hens	50*	4.8	2	25	3
1 kg of LW in rearing chickens	181.25	5.8	2	21.3	1.7

Table B2. Organic matter (VS) produced by the animals. Laying hens submodel. Adapted from MARM (2008).

Type of animal	VS excreted (kg MS/animal day)
Laying hen	0.0146
Rearing chicken	0.0071

Gas Emission

Table B3. Type of manure removal systems for laying hens and rearing chicken buildings. Effect on emissions. Regionalizable default values.

Type of removal system	Frequency manure removal	% Emission reduction comparing to the reference system*			Source
		RF_{NH3Rem}	RF_{N2ORem}	RF_{CH4Rem}	
Manure belt without drying system	Twice a week	65.5	--	--	MARM (2010b) EC (2003)
Manure belt without drying system	Twice a day	93	--	--	
Vertical tiered cages with manure belts and forced air drying	Once a week	58	--	--	
Vertical tiered cages with manure belts and whisk-forced air drying	Once a week	60	--	--	
Vertical tiered cages with manure belts and improved forced air drying	Every 5 days	79	--	--	
Mid deep pit*	>1Week	0	0	0	
Deep pit*	>1Week	0	0	0	
External drying system with perforated manure belts	Daily	70	--	--	

*Deep pit and Mid Deep pit (Reference System): Battery system with manure collected in a channel or pit under the cages and removed once every production cycle.
-- No data available, no effect considered in calculations for the moment

Equation B7: Variation Factor of gas X associated to type of manure removal system (VF_{XRem})

$$VF_{XRem} = 1 - \frac{RF_{XRem}}{100}$$

RF_{Xfreq}: Emission reduction of gas X associated to type of manure removal system (Table B3)

Table B4. VF_{Vent} values depending on ventilation rates (VR). [1]Adapted from Fabbri et al. (2007) [2] Alberdi et al. (2016)

Manure removal system	Pits[1]	Belts[2]
VR_{Lower} (m^3/h hen)	5.200	1.109
VF_{Vent} ($VR \leq VR_{Lower}$)	0.419	0.347
VR_{Upper} (m^3/h hen)	10.400	6.559
VF_{Vent} ($VR \geq VR_{Upper}$)	1.400	2.050
VF_{Vent}($VR_{Lower} \geq VR \geq VR_{Upper}$)	0.1885*(VR-8.28) + 1	0.3125*(VR-3.2) + 1

Table B5. Default values for ventilation rates (VR) depending on the type of removal system and frequency. Regionalizable default values.

	Ventilation rates laying hens (m^3/h hen)											
	Jan	Feb	Mar	April	May	June	July	Aug	Sept	Oct	Nov	Dec
Manure belt without drying system (Twice a week)	1.252	1.860	2.678	2.785	3.112	4.599	8.297	7.472	5.322	4.223	2.543	2.280
Manure belt without drying system (Twice a day)	1.252	1.860	2.678	2.785	3.112	4.599	8.297	7.472	5.322	4.223	2.543	2.280
Vertical tiered cages with manure belts and forced air drying (Once a week)	1.252	1.860	2.678	2.785	3.112	4.599	8.297	7.472	5.322	4.223	2.543	2.280
Vertical tiered cages with manure belts and whisk-forced air drying (Once a week)	1.252	1.860	2.678	2.785	3.112	4.599	8.297	7.472	5.322	4.223	2.543	2.280
Vertical tiered cages with manure belts and improved forced air drying (Every 5 days)	1.252	1.860	2.678	2.785	3.112	4.599	8.297	7.472	5.322	4.223	2.543	2.280
Mid deep pit	6.960	6.080	5.200	8.000	8.400	9.200	9.800	10.400	10.000	9.600	8.720	7.840
Deep pit	6.960	6.080	5.200	8.000	8.400	9.200	9.800	10.400	10.000	9.600	8.720	7.840
External drying system with perforated manure belts (Daily)	1.252	1.860	2.678	2.785	3.112	4.599	8.297	7.472	5.322	4.223	2.543	2.280
Defined by the user[1]	1.252	1.860	2.678	2.785	3.112	4.599	8.297	7.472	5.322	4.223	2.543	2.280

[1] Values per default considered for systems defined by the user.

Source: Mid deep pit and Deep pit from Fabbri et al. (2007). Belt systems from Alberdi et al. (2016)

	Ventilation rates rearing chicken (m^3/h chicken)[1]											
	Jan	Feb	Mar	April	May	June	July	Aug	Sept	Oct	Nov	Dec
Manure belt without drying system (Twice a week)	0.626	0.930	1.339	1.393	1.556	2.300	4.149	3.736	2.661	2.112	1.272	1.14
Manure belt without drying system (Twice a day)	0.626	0.930	1.339	1.393	1.556	2.300	4.149	3.736	2.661	2.112	1.272	1.14
Vertical tiered cages with manure belts and forced air drying (Once a week)	0.626	0.930	1.339	1.393	1.556	2.300	4.149	3.736	2.661	2.112	1.272	1.14
Vertical tiered cages with manure belts and whisk-forced air drying (Once a week)	0.626	0.930	1.339	1.393	1.556	2.300	4.149	3.736	2.661	2.112	1.272	1.14
Vertical tiered cages with manure belts and improved forced air drying (Every 5 days)	0.626	0.930	1.339	1.393	1.556	2.300	4.149	3.736	2.661	2.112	1.272	1.14
Mid deep pit	3.480	3.040	2.600	4.000	4.200	4.600	4.900	5.200	5.000	4.800	4.360	3.920
Deep pit	3.480	3.040	2.600	4.000	4.200	4.600	4.900	5.200	5.000	4.800	4.360	3.920
External drying system with perforated manure belts (Daily)	0.626	0.930	1.339	1.393	1.556	2.300	4.149	3.736	2.661	2.112	1.272	1.14
Defined by the user[2]	0.626	0.930	1.339	1.393	1.556	2.300	4.149	3.736	2.661	2.112	1.272	1.14

[1] Half of the ventilation rates in laying hens.

[2] Values per default considered for systems defined by the user.

APPENDIX C: Broilers

References

Al Homidan A., Robertson J.F. and Petchey, A.M. (1997). Effect of temperature, litter and light intensity on ammonia and dust production and broiler performance.British Poultry Science, 38, S5-S6.

Calvet S., Cambra-López M., Estellés F. and Torres A.G. (2011). Characterization of gas emission from a Mediterranean broiler farm. Poultry Science, 90, 534-542.

Corpen (2006). Estimation des rejets d`azote-phosphore-potassium-calcium-cuivre et zinc par les élevages avicoles. Influence de la conduite alimentaire et du mode de logement des animaux sur la nature et la gestion des déjections produites, 55 pp.

Corpen (2013). Estimation des rejets d`azote-phosphore-potassium-calcium-cuivre et zinc par les élevages avicoles. Mise à jour des références CORPEN-Volailles de 2006, 63 pp.

Da Borso F. and Chiumenti R. (1999). Poultry housing and manure management systems: recent developments in Italy as regards ammonia emissions. Proceedings of the 8th International Conference of the FAO ESCORENA Network on the Recycling of Agricultural, Municipal and Industrial Residues in Agriculture, Rennes (France), 15-21.

Elwinger and Svensson (1996). Effect of Dietary Protein Content, Litter and Drinker Type on Ammonia Emission from Broilers Houses. J. Agric. Engng Res. 64, 197-208.

MARM (2008). Metodología para la estimación de las emisiones a la atmósfera del sector ganadero para el inventario nacional de emisiones. Ministerio de Medio Ambiente y Medio Rural y Marino, España, 221 pp.

Nicholson F.A., Chambers B.J. and Walker A.W. (2004). Ammonia emissions from broiler litter and laying hen manure management systems. Biosystems Engineering, 89(2), 175-185.

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Animal production

Equation C1. kg live weight produced (LW)

LW= S*CY*AD*FW*((100-MOR)/100)

S: Surface (m^2)

CY: Cycles per year

AD: Animals per m^2

FW: Final weight (kg)

MOR: Mortality (%)

Nutrient balance

Table C1. Nutrients retained by the animals Broilers submodel. Regionalizable default values. CORPEN (2013)

Nutrient	Retention
Crude Protein (g/kg)	181.25
Phosphorus (mg/kg)	5800
Copper (mg/kg)	1.7
Zinc (mg/kg)	21.3

Table C2. Organic matter produced by the animals. Broilers submodel. Regionalizable default values. Adapted from MARM (2008)

Type of broiler	kg VS/animal day
Light chicken 1	0.0041
Light chicken 2	0.0074
Standard lightweight	0.0069
Standard heavyweight	0.0101

Manure production

The quantity of manure produced is calculated following these steps:

1. Calculation of reference values for manure production according to Table C3.

Table C3. Reference quantities of manure production. Broilers submodel. Regionalizable default values. Adapted from CORPEN (2006)

Type of broiler	kg manure/kg LW
Light chicken 1	0.66
Light chicken 2	0.75
Standard lightweight	0.87
Standard heavyweight	0.95

2. Calculation of reference values for the amount of bedding used considering values from Table C4.

Table C4. Reference quantities of bedding material. Broilers submodel. Regionalizable default values. INTIA database

Type of broiler	kg/m^2 cycle
Light chicken 1	3.9
Light chicken 2	5.8
Standard lightweight	6.5
Standard heavyweight	7

3. Calculation of the animal faeces as the difference of the previous values (Reference manure production minus reference bedding).
4. Calculation of farm manure production adding the amount of bedding material indicated by the user to the faeces calculated in the previous step.

Gas Emission

Equation C2. Variation Factor of gas X associated to type of drinkers (VF_{Xdrink})

$$VF_{Xdrink} = 1 - \frac{RF_{Xdrink}}{100}$$

RF_{Xdrink}: Emission reduction of gas X associated to type of drink (Table C5)

Table C5. Effect of type of drinkers on broilers emissions at housing. Regionalizable default values.

Type of drinker	% Emission reduction comparing to the reference system: nipple drinker			Source
	$RF_{NH3floor}$	$RF_{N2Ofloor}$	$RF_{CH4floor}$	
Nipple drinkers	0	--	--	
Drinkers with a drip-cup	40	--	--	Da Borso & Chiumenti (1999)
Water troughs	-30	--	--	Da Borso & Chiumenti (1999) and Nicholson et al. (2004)
Round drinkers	-30	--	--	

-- No data available, no effect considered in calculations for the moment.

Equation C3. Variation Factor of gas X associated to bedding thickness (VF_{Xthick})

$$VF_{Xthick} = 1 - \frac{RF_{Xthick}}{100}$$

RF_{Xdrink}: Emission reduction of gas X associated to bedding thickness. Default value for $RF_{NH3thick}$ 27% (Al Homidan et al., 1997) if the thickness is over 4.5 cm. Rest of situations, RF_{Xthick} =0.

Equation C4. Variation Factor of gas X associated to type of bedding material (VF_{Xbed})

$$VF_{Xbed} = 1 - \left[\sum_{i=1}^{2}\left(\frac{RF_{Xbed_i}}{100} * \frac{Q_i}{Q_{total}}\right)\right]$$

RF_{Xbedi}: Emission reduction of gas X associated to bedding material i. Per default, the model considers that the type of material has no effect on emission (Nicholson et al., 2004, Elwinger & Svensson, 1996), RF_{Xbed} =0. This can be changed by users.

Q_i: amount of bedding material i (kg).

Q_{total}: amount of total bedding material (kg)

Please note that up to 2 different types of bedding material can be selected.

Table C6. VF_{Temp} values depending on indoor temperature (T). Adapted from Calvet et al. (2011).

	Jan-March	April-June	Jul-Sept	Oct-Dec
T_{Lower} (ºC)	25	25	25	24
VF_{Temp} if $T \leq T_{Lower}$	0.83	0.83	0.77	0.79
T_{Upper} (ºC)	28	28	29	27
VF_{Temp} if $T \geq T_{Upper}$	1.20	1.14	1.18	1.16
VF_{Temp} if $T_{Lower} \geq T \geq T_{Upper}$)	0.123*(T-26.36)+1	0.102*(T-26.66)+1	0.102*(T-27.22)+1	0.123*(T-25.74)+1

Table C7. Average indoor temperature in animal (T, ºC). Regionalizable default values. INTIA database.

	Jan	Feb	Mar	April	May	June	July	Aug	Sept	Oct	Nov	Dec
Light chicken 1	26.36	26.36	26.36	26.32	26.32	27.35	27.35	27.15	27.15	25.43	25.43	26.36
Light chicken 2	26.36	26.36	26.36	26.32	26.32	27.35	27.35	27.15	27.15	25.43	25.43	26.36
Standard lightweight	26.36	26.36	26.36	26.32	26.32	27.35	27.35	27.15	27.15	25.43	25.43	26.36
Standard heavyweight	26.36	26.36	26.36	26.32	26.32	27.35	27.35	27.15	27.15	25.43	25.43	26.36

APPENDIX D: Dairy cattle

Nomenclature

$CALF_{nº}$	Number of calves in the herd
$CIFS_{conc}$	Nutrient concentration (N, P, K, Cu, Zn) of the concentrate saved (% or mg/kg)
$CIFS_{dmi}$	Dry matter intake saved by computerized individual feeding system (kg/cow/day)
$CIFS_{nut}$	Nutrient intake (N, P, K, Cu, Zn) saved by computerized individual feeding system (g/day)
COW_{milk}	Mean number of cows milking
COW_{milk_ratio}	Ratio of milking cows in the herd
$COW_{nº}$	Number of cows in the herd
$Days_{month}$	Days per month
DRY_{cycle}	Mean duration of dry period (days/year)
$HEIFER_{nº}$	Number of heifers in the herd
INT_{factor}	Percentage of nutrient intake increase (%)
$MILK_{base}$	Milk yield equivalent to 6000 kg/cow/year
$MILK_{cow}$	Mean annual milk yield (kg/cow/year)
$MILK_{cycle}$	Mean duration of milking period (days/year)
$MILK_{farm}$	Milk yield at farm level
$NUTCIFS_{save}$	Nutrient (N, P, K, Cu, Zn) saved by computerized individual feeding system
$NUTINT_{base\_calf}$	Default nutrient intake values reported by Corpen (1999) for calves (g/day)
$NUTINT_{base\_dc}$	Default nutrient intake values reported by Corpen (1999) for dry cows (g/day)
$NUTINT_{base\_heifer}$	Default nutrient intake values reported by Corpen (1999) for heifers (g/day)
$NUTINT_{base\_mc}$	Default nutrient intake (N, P, K, Cu, Zn) values reported by Corpen (1999) for milking cows (g/day)
$NUTINT_{calf}$	Nutrient (N, P, K, Cu, Zn) intake by calves
$NUTINT_{dc}$	Nutrient (N, P, K, Cu, Zn) intake by dry cows
$NUTINT_{heifer}$	Nutrient (N, P, K, Cu, Zn) intake by heifers
$NUTINT_{mc}$	Nutrient (N, P, K, Cu, Zn) intake by milking cows
NUT_{meat}	Default nutrient retention values reported by Corpen (1999) for calves (g or mg/kg body weight gain)
NUT_{milk}	Default nutrient retention values reported by Corpen (1999) for milking cows (g or mg/kg milk)
$NUTPFS_{save}$	Nutrient (N, P, K, Cu, Zn) saved by phase feeding system
$NUTRET_{calf}$	Nutrient (N, P, K, Cu, Zn) retention by calves
$NUTRET_{heifer}$	Nutrient (N, P, K, Cu, Zn) retention by heifers
$NUTRET_{mc}$	Nutrient (N, P, K, Cu, Zn) retention by milking cows
PFN_{nut}	Nutrient intake (N, P, K, Cu, Zn) saved by phase-feeding nutrition (g/day)
PFS_{dmi}	Dry matter intake saved by phase feeding system (kg/cow/day)
SLU_{bal}	Slurry production at dairy facilities
SLU_{bmp}	Slurry production/abatement due to BAT installation (m^3/month)
SLU_{clean}	Slurry production due to barn cleaning operations (m^3/month)
$SLU_{default}$	Default slurry production by type of facility and animal (m^3/cow/month)
SLU_{parlor}	Slurry production from wastewater of milking parlors (m^3/month)
TMR_{conc}	Nutrient concentration (N, P, K, Cu, Zn) of TMR diets (% or mg/kg)
WAT_{bal}	Water balance at dairy facilities
WAT_{bmp}	Water consumption/abatement due to BAT installation (m^3/year)
WAT_{cattle}	Water consumption by dairy cattle (adult cows and young cattle) (m^3/year)
WAT_{clean}	Water consumption due to barn cleaning operations (m^3/year)
WAT_{parlor}	Water consumption of milking parlors (m^3/year)
$Wcalf_{1year}$	Weight of calves 1 year (kg)
$Wcalf_{2year}$	Weight of heifers 2 year (kg)
$Wcalf_{birth}$	Weight of calves at calving (kg)

References

Corpen (1999). Estimation des flux d'azote, de phosphore et de potassium associés aux vaches laitières et à leur système fourrager Influence de l'alimentation et du niveau de production, 18 pp.

Institut de L'Elevage, Itavi, ITCF and ITP (2001). Fertiliser avec les engrais de ferme, 104 pp.

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Animal production

Equation D1. Milk yield at farm level ($MILK_{farm}$)

$MILK_{farm} = MILK_{cow} * COW_{milk}$

Equation D2. Ratio of milking cows in the herd (COW_{milk_ratio})

$COW_{milk\_ratio} = MILK_{cycle} / (DRY_{cycle} + MILK_{cycle})$

Equation D3. Number of milking cows (COW_{milk})

$COW_{milk} = COW_{nº} * COW_{milk\_ratio}$

Nutrient balance

Equation D4. Nutrient (N, P, K, Cu, Zn) intake by milking cows ($NUTINT_{mc}$)

$NUTINT_{mc} = ((NUTINT_{base\_mc} * (1+(((MILK_{cow} - MILK_{base})/1000) * INT_{factor}/100)) - (CIFS_{nut} + PFN_{nut})) * COW_{milk} * Days_{month}$

Equation D5. Nutrient (N, P, K, Cu, Zn) intake by dry cows ($NUTINT_{dc}$)

$NUTINT_{dc} = NUTINT_{base\_dc} * (COW_{nº} - COW_{milk}) * Days_{month}$

Equation D6. Nutrient (N, P, K, Cu, Zn) intake by calves ($NUTINT_{calf}$)

$NUTINT_{calf} = NUTINT_{base\_calf} * CALF_{nº} * Days_{month}$

Equation D7. Nutrient (N, P, K, Cu, Zn) intake by heifers ($NUTINT_{heifer}$)

$NUTINT_{heifer} = NUTINT_{base\_heifer} * HEIFER_{nº} * Days_{month}$

Equation D8. Nutrient (N, P, K, Cu, Zn) saved by computerized individual feeding system (CIFS) ($NUTCIFS_{save}$)

$NUTCIFS_{save} = CIFS_{dmi} * CIFS_{conc}$

Equation D9. Nutrient (N, P, K, Cu, Zn) saved by phase feeding system (PFS) ($NUTPFS_{save}$)

$NUTPFS_{save} = PFS_{dmi} * TMR_{conc}$

Equation D10. Nutrient (N, P, K, Cu, Zn) retention by milking cows ($NUTRET_{mc}$)

$NUTRET_{mc} = NUT_{milk} * MILK_{cow} * COW_{milk} * Days_{month} / 365$

Equation D11. Nutrient (N, P, K, Cu, Zn) retention by calves ($NUTRET_{calf}$)

$NUTRET_{calf} = NUT_{meat} * (Wcalf_{1year} - Wcalf_{birth}) * CALF_{nº} * Days_{month} / 365$

Equation D12. Nutrient (N, P, K, Cu, Zn) retention by heifers ($NUTRET_{heifer}$)

$NUTRET_{heifer} = NUT_{meat} * (Wcalf_{2year} - Wcalf_{1year}) * HEIFER_{nº} * Days_{month} / 365$

Water balance

Equation D13. Water balance at dairy facilities (WAT_{bal})

$WAT_{bal} = WAT_{cattle} + WAT_{parlor} + WAT_{clean} + WAT_{bmp}$

Table D1. Water consumption related to the type of milkin parlor. Adapted from Institut d'Élevage et al. (2001).

Type of milking parlor	Grey Water (m^3/month)	Wash Water (m^3/month)	Total Wash Water (m^3/month)
Herringbone Double Line 2x3	10	10	20
Herringbone Double Line 2x4	12	12	24
Herringbone Double Line 2x5	14	14	28
Herringbone Double Line 2x6	16	18	34
Herringbone Double Line 2x8	20	24	44
Herringbone Double Line 2x10	27	29	56
Herringbone Double Line 2x12	32	34	34
Parallel 2x8	29	24	53
Parallel 2x10	28	29	57
Parallel 2x12	45	34	79
Herringbone Intermediate Line 2x3	9	6	15
Herringbone Intermediate Line 2x4	11	8	19
Herringbone Intermediate Line 2x5	13	9	22
Herringbone Intermediate Line 2x6	15	10	25
Robot	15	5	20

Manure production

Equation D14. Slurry production at dairy facilities (SLU_{bal})

$SLU_{bal} = SLU_{default} + SLU_{parlor} + SLU_{clean} + SLU_{bmp}$

Table D2. Manure production (slurry and/or solid manure) by adult cows. Adapted from Institut d'Élevage et al. (2001).

Type of facility	Solid manure (t/cow/month)	Slurry (t/cow/month)
Tie-stall and Slatted Floor	0.00	1.67
Tie-stall and Gutter System	0.67	0.84
Freestall and Deep Litter Whole Farm	1.50	0.00
Freestall and Deep Litter and Mechanical Removal System	0.80	1.00
Freestall and Deep Litter and Slatted Floor	0.80	1.00
Freestall and Concrete Floor with Slope	1.20	0.33
Freestall and Deep Litter and Yard	1.40	0.00
Freestall and Cubicles	0.00	1.67

Table D3. Manure production (slurry and/or solid manure) by calves. Adapted from Institut d'Élevage et al. (2001).

Type of facility	Solid manure (t/cow/month)	Slurry (t/cow/month)
Freestall and Deep Litter Whole Farm	0.38	0.00
Freestall and Deep Litter and Mechanical Removal System	0.23	0.17
Freestall and Deep Litter and Slatted Floor	0.23	0.17
Freestall and Concrete Floor with Slope	0.30	0.09
Freestall and Deep Litter and Yard	0.30	0.00
Freestall and Cubicles	0.00	0.42

Table D4. Manure production (slurry and/or solid manure) by heifers. Adapted from Institut d'Élevage et al. (2001).

Type of facility	Solid manure (t/cow/month)	Slurry (t/cow/month)
Freestall and Deep Litter Whole Farm	0.60	0.00
Freestall and Deep Litter and Mechanical Removal System	0.37	0.27
Freestall and Deep Litter and Slatted Floor	0.37	0.27
Freestall and Concrete Floor with Slope	0.47	0.14
Freestall and Deep Litter and Yard	0.50	0.00
Freestall and Cubicles	0.00	0.67

Table D5. Default amount of bedding considered by adult cows.

Type of facility	Default bedding (kg/cow/day)
Tie-stall and Slatted Floor	0.0
Tie-stall and Gutter System	2.0
Freestall and Deep Litter Whole Farm	6.0
Freestall, Deep Litter and Mechanical Removal	6.0
Freestall, Deep Litter and Slatted Floor	0.5
Freestall and Concrete Floor with Slope	6.0
Freestall, Deep Litter and Yard	6.0
Freestall and Cubicles	0.0

Table D6. Default amount of bedding considered by young cattle.

Type of facility	Default bedding (kg/animal/day)
Freestall and Deep Litter Whole Farm	6.0
Freestall, Deep Litter and Mechanical Removal	6.0
Freestall, Deep Litter and Slatted Floor	0.5
Freestall and Concrete Floor with Slope	6.0
Freestall, Deep Litter and Yard	6.0
Freestall and Cubicles	0.0

Table D7. Type of bedding and composition

Materials	DM (%)	N (%)	P (%)	K (%)	Cu (mg/kg)	Zn (mg/kg)	OM (%)
Sand	99.50	0.00	0.08	1.90	4.30	0.00	0.50
Cereal Straw	91.70	0.54	0.07	0.84	3.70	9.10	82.80
Sawdust / Wood Shavings	88.90	0.24	0.06	0.29	6.00	9.00	97.00
Rice Hulls	86.50	0.47	0.08	0.47	2.60	18.60	74.30
Data Selected by User	88.90	0.24	0.06	0.29	6.00	9.00	97.00

APPENDIX E: Nutrients in the manure removed from buildings

Swine

The nitrogen content in the slurry from buildings is calculated in the model as the nitrogen excreted minus the nitrogen volatilized: ammonia and nitrous oxide.

The ammoniacal nitrogen respect to total nitrogen, the pH and the slurry density, are calculated considering slurry analysis and will vary according to the type of animals housed and the type of drinkers (Table E1).

Table E1. Data from slurry analysis. Regionalizable default values. INTIA's slurry analysis database.

Type of animals	Type of drinker	% TAN respect to Total N	Density (kg/m3)	Number of analysis
Fattening	Nipple drinker in a cup	69.8	1031	10
	Biting nipple	70.4	998	6
	Liquid feeding	71.1	1031	13
	Wet feeder	66.0	1038	18
Prefattening	Nipple drinker in a cup	63.4	1019	18
	Wet feeder	67.2	1035	3
	Biting nipple	65.3	1027	---*
Sows	All	73	1017	59

*The average of the previous values are considered

The model does not consider phosphorus, potassium, copper and zinc losses in the slurry pit, therefore the slurry content of these nutrients is equal to the quantity excreted by the animals.

Organic matter content in the removed slurry is calculated considering the quantity of carbon volatilized in the pit and contemplating that the methane emission from manure management (E_{CH4Man}, kg CH_4) represents the 66% of the total degradated carbon (C_{deg}, kg) (Equation E1).

$$C_{deg} = \frac{E_{CH4Man}}{0.66} * \frac{12}{16}$$ **(Equation E1)**

The model considers a factor of 1.724 to calculate the excreted carbon (C_{ex}, kg) from the organic matter excreted (VS_{ex}, kg) (Equation E2).

$$C_{ex} = \frac{VS_{ex}}{1.724}$$ **(Equation E2)**

The degradation rate of the organic matter will be the same as the carbon, so the final organic matter content (VS_{Slurry}, kg) can be calculated according to Equation E3.

$$VS_{Slurry} = VS_{ex} - \left(VS_{ex} * \frac{C_{deg}}{C_{ex}}\right)$$ **(Equation E3)**

Equations E4 and E5 show the calculation of the final dry matter (DM_{Slurry}, kg) and carbon content (C_{Slurry}, kg), respectively, being DM_{ex} the dry matter excreted (kg), E_{NH3} the NH_3 emission (kg) and E_{N2O} the N_2O emission (kg).

$$DM_{Slurry} = DM_{ex-}\left(VS_{ex} * \frac{C_{deg}}{C_{ex}}\right) - \left(E_{NH3} * \frac{14}{17}\right) - \left(E_{N2O} * \frac{28}{44}\right)$$ **(Equation E4)**

$$C_{Slurry} = C_{ex} - C_{deg}$$ **(Equation E5)**

The maximum methane producing capacity of the removed slurry (Bo_{Slurry}, m^3 CH_4/kg VS) is calculated from the initial maximum methane producing capacity (Bo_{ini}, m^3 CH_4/kg VS) according to Equation E6.

$$Bo_{Slurry} = \frac{(VS_{ex} * Bo_{ini}) - \left(E_{CH4Man} * \frac{22.4}{16}\right)}{VS_{Slurry}}$$ **(Equation E6)**

Laying hens

The nitrogen content in the manure from buildings is calculated in the model as the nitrogen excreted minus the nitrogen volatilized. The manure bulk density, the ammoniacal nitrogen content respect to total nitrogen and the pH are calculated considering manure analysis and can vary according to the type of manure removal system (Table E2).

Table E2. Manure composition default values depending on the type of removal system and frequency. Regionalizable default values.

Type of removal system	Frequency	Dry matter content (%)	%TAN respect to TN	kg OM/kg DM	pH	Bulk Density (t/m3)
Manure belt without drying system	Twice a week	25	47.715	0.653	7.629	0.820
Manure belt without drying system	Twice a day	25	47.715	0.653	7.629	0.820
Vertical tiered cages with manure belts and forced air drying	Once a week	37	47.715	0.653	7.629	0.752
Vertical tiered cages with manure belts and whisk-forced air drying	Once a week	37	47.715	0.653	7.629	0.752
Vertical tiered cages with manure belts and improved forced air drying	Every 5 days	37	47.715	0.653	7.629	0.752
Mid deep pit	>1Week	40	47.715	0.653	7.629	0.735
Deep pit	>1Week	75	47.715	0.653	7.629	0.539
External drying system with perforated manure belts	Daily	82	48.000	0.673	9.100	0.500
Defined by the user[1]	Defined by the user	Defined by the user	47.715	0.653	7.629	0.820

Source: INTIA's analysis database 9 samples for manure belt system without drying system and one sample for external drying system with perforated manure belts. Dry matter values established considering INTIA's database and EC (2003), bulk density calculated according to DM values by linear interpolation.
[1] Values per default considered for systems defined by the user.

The model does not consider phosphorus, potassium, copper and zinc losses, therefore the manure content of these nutrients is equal to the quantity excreted by the animals.

Organic matter content in the removed manure is calculated considering the quantity of carbon volatilized in the pit and contemplating that the methane emission from manure management (E_{CH4Man}, kg CH_4) represents the 65.1% of the total degradated carbon (C_{deg}, kg).

$$C_{deg} = \frac{E_{CH4Man}}{0.651} * \frac{12}{16}$$ **(Equation E7)**

The model considers a factor of 1.724 to calculate the excreted carbon (C_{ex}, kg) from the organic matter excreted (VS_{ex}, kg) (Equation E8).

$$C_{ex} = \frac{VS_{ex}}{1.724}$$ **(Equation E8)**

The degradation rate of the organic matter will be the same as the carbon, so the final organic matter content (VS_{Man}, kg) can be calculated according to Equation E9.

$$VS_{Man} = VS_{ex} - \left(VS_{ex} * \frac{C_{deg}}{C_{ex}}\right)$$ **(Equation E9)**

Equations E10 and E11 show the calculation of the final dry matter (DM_{Man}, kg) and carbon content (C_{Man}, kg), respectively.

$$DM_{Man} = \frac{VS_{Man}}{Ratio_{OM/DM}}$$ **(Equation E10)**

$$C_{Man} = C_{ex} - C_{deg}$$ **(Equation E11)**

Where, the $Ratio_{OM/DM}$ is the relation between organic and dry matter content in laying manure (Table E2).

The maximum methane producing capacity of the removed manure (Bo_{Man}, m^3 CH_4/kg VS) is calculated from the initial maximum methane producing capacity (Bo_{ini}, m^3 CH_4/kg VS) according to the following equation.

$$Bo_{Man} = \frac{(VS_{ex} * Bo_{ini}) - \left(CH_{4Man} * \frac{22.4}{16}\right)}{VS_{Man}}$$ **(Equation E12)**

The manure produced (Q, t) is calculated considering the quantity of dry matter removed from the buildings (DM_{Man}, kg) (Equation E10) and the dry matter content of the manure according to the system selected (DM_{Sys}, %) (Table E2).

$$Q = \frac{DM_{Man} * 100}{DM_{Sys} * 1000}$$ **(Equation E13)**

Broilers

The nitrogen content in the manure from buildings is calculated in the model as the nitrogen excreted plus the nitrogen from the bed, minus the nitrogen volatilized: ammonia and nitrous oxide.

The manure bulk density, the ammoniacal nitrogen content respect to total nitrogen and the pH are calculated considering manure analysis (Table E3) and the type/s and quantities of the bedding material/s used.

Table E3. Data from manure analysis. Regionalizable default values. INTIA database

Bedding material used	%TAN respect to TN	Organic matter/Dry matter (kg/kg)	pH	Bulk density (t/m3)	Number of analysis
Rice husks	17	0.78	7	0.6	16
Sawdust	19	0.79	7	0.6	6
Straw	17	0.78	7	0.6	---*

*Rice husks values used

The model does not consider phosphorus, potassium, copper and zinc losses in the manure, therefore the content of these nutrients is equal to the quantity excreted by the animals plus the quantity provided by the bed material.

Organic matter content in the removed manure is calculated considering the quantity of carbon volatilized and contemplating that the methane emission from manure management (E_{CH4Man}, kg CH_4) represents the 54.1% of the total degradated carbon (C_{deg}, kg).

$$C_{deg} = \frac{E_{CH4Man}}{0.541} * \frac{12}{16}$$ **(Equation E14)**

The model considers a factor of 1.724 to calculate the excreted carbon (C_{ex}, kg) from the organic matter excreted (VS_{ex}, kg).

$$C_{ex} = \frac{VS_{ex}}{1.724}$$ **(Equation E15)**

The degradation rate of the organic matter will be the same as the carbon, so the final organic matter content (VS_{Man}, kg) can be calculated according to Equation E16, where VS_{bed} will be the organic matter content of the bedding material (kg).

$$VS_{Man} = VS_{ex} - \left(VS_{ex} * \frac{C_{deg}}{C_{ex}}\right) + VS_{bed}$$ **(Equation E16)**

Equations E17 and E18 show the calculation of the final dry matter (DM_{Man}, kg) and carbon content (C_{Man}, kg), respectively.

$$DM_{Man} = \frac{VS_{Man}}{Ratio_{OM/DM}}$$ **(Equation E17)**

$$C_{Man} = C_{ex} - C_{deg} + C_{bed}$$ **(Equation E18)**

Where, the $Ratio_{OM/DM}$ is the relation between organic and dry matter content for broiler manure (Table E3) and C_{bed} is the carbon content of the bedding material (kg).

The maximum methane producing capacity of the removed manure (Bo_{Man}, m^3 CH_4/kg VS) is calculated calculated from the initial maximum methane producing capacity (Bo_{ini}, m^3 CH_4/kg VS) according to the following equation.

$$Bo_{Man} = \frac{(VS_{ex} * Bo_{ini}) - \left(CH_{4Man} * \frac{22.4}{16}\right)}{VS_{Man}}$$ **(Equation E19)**

Dairy cattle

N content in slurry produced at dairy facilities is calculated as the N excreted minus the N volatilized: NH_3 and N_2O. The ammonium-N content of liquid slurry is assumed to be 46% of total N content. N content in solid manure is calculated as the N excreted plus N from the bedding materials minus N volatilized as NH_3 and N_2O. The ammonium-N content of solid manure is assumed to be 11% of total N content. The model does not consider P. K. Cu and Zn losses from slurry and solid manure, therefore the slurry content of these nutrients is equal to those excreted by dairy herd.

Organic matter content in the manure considers animal excretion for slurry (VS_{ex}) and animal excretion (VS_{ex}) plus organic matter content of bedding in solid manure (VS_{bed}). Afterwards, the model considers the quantity of C volatilized in the pit, assuming that CH_4 emission from manure management (E_{CH4Man}) represents 60.9 and 63.7% of total degraded C in slurry and solid manure, respectively. The model considers a factor of 1.724 to calculate the excreted C from the organic matter excreted.

$$C_{deg} = \frac{E_{CH4Man}}{0.609 \text{ or } 0.637} * \frac{12}{16} \qquad \textbf{(Equation E20)}$$

$$C_{ex} = \frac{VS_{ex}}{1.724} \qquad \textbf{(Equation E21)}$$

The degradation rate of the organic matter will be the same as the carbon. So the final organic matter content (VS_{Man}. kg) is calculated as:

$$VS_{Man} = VS_{ex} - \left(VS_{ex} * \frac{C_{deg}}{C_{ex}}\right) + VS_{bed} \qquad \textbf{(Equation E22)}$$

To calculate final dry matter (DM), the model considers that VS_{Man} is 84% of total DM for liquid slurry. In relation to solid manure, is estimated to be the 81% of total DM.

Final carbon content (C_{Man}, kg) is calculated using Equation E23, where C_{bed} is the carbon content of the bedding material (kg).

$$C_{Man} = C_{ex} - C_{deg} + C_{bed} \qquad \textbf{(Equation E23)}$$

The maximum CH_4 producing capacity (Bo_{Man}, m^3 CH_4/kg VS) of the removed manure (slurry and/or solid manure) is calculated according to Equation E24.

$$Bo_{Man} = \frac{(VS_{manure} * Bo_{ini}) - \left(CH_{4Man} * \frac{22.4}{16}\right)}{VS_{Man}} \qquad \textbf{(Equation E24)}$$

Finally, the model uses default pH values for slurry and solid manure produced at dairy facilities considering INTIA and NEIKER analysis database as source, being 7 for liquid slurry and 8 for solid manure.

APPENDIX F: Grazing

Table F1. Emission factor for grazing animals. Regionalizable default values.

	Volatilization rate (% of N excreted)	Source
NH_3	10	Amann et al. (2004)[1]
N_2O Direct	1.0	
N_2O Indirect*	1.0	IPCC (2006)[2]
N_2O Leaching	2.5	

[1]Amann M., Cofala J., Klimont Z. and Schöpp, W.(2004). RAINS Review 2004 – Modelling of Emission Control Potentials and Cost. IIASA, Austria

[2]IPCC (2006). IPCC guidelines for national greenhouse gas inventories, prepared by the national greenhouse gas inventories programme. Eds H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, K. Tanabe, Hayama, Japan: IGES, 87 pp.

*% of NH_3-N

APPENDIX G: Liquid Storage

References

Bicudo J., Clanton C., Schmidt D., Powrs W., Jacobson L. and Tengman C. (2004). Geotextile covers to reduce odor and gas emissions from swine manure storage. Applied Engineering in Agriculture, 20 (1), 65-75.

IPCC (2006). IPCC guidelines for national greenhouse gas inventories, prepared by the national greenhouse gas inventories programme. Eds H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, K. Tanabe, Hayama, Japan: IGES, 87 pp.

MAGRAMA (2014). Evaluación de Técnicas de Reducción de Emisiones en Ganadería. Sectores de Porcino y Avicultura de Carne y Puesta. Ministerio de Agricultura, Alimentación y Medio Ambiente, Madrid, 114 pp.

Ndegwaa P., Hristovb A., Arogoc J. and Sheffieldd R. (2008). A review of ammonia emission mitigation techniques for concentrated animal feeding operations. Biosystems Engineering, 100, 453-469.

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Equation G1. Variation Factor associated to slurry temperature (VF_{Temp})

$$VF_{Temperature} = \frac{e^{(0.08 * TempEfluent)}}{2.612}$$

$Temp_{Effluent}$: Slurry temperature (ºC) estimated from ambient temperature (Equation G2).

Equation G2. Slurry temperature (ºC) estimated from ambient temperature ($Temp_{Effluent}$)

$$Temp_{Effluent} = 0.9614\ T + 1.6889$$

T: ambient temperature (ºC)

Equation G3. Variation Factor associated to slurry dilution (VF_{Ndil})

$$VF_{Ndil} = TAN/\ 1.03$$

TAN: Total amoniacal nitrogen (mg/kg)

Equation G4. Variation Factor associated to natural crust ($VF_{NH3Crust}$)

$$VF_{NH3Crust} = 1 - \frac{RF_{NH3Crust}}{100}$$

$RF_{NH3Crust}$: Emission reduction of NH_3 associated to natural crust (Table G1)

Table G1. Effect of natural crust in slurry storages emissions. IPPC (2006) and MAGRAMA (2014).

Natural Crust	$RF_{NH3Crust}$ (% emission reduction)	$EF_{N2OCrust}$ (kg N_2O-N/kg Total N)
With	28	0.005
Reference: Without	*0*	*0*

Equation G5. Variation Factor associated to covers (VF_{XCover})

$$VF_{XCover} = 1 - \frac{RF_{XCover}}{100}$$

RF_{XCover}: Emission reduction of gas X associated to cover (Table G2)

Table G2. Effect of covers in slurry storage emissions. Bicudo et al. (2004).

Type of Cover	$RF_{NH3Cover}$	$RF_{CH4Cover}$	$EF_{N2OCover}$
	(% emission reduction)		(kg N_2O-N/kg Total N)
Rigid Roof	80	--	--
Flexible Cover (tent-floatting-swollen)	85	--	--
Floatting Maters (straw-polystyrene)	65	-20*	0.001
Reference: No Cover	*0*	*0*	*0*

* Increase emission 20%
-- No data available, no effect considered in calculations for the moment

Equation G6. Variation Factor associated to additives (VF_{XAdd})

$$VF_{XAdd} = 1 - \frac{RF_{XAdd}}{100}$$

RF_{XAdd}: Emission reduction of gas X associated to additives (Table G3)

Table G3. Effect of additives in slurry storage emissions. Ndegwaa et al. (2008).

Type of Additive	RF_{NH3Add}	RF_{CH4Add}	EF_{N2OAdd}
		(% emission reduction)	
Zeolite	71	--	--
Saponite	23	--	--
Alliance	23	--	--
Reference: No Additive	*0*	*0*	*0*

-- No data available, no effect considered in calculations for the moment

Equation G7. Water losses through evaporation ($H_2O_{\_evaporation}$, m^3)

$$H_2O_{\_evaporation} = \frac{0.484 * (1 + 0.6\,Wind) * (1 - \frac{HR}{100}) * 611 * e^{\frac{17.27\,Temp}{237.3 + Temp}} * 31 * Stg.\,surface}{1000000} * R_{cover}$$

Wind: m/s

HR: Air relative humidity

Temp: Air temperature (ºC)

Stg.surface: Storage surface (m^2)

R_{cover}: Correction coefficient when a cover is selected. Default value of 0.5.

Equation G8. Carbon dioxide emissions (CO_2, kg)

$$CO_2 = \frac{CH_4 * (1 - R) * 3.667}{R * 1.333}$$

CH_4: Methane emissions (kg)

R: Ratio CH_4 vs CO_2. Default value, 0.6.

Equation G9. Final organic matter (VS, kg)

$VS=VS_{ini}*C_{ini}/(C_{ini}-C_{CH4}-C_{CO2})$

VS_{ini}: Initial organic matter content (kg)

C_{ini}: Initial carbon content (kg)

C_{CH4}: Carbon volatilized in CH_4 form (kg)

C_{CO2}: Carbon volatilized in CO_2 form (kg)

Equation G10. Final N (N, kg)

$N=N_{ini}-N_{NH3}-N_{N2O}$

N_{ini}: Initial N content (kg)

N_{NH3}: N volatilized in NH_3 form (kg)

N_{N2O}: N volatilized in N_2O form (kg)

Equation G11. Final ammoniacal N content (N_{NH4}, kg)

$N_{NH4}=N_{NH4ini}-(N_{ini}-N)+(N_{ini}-N_{NH4ini})*(1-C_{ini}/(C_{ini}-C_{CH4}-C_{CO2}))$

N_{NH4ini}: Initial ammoniacal N content (kg)

N_{ini}: Initial N content (kg)

N: Final N content (kg) (Equation G10)

C_{ini}: Initial carbon content (kg)

C_{CH4}: Carbon volatilized in CH_4 form (kg)

C_{CO2}: Carbon volatilized in CO_2 form (kg)

Equation G12. Final dry matter content (DM, kg)

$DM=DM_{ini}-(VS_{ini}-VS)-(N_{ini}-N)$

DM_{ini}: Initial dry matter content (kg)

VS_{ini}: Initial organic matter content (kg)

VS: Final organic matter content (kg) (Equation G9)

N_{ini}: Initial N content (kg)

N: Final N content (kg) (Equation G10)

Equation G13. Total matter calculation (TM, tons)

$TM=TM_{ini}-H_2O_{\_evaporation}+Rain-(VS_{ini}-VS)-(N_{ini}-N)$

TM_{ini}: Initial total matter (tons)

$H_2O_{\_evaporation}$: Water losses through evaporation (m^3)

Rain: Water entrance due to precipitation (m^3)

VS_{ini}: Initial organic matter content (tons)

VS: Final organic matter content (kg) (Equation G9)

N_{ini}: Initial N content (tons)

N: Final N content (tons) (Equation G10)

APPENDIX H: Solid Storage

References

Moscatelli G., Fabbri C., Valli L. and Mantovi P. (2008). External drying tunnel for droppings of laying hens: ammonia, GHG and odour emissions. 13th RAMIRAN International Conference, Albena, Bulgaria.

Rigolot C., Espagnol S., Pomar P., Hassouna M., Béline F., Paillat J. M. and Dourmad J.Y. (2010). Modelling of manure production by pigs and NH_3, N_2O and CH_4 emissions. Part II: effect of animal housing, manure storage and treatment practices. Animal 4 (8), 1413-1424.

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Table H1. Emission Factor coefficients (EF), Variation Factors (VF) and Total N and water losses. Adapted from Rigolot et al. (2010) considering Moscatelli et al. (2008) and experts opinions.

		Emissions				Losses	
		NH_3-N	N_2O-N	CO_2-C	CH_4-C	Total N	H_2O
EF		0.45	0.03	0.45	0.015	0.50	0.75
VF_{Type}							
C:N	DM (%)						
<15	<25	0.4	1	0.6	1	1	0.8
	25 to 35	1	0.3	0.9	0.03	1	0.8
	36 to 74	0.7	0.1	0.8	0.02	0.8	0.5
	>75	0.09	0.01	0.1	0	0.1	0.04
15 to 25	<25	0.4	0.5	1	0.24	1	0.7
	25 to 35	0.8	0.3	1	0.04	0.9	1
	36 to 74	0.5	0.1	0.8	0.02	0.5	0.5
	>75	0.09	0.01	0.1	0	0.1	0.04
>25	<25	0.3	0.5	1	0.1	0.7	0.6
	25 to 35	0.5	0.3	1	0.02	0.6	0.9
	36 to 74	0.2	0.1	0.8	0.02	0.2	0.6
	>75	0.09	0.01	0.1	0	0.1	0.04
VF_{Dur}		1.1	1.4	1.4	1.4	1.4	1.4
VF_{Temp}		*	1	**	**	**	**

*VF_{Temp}= (0.2-15*0.14/18)/375*temperature^2+0.14/18*temperature+(33.1-210*0.14/18)/48
**VF_{Temp}=(0.2-15*0.34/18)/375*temperature^2+0.34/18*temperature+(33.1-210*0.34/18)/48

APPENDIX I: Landspreading

References

Bittman S., Dedina M., Amon B., Menzi H., Webb J., Groenestein K., Misselbrook T., Hutchings N., Dohler H., van der Hoek K., Gyldenkærne S., Valli L., Pallière C., Howard C., Oenema O. and Sutton M. (2015). Framework Code for Good Agricultural Practice for Reducing Ammonia Emissions. European Commission on behalf of the Task Force on Reactive Nitrogen of the UNECE Convention on Long-range Transboundary Air Pollution, 32 pp.

EC (2013). Best Available Techniques (BAT) Reference Document for the Intensive Rearing of Poultry and Pigs. Integrated Pollution Prevention and Control (IPPC), Draft 2-August 2013, 839 pp.

MAGRAMA (2014). Evaluación de Técnicas de Reducción de Emisiones en Ganadería. Sectores de Porcino y Avicultura de Carne y Puesta. Ministerio de Agricultura, Alimentación y Medio Ambiente, Madrid, 114 pp.

IPCC (2006). IPCC guidelines for national greenhouse gas inventories, prepared by the national greenhouse gas inventories programme. Eds H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, K. Tanabe, Hayama, Japan: IGES, 87 pp.

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Table I1. Default DM contents of liquid manures. Regionalizable default values. INTIA database.

Type of product	DM (kg/t)
Dairy Cattle slurry	110
Beef Cattle slurry	50
Pig slurry	43

Table I2. Values of abatement of ammonia emissions for application methods and incorporation in soil. Regionalizable default values. EC (2013), MAGRAMA (2014), Bittman et al. (2015).

Methods		Incorporation	
% Emission reduction comparing to the reference system*			
Trailing Hose	30%	Immediate (Plough)	90%
Trailing Shoe	60%	Immediate (Shallow)	70%
Shallow injection	70%	< 4 hrs	55%
Deep injection	90%	4-24 hrs	30%
		> 24 hrs	0%

*Reference system: broadcast or splashplate application with no incorporation.

Nitrous oxide emissions

The Net N application rate (NNET, Equation I1) is used for calculating N_2O-N emissions following manure application. Direct emissions (N2ODIR, Equation I2) are calculated as 1% of the N applied (net of NH_3 emissions at landspreading) (IPCC, 2006). Indirect emissions (N2OINDIR, Equation I3) are calculated as the N_2O-N emission associated with the redeposition of NH_3-N emitted following landspreading.

The assumption in the model is that 100% of the NH_3-N emitted is redeposited, and that 1% of this N is emitted as N_2O-N. In effect, the assumption of 100% redeposition means that total N_2O-N emissions are always equal to 1% of total N applied in manure.

Since the reduced ammonia emissions will offset fertilizer requirements, the emissions of N_2O-N associated with complementary mineral N fertilizer application (N2OFERT, Equation I6), and the N_2O-N associated with the potential leaching of that fertilizer N (N2OLEACH, Equation I7), will be reduced by a BAT that reduces NH_3 loss. IPCC default values of % N leached, and N_2O-N emissions from manure N, mineral fertilizer N and N leached (assumed to be 30%, 1%, 1% and 2.5%, respectively) can be used to include the effect of change in fertilizer N application on the net impact on N_2O-N emissions arising from the manure application (N2ONET, Equation I9).

Equation I1. Net N application Rate (NNET, kg/ha)

$NNET=RN-(E_{NH3}*14*17^{-1})$

E_{NH3}: Total ammoniacal N loss (kg NH_3 ha^{-1}) (Equation 22)

RN: kg total N per ha.

Equation I2. N_2O-N Emission - Direct emissions from N application (N2ODIR, kg/ha)

N2ODIR=NNET*N2ODIREF

NNET: Net N application Rate (kg/ha) (Equation I1)

N2ODIREF: 0.01 (Regionalizable default value)

Equation I3. N_2O-N Emission - Indirect emissions from redeposition of volatilised NH_3-N (N2OINDIR, kg/ha)

$N2OINDIR=(E_{NH3}*14*17^{-1})*N2OINDIREF$

E_{NH3}: Total ammoniacal N loss (kg NH_3 ha^{-1})

N2OINDIREF: 0.01 (Regionalizable default value)

Equation I4. Total N_2O-N emissions from manure application (N2OMAN, kg/ha)

N2OMAN=N2ODIR+N2OINDIR

N2ODIR: N_2O-N Emission - Direct emissions from N application (kg/ha) (Equation I2)

N2OINDIR: N2O-N Emission - Indirect emissions from redeposition of volatilised NH_3-N (kg/ha) (Equation, I3)

Equation I5. Fertilizer N replaced (FERTN, kg/ha)

$FERTN=TAN_{ABATE}*TANSUBS$

TAN_{ABATE}: Total ammoniacal N abated (Equation 23)

TANSUBS: 1 (fertilizer replacement value of ammonia not volatilized, i.e. 1 infers that ammonia-N not volatilised will replace 1 kg of mineral N fertilizer) (Regionalizable default value)

Equation I6. N_2O-N Emissions abated due to mineral N fertilizer replacement (kg/ha, N2OFERT)

N2OFERT=FERTN*N2OFERTEF

FERTN: Fertilizer N replaced (kg/ha) (Equation I5)

N2OFERTEF: 0.01 (Regionalizable default value)

Equation I7. N_2O-N Emissions abated due to mineral N fertilizer replaced not leached (kg/ha, N2OLEACH)

N2OLEACH=FERTN*FERTNLEACH*N2OLEACHEF

FERTN: Fertilizer N replaced (kg/ha) (Equation I5)

FERTNLEACH: 0.3 (proportion of fertilizer N applied that is leached through the soil) (Default value for Spain)

N2OLEACHEF: 0.025 (Regionalizable default value)

Equation I8. Total N_2O-N abated due to fertilizer replacement (N2OABATE, kg/ha)

N2OABATE=N2OFERT+N2OLEACH

N2OFERT: N_2O-N Emissions abated due to mineral N fertilizer replacement (kg/ha) (Equation I6)

N2OLEACH: N_2O-N Emissions abated due to mineral N fertilizer replaced not leached (kg/ha) (Equation I7)

Equation I9. Net N_2O-N emission (N2ONET, kg/ha)

N2ONET=N2OMAN-N2OABATE

N2OMAN: Total N_2O-N emissions from manure application (kg/ha) (Equation I4)

N2OABATE: Total N_2O-N abated due to fertilizer replacement (kg/ha) (Equation I8)

APPENDIX J: Energy subroutine

References

MARM (2011). Inventario de emisiones de gases de efecto invernadero de España e información adicional 1990-2009. Secretaría de Estado de Cambio Climático, Dirección General de Calidad y Evaluación Ambiental, D.G. Oficina Española de Cambio Climático, Madrid, 706 pp.

IPCC (2006). IPCC guidelines for national greenhouse gas inventories, prepared by the national greenhouse gas inventories programme. Eds H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, K. Tanabe, Hayama, Japan: IGES, 87 pp.

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Farm energy balance (E_{Farm}, KWh/year) is calculated subtracting the energy that it is produced in the farm (biogas) from the energy produced by purchased combustibles and electricity (Equation J1).

$$E_{Farm} = \langle E_{Elec\ Count} - E_{Elec\ Biogas} \rangle + \langle E_{Comb\ Purch} - E_{Comb\ Biogas} \rangle \pm E_{BATs} \qquad \textbf{(Equation J1)}$$

Where: $E_{ElecCount}$ is the energy provided by the electricity consumed in the farm (KWh/year); $E_{ElecBiogas}$ is the energy provided by the electricity produced by biogas co-generation engines (KWh/year); $E_{CombPurch}$ is the energy provided by the combustibles purchased (KWh/year); $E_{CombBiogas}$ is the energy provided by the heat of combusted biogas (KWh/year) and E_{BATs} will be the effect of housing BAT on energy when it is included in the balance by the user (KWh/year), with negative value when BAT decrease energy consumption.

Equation J2 shows the farm CO_2 equivalent balance (Eq_{Farm}, t/year).

$$Eq_{Farm} = \langle Eq_{Elec\ Count} - Eq_{Elec\ Biogas} \rangle + Eq_{Comb\ Purch} + Eq_{Comb\ Biogas} + Eq_{BATs} \qquad \textbf{(Equation J2)}$$

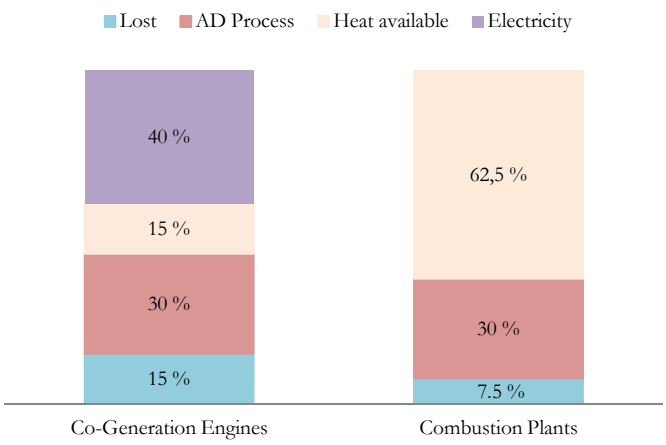
Where: $Eq_{ElecCount}$ is the CO_2 equivalent due to the electricity consumed in the farm (t/year); $Eq_{ElecBiogas}$ is the CO_2 equivalent due to the electricity produced by biogas co-generation engines (t/year); $Eq_{CombPurch}$ is the CO_2 equivalent due to the combustion of combustibles purchased (t/year); $Eq_{CombBiogas}$ is the CO_2 equivalent due to N_2O and CH_4 emissions produced during anaerobic digestion (biogas leakage and non-combusted biogas) (t/year) and Eq_{BATs} is the effect of housing BAT energy consumption on CO_2 equivalents (t/year).

Table J1. Combustibles in the energy subroutine. Regionalizable default values.

Type	Heatingvalue (GJ/t)	Emission factor CO_2 (tCO_2/TJ)	Emission factor CH_4 (kg CH_4/TJ)[3]	Emission factor N_2O (kg N_2O/TJ)[3]	Density (kg/l)
Propane	46.2[1]	64.2[1]	1	0.1	
Natural Gas	48.59[1]	56.3[1]	1	0.1	
Gas oil	42.4[2]	73.7[1]	3	0.6	0.832
Pellet	19.1				
Olive Stone	17.4				
Almond Shell	36.8				
Walnuts Shell	32.0				
Olive marc	17.9				
Grape marc	19.1	0[4]	30	4	
Drywood	19.0				
Olive wood	19.6				
Pinecones	18.8				
Vine shoots	18.8				
Charcoal	16.7				
Otherbiomass	19.0				

Sources:[1] MARM (2011), Anexe 8.
[2]Pellet: Several analyses
[3]IPCC 2006, Table 2.5. In the case of biomass products: solid biofuels/wood/wood waste.
[4]2009/28/CE Directive considers null emissions during combustion for biomass products.

Fig. J1. Biogas heat use considered by the model. IRSTEA Database.



APPENDIX K: FIOS

Table K1. FIO / pathogen risk ratings adopted in the model (0 = negligible effect; 1 = some effect; 2 = high effect)

Model part	Sub-system	Technique/Management	Risk rating	Notes
Housing	All	All BAT except those listed below	0	
	Poultry	Vertical tiered cages with manure belts and forced air drying	1	Drying leading to FIO reduction (manure DM≥75%)
	Cattle	Aeration of slurry under slats	1	Oxygen pressure on some pathogenic organisms
		Access to Freshwater for Grazing Animals:		Livestock that have free access to watercourses may impact the water quality as livestock manure contains a wide range of bacterial, viral, and parasitic pathogens. In extensive grazing systems, livestock frequently have direct access to streams and rivers, and defecation into water courses is not uncommon. This has negative implications for water quality because pathogens flow with the current, travel long distances, and come in direct contact with humans using water for recreational purposes.
		* No access - drinkers, troughs etc	2	
		* Restricted access - fencing	1	
		* Unrestricted access - poaching	0	
		Minimum Frequency of Manure Removal without the Addition of Fresh Manure:		Especially for dairy cattle, long-term storage of slurry under slats can have an effect of FIOs and pathogenic organisms: long term storage in itself has some reducing effect even though the constant addition of fresh slurry reduces this. In farms where the slurry stays in storage after the animals leave the buildings during the grazing season; this long term storage without further addition of fresh slurry could have a significant effect or FIO numbers in the stored slurry. Frequent of removal of manure from surfaces can in itself reduce the risk of re-infection of the herd, and thus have a positive effect. However, this is dependent on other factors such as herd infection rate and other infection control mechanisms, and thus not considered in the model.
		<1 month	0	
		1-3 months	1	
		>3 months	2	
	All	Nutrition	0	Little information on FIO. There is some information that forage leads to longer E Coli shedding than grain feed but other feed additives / phase feeding information is inconclusive.

Table K1. FIO / pathogen risk ratings adopted in the model (0 = negligible effect; 1 = some effect; 2 = high effect)

Model part	Sub-system	Technique/Management	Risk rating	Notes
Storage	Liquids and Solids	Static storage (no additions):		FIO/ Pathogen decline most pronounced if storage without further addition of slurry (sequential filling of lagoons/tanks). Most pathogens levels will be reduced within 30 days, whereas addition of new loads will cause re-infection.
		<3 months	0	
		3-6 months	1	
		>6 months	2	
Treatment	Liquids	Pre-treatment by solids separation	1	Coarse solids separation is likely to lead to a maximum FIO reduction of one order of magnitude
		Anaerobic digestion (mesophilic)	1	Pathogen inactivation from mesophilic treatment is less successful than thermophilic but time is also an important factor, as is sequential operation.
		Anaerobic digestion (thermophilic)	2	Very successful in inactivating E. coli - T90 at 55oC is 0.02 days - although pH and free NH_3 during process may contribute
	Solids	Composting (open windrow) mesophilic	1	Similar effect to mesophilic treatment of liquids but larger risk of re-infection during turning. Pile separation or 2-stage process is recommended.
Land-spreading	Liquids	Broadcast	0	Measures to reduce FIO pollution should be aimed at retaining any FIOs in the soil, i.e. reducing the risk of run-off after application. The model therefore allocates the highest effects to measure that reliably and significantly reduces the presence of FIOs on surfaces (e.g. injection or immediate incorporation).
		Deep injection	2	
		Shallow injection	2	
		Trailing Hose	1	
		Trailing Shoe	1	
	Solids and Liquids	Incorporation:		
		< 4 hrs	2	
		> 24 hrs	1	
		4-24 hrs	2	
		No Incorporation	0	

APPENDIX L: Scenario comparison

Table L1. Parameters selected for farm comparison

Parameter	Farm Stage*				
	H	S	T	L	WF
Live weight produced (kg/year)	√				
Milk Yield (l/herd/year)	√				
Dozen of eggs per laying hen and year	√				
Protein and Phosphorus consumed (g per PU**)	√				
Total water consumption (m3/year)	√				
Total manure production (t/year)	√				
Ammonia emission (kg/year)	√	√	√	√	√
Nitrous oxide emission (kg/year)	√	√	√	√	√
Methane emission (kg/year)	√	√	√		√
Carbon dioxide emission (kg/year)		√	√		
Fertilizer units (N, P, K) after application (kg/year)				√	
Fertilizer units (N) lost during application (kg/year)				√	
Fertilizer units (N) after application respect total N excreted (%)				√	
Ammonia emission (g per PU**)					√
Nitrous oxide emission (g per PU**)					√
Carbon dioxide emission (g per PU**)					√

√: Parameter selected
*H: Housing; S: Storage; T: Treatment; L: Landspreading; WF: Whole Farm
**PU: Production Unit (kg of live weight produced or per egg produced or per milk liter or per place)

A whole farm model for Ammonia and Greenhouse Gas Emissions from Livestock Operations

Part 2: Application in swine farms

To be submitted as:

Aguilar M., Arriaga H., Dupard P., Lalor S., Fragoso R., Pahl O., Abaigar A., Cordovín L., Boyle M., Lanigan G., Loyon L., Duarte E., Merino P. A whole farm model for Ammonia and Greenhouse Gas Emissions from Livestock Operations. Part 2: BATFARM model application in swine farms.

A whole farm model for Ammonia and Greenhouse Gas Emissions from Livestock Operations. Part 2: Application in swine farms.

Abstract

This article presents the application of the whole farm model described in the previous Chapter 3 to assess the emissions of nitrous oxide, methane and ammonia as a consequence of different strategies and techniques implemented on livestock intensive farms. Model testing has been carried out in typical swine farms under Navarre region conditions (North of Spain) as case study. Model's response to systematic variation of different climates, diets, manure managements and landspreading strategies has been studied. A basic quality check of the model has been performed conducting N and C mass balance and comparing its results against literature data. Simulation scenarios were used to illustrate potential applications at farm scale and to further demonstrate its coherent behaviour over different sets of parameter values. Finally, software usability was evaluated by regional stakeholders. Results seem to indicate that the model considers relevant interactions between farm components and therefore it is useful for integrated scenario development and evaluation. The model captures the effect of factors having an important impact on gas emission. However, further validation would be required to evaluate its reliability and enable broaden uses, not only for scenario comparisons but also for the calculation of absolute emissions. The application scenarios illustrated well model flexibility and its potential for management purposes, but results need to be completed with current economic data. Despite its limitations, this model advances in our understanding of the mitigation strategies effect in whole farm systems and could be used as a complementary tool for BAT implementation.

Keywords

Gaseous emission, Model testing, Manure Management, Nutrition, Sensitivity Test, Simulation.

4.1. Introduction

Current trends in livestock production towards more specialized and intensive production had remarkable impact on environment and led to a complex set of new and changing policies in order to minimize those damages. In particular, farms with more than 40000 poultry, 2000 fattening pig or 750 sow places which are under the Industrial Emissions Directive IED 2010/75/EU (former IPPC) must have a permit to operate based on the implementation of Best Available Techniques (BAT) in the whole farm production process.

BAT are defined as the most effective techniques to accomplish high general level of environmental protection under economically and technically viable conditions in a particular farm situation (IED 2010/75/EU). Strategies contemplated involve whole farm systems including nutritional techniques, housing design to reduce emission, techniques to reduce water and energy consumptions, to improve manure handling during storage, reduced emission spreading methods and treatment techniques (EC, 2003; EC, 2013).

The abundance of possible techniques, the diversity of the farming systems and of socioeconomic conditions, complicates the decision process. Consideration of the global effect of the proposed techniques in the whole farm system will be therefore essential for selection of a valid strategy. Computer-based models and decision support tools can be very useful for environmental authorities, farmers and technicians in this task.

Different models have been developed to estimate gaseous emissions from agriculture (Schils, Olesen, del Prado, Soussana, 2007; Menzi, Bonjour, Zaucker, Leuenberger, Reidy, 2009; Hutchings, Sommer, Andersen, and Asman, 2001; Webb et al., 2006). However, there is a lack of support and knowledge transfer models designed with methodological rigour and as a user-friendly tool which can be optimised for specific farms.

A whole farm model was recently developed by Aguilar et al. (Thesis Chapter 3) to assess the emissions of N_2O, CH_4 and NH_3 resulting from different strategies and techniques implemented on intensive cattle, pig and poultry farms. Results provided allow to identify the key stages giving rise to farm emission, which would most benefit from implementing environmental techniques. The model also predicts farm emission variation

under different scenarios. Detailed descriptions of model methods can be found in paper Part 1 (Thesis Chapter 3). This Part 2 aims to present model application in swine farms studying how simulated gas emissions response to systematic variation of management parameters; conducting N and C mass balance; comparing model results against literature data; simulating scenarios to illustrate potential applications and finally showing model evaluation by main stakeholders.

4.2. Material and Methods

Funded by the BATFARM Interreg-Atlantic Area Project (2009-1/071), the model has been developed specifically by authors from Portugal, Spain, France, Scotland and Ireland but it could be adapted to other climatic conditions and farm practices. A mass balance is carried out throughout housing, storage, treatment and landspreading stages, on a cumulative monthly and annual basis in order to estimate manure evolution (mass, dry and organic matter, N, P, K, Cu and Zn contents), related emissions (N_2O, CH_4 and NH_3), consumptions and animal production. A combination of methodologies including emission factors, empirical equations and process-oriented mechanisms has been used for its construction. Different mitigation strategies, farmer practices and climatic conditions which have significant effects on gaseous emissions have been considered (Thesis Chapter 3).

The links among the different subroutines in the software structure and the types of reports produced are shown in Figure 1. Further software information and availability can be found in Table 1.

Fig. 1. General structure of the software. Numbers indicate the navigation sequence. Phase 1: Model Language and General Farm Information; Phase 2: Definition of farm situation; Phase 3: Comparison of two farm situations.

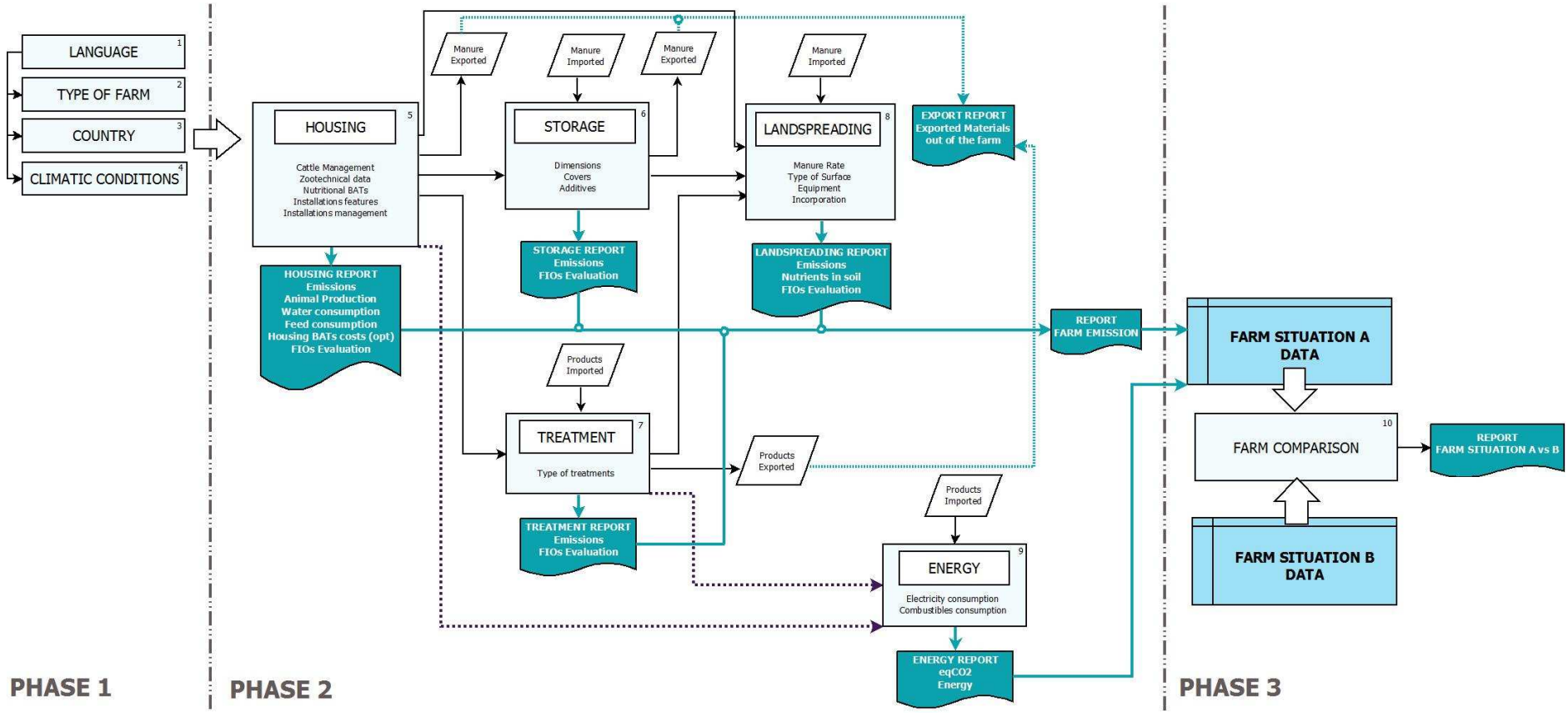


Table 1. BATFARM software availability.

Name	BATFARM Software.
First year available	2015
Operating Systems required	XP, Vista, Windows 7 or Windows 8. "Microsoft .NET Framework 3.5." is also necessary. The software will be downloaded compressed in rar format.
Availability	http://www.intiasa.es/batfarm-software.html
Costs	Free use tool
Program Language	Visual Basic
Programming software	Visual Studio Net 2008
Technical documentation	User manual, BAT documents, specific screen help documents, specific advanced users help.
Languages	English (EN), Spanish (SP), Portuguese (PT) and French (FR).
Software structure	8 Main subroutines. The software produces a report for each subroutine (exportable to Excel®) and three more reports containing: i) an export report shows the quantity and composition of manure sent out of the farm; ii) farm emission report displays whole farm emissions up to the moment of landspreading and iii) a comparison report collates the main parameters of two different situations simulated.

A full validation of model at the scale of a farming system is often difficult to carry out because the information required is often missing in protocol description from published studies (i.e. outdoors climatic conditions, housing characteristics or feed composition) (Meda, Robin, Aubert, Dourmad, Hassouna, 2012; Rigolot *et al.*, 2010). At this stage, a model test was performed in typical swine farms under Navarre region conditions. Navarre region is part of the Ebro Valley (Spain), an area characterized by a high concentration of livestock production, especially swine farms under IED.

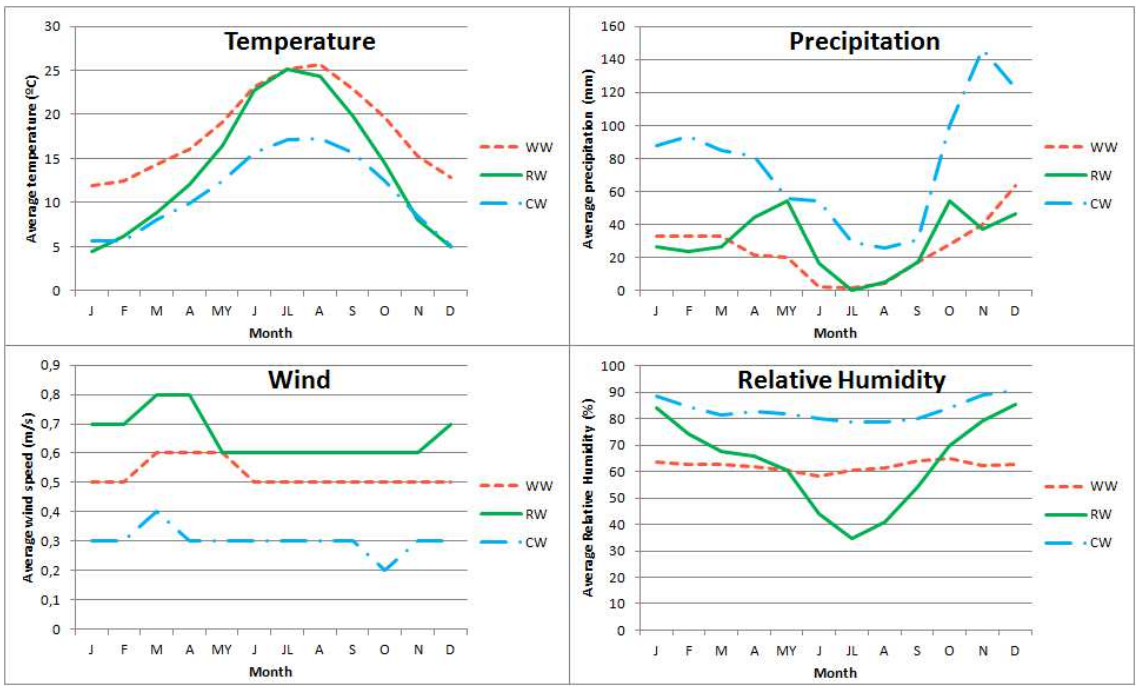
A set of model runs was carried out in which BATFARM model was run individually to evaluate the effect on gas emissions of a combination of different climates, diets, manure managements and landspreading equipments under common production conditions in Navarre region. Diet manipulation and manure management are frequently used as mitigation techniques in IED farms in Navarre region and have been shown to affect gaseous emissions (Webb, Broomfield, Jones, Donovan, 2014). Nutritional strategies aim at reducing the level of N and P excretion and the subsequent emission of NH_3 from manure, without compromising animal performance, health or welfare. They offer the advantage of being easy to implement and rapid to adapt (Philippe, Cabaraux, Nicks, 2011). Manure management strategies consider frequencies of slurry removal, storage design, equipments for NH_3 emission mitigation and good practices for landspreading (e.g. seasonal timing of

manure application to correspond to crop needs). Proper agricultural manure handling, optimizing its fertilizer value, is another basic strategy in order to minimize farm pollution at reasonable cost.

The farm simulated had 4160 animal places (from 19 to 110 kg), 3% of mortality, 2.6 kg feed consumed kg LW^{-1} and an increase of 680 g LW day^{-1}. Fully slatted concrete floor and nipple drinker in a cup were used in animal housing. The slurry was monthly removed from building.

Simulated climates represent either the predominant climate in which pigs are reared in Navarre (Reference Weather, RW) or two likely extreme climates in the region (Cold Weather, CW and Warm Weather, WW)[1] (Figure 2). Per default, the model considers that indoor temperatures at swine housings presents a low variation along the different months of the year, regardless of the type of climate selected, due to environment control systems implemented in these farms. To achieve this, we have considered that ventilation rates differed ±10% in CW and WW regions in relation to RW during summer season.

Fig. 2. Model testing on climatic parameters. Climates simulated: Reference (RW), Cold (CW) and Warm (WW).



[1] In BATFARM software CW can be found as Oceanic transition, RW as Continental Mediterranean warm summers and WW as Mediterranean arid and semiarid.

Table 2 shows the main characteristics of the diets selected in order to reduce N and P excretion.

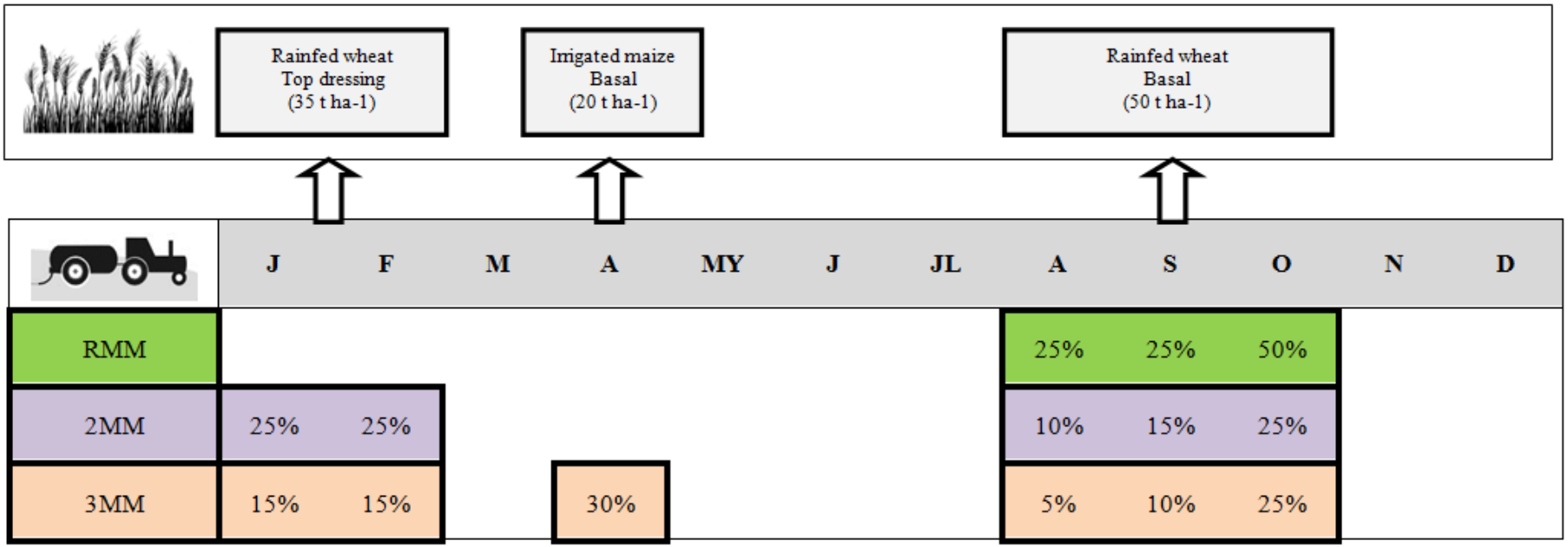
Table 2. Model testing on feeding parameters. Diets simulated: Reference (RD), Biphase (BD) and Biphase+Aminoacids+Phytases (BAPD)

Type of diet	Type of feed	Total feed consumption (%)	CrudeProtein (%)		Phosphorus (%)
RD	Growing	40	17	16.4 (Average)	0.55
	Finishing	60	16		0.55
BD	Growing	40	16	15.1 (Average)	0.5
	Finishing	60	14.5		0.45
BAPD	Growing	40	15	14.1 (Average)	0.4
	Finishing	60	13.5		0.4

In manure management strategies, the slurry storage duration was modified depending on landspreading practices. Slurry from buildings was stored in a single tank without natural crusting (6000 m^3, 1500 m^2). No covers or additives were simulated. In the reference situation for manure management (RMM) in Navarre, there was only one period of slurry application in arable land from August to October (50 t ha^{-1}, rainfed wheat basal fertilization). In the second situation (2MM), slurry was additionally removed in winter for rainfed wheat top dressing fertilization (35 t ha^{-1}). Therefore, there were 2 periods of slurry application: January-February and August-September-October. Finally, in the third simulated situation (3MM) there was also spring application for basal fertilization in irrigated maize (20 t h^{-1}). As a consequence, there were 3 periods of slurry application on land: January-February, April and August-September-October. Figure 3 shows these different strategies of manure management.

Three different equipments for landspreading were also simulated. The reference landspreading (RL) equipment selected was broadcast by splash plate, the THL situation uses a Trailing Hose (used by 100% of swine farms in Navarre under IED directive) and SIL corresponds to Shallow Injection.

Fig. 3. Model testing on manure management strategies. Percentage of total slurry removed each month for Reference (RMM), 2 periods (2MM) and 3 periods (3MM) strategies.



As a result a total of 21 simulations (3 climates with 3 different diets, 3 manure managements and 3 landspreading practices) were conducted (Table 3). In each simulation, the reference situation was selected in all production stages except for the strategy under study: when the different diets were simulated, RMM and RL were selected in storage and landspreading; when different manure management was studied, RD and RL were selected at housing and at landspreading, respectively; when different equipment for landspreading was simulated, RD was selected at housing and storage management was RMM.

Table 3. Model testing. Combinations of practices simulated.

Simulation	Climate	Diet	Manure Management	Landspreading Equipment
1	CW	RD	RMM	RL
2		BD		
3		BAPD		
4	RW	RD		
5		BD		
6		BAPD		
7	WW	RD		
8		BD		
9		BAPD		
10	CW	RD	2MM	
11			3MM	
12	RW		2MM	
13			3MM	
14	WW		2MM	
15			3MM	
16	CW		RMM	THL
17				SIL
18	RW			THL
19				SIL
20	WW			THL
21				SIL

Secondly, basic quality check of the model has been carried out conducting N and C mass balance at the scale of the whole system (from feeding to field application) and comparing model results against literature data. The validated output values from the model were NH_3, N_2O and CH_4 emissions, slurry volume produced and slurry nutrient content.

Thirdly, simulation scenarios were used to illustrate potential applications of this model as a management tool and to further demonstrate its coherent behaviour over different sets of parameter values.

A typical Spanish farrow to finish pig farm was simulated to basic quality check and as reference situation in simulation scenarios. Farrow to finish farms are the most

traditional type of swine production in Spain, accounting for the 57% of farms and 36% of animals (MARM, 2010). The farm was located in RW climate (Figure 2) and had 464 sows housed, 2000 weaner places (from 5 to 22 kg) and 3700 places for grower-finishing pigs (from 22 to 110 kg). Standard feeding was selected in all phases and nipple drinker in a cup in pre-fattening and fattening houses. The floor was concrete fully-slatted. Slurry removal frequency was less than one month in farrowing sows buildings and more than monthly for the rest of animal housing. Fogging coolers were implemented in gestating sows buildings. Slurry was stored in a single lagoon without natural crusting (8200 m^3 and 3280 m^2).The slurry was monthly applied by splash-plate on an arable field from July to October (50 t ha^{-1}). Table 4, shows the additional mitigation techniques selected in the four alternative scenarios proposed to this reference farm case.

Table 4. Application of abatement measures in alternative scenarios.

Activity	Reference (REF scenario)	Alternative Scenarios					
		Name				Abatement measure	
		T	S	N	AD	Description	Main effect
Fattening Feeding	Average C.P.16.4%, P. 0.55% Water: Nipple drinker in a cup			√		Average C.P. 14.10%, P. 0.40% Water: Weet feeder	Reduction off N and P excreted
Buildings	Slurry removal frequency major than monthly			√		Frequent slurry removal (weekly) 50% gestating sows buildings	Gas emission reduction: NH_3: 20% N_2O: 83% CH_4: 78-92%
	No exhaust air purification		√			Air scrubber in fattening houses[1]	NH_3 emission reduction: 74% (N_2O increases)
Storage	No cover			√		Flexible floating cover (impermeable)[2]	NH_3 emission reduction: 85%
Treatment	No treatment				√	Anaerobic digestion+Phase separation (screw press)[3]	CH_4 emission reduction
Landspreading (slurry)	Splash plate	√		√	√	Trailing hose	NH_3 emission reduction: 30%

[1] Lagoon surface increased to 3600 m^2 and capacity to 9000 m^3 (scrubber increases slurry production in 200 l/fattening place per year).
[2] Lagoon surface reduced to 2480 m^2 and capacity to 6200 m^3.
[3] 4500 t of fruits and vegetables wastes added to the reactor. Liquid phase after separation stored in 2500 m^2 tank.

Finally, software usability was evaluated by main stakeholders, through round table discussions and workshops.

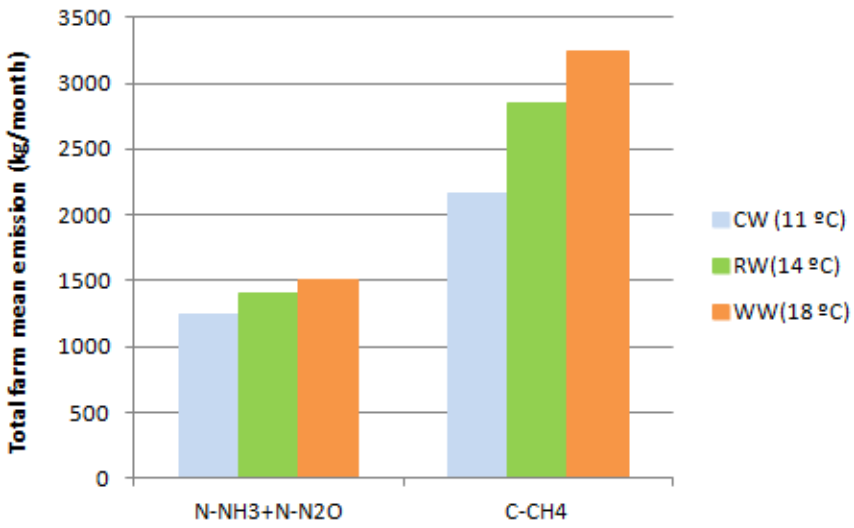
4.3. Results and Discussion

4.3.1. Model test

4.3.1.1. Climates

Predicted NH_3 and CH_4 were sensitive to temperatures both among the different climates selected and along the different seasons of the year (Figure 4 and Tables 5, 6, 7). As expected, the increase of outside temperature promoted NH_3 and CH_4 emission in all farm stages (housing, storage and landspreading). This pattern was also observed by other authors (Aarnink, Wagemans, Keen, 1993; Berthiaume, Bigras-Poulin, Rousseau, 2007; Sommer & Hutchings, 2001; Sommer *et al.*, 2006). In the case of N_2O, there was a slight decrease in total farm emission up to landspreading as the temperature in the different climates rises. This could be due to the reduction in the quantity of N in the slurry susceptible to be emitted as higher NH_3 volatilization occurs in previous phases.

Fig. 4. Predicted total farm mean emission (kg/month) under different climates. All combinations considered. Yearly mean temperature shown in brackets.



4.3.1.2. Nutritional practices

The different submodels captured the difference between climates and diets (Table 5). Higher emissions can be found in Warm Weather (WW) due to higher temperatures and lower precipitation. The analysis shows that decreasing protein consumption leads to a decrease in NH_3 and N_2O emissions. No effect on CH_4 was found as only protein and P content in feed were modified.

Table 5. Predicted gas emission under different climates and diets. Reference situation (RW and RD, grey colour) in kg per year, rest expressed in % of variation respect reference situation values.

Climate	Stage*	Diet	Gaseous losses			Diet	Gaseous losses			Diet	Gaseous losses		
			NH_3	N_2O	CH_4		NH_3	N_2O	CH_4		NH_3	N_2O	CH_4
CW	H		-1.9	0.0	0.0		-21.5	-12.6	0.0		-35.2	-22.3	0.0
	S		-37.0	0.0	-28.7		-43.8	0.0	-28.7		-49.2	0.0	-28.7
	L		-9.7	5.7	0.0		-18.9	-5.5	0.0		-26.6	-14.5	0.0
	Total		-11.7	4.4	-25.3		-24.8	-7.1	-25.3		-34.6	-16.3	-25.3
RW	H		9494	121	5792		-20.0	-12.6	0.0		-33.9	-22.3	0.0
	S	RD	4330	0	43631	BD	-10.7	0.0	0.0	BAPD	-19.4	0.0	0.0
	L		8576	396	0		-10.2	-10.5	0.0		-18.6	-19.0	0.0
	Total		22399	517	49423		-14.5	-11.0	0.0		-25.2	-19.8	0.0
WW	H		1.9	0.0	33.0		-18.4	-12.6	33.0		-32.6	-22.3	33.0
	S		23.9	0.0	9.9		10.7	0.0	9.9		0.0	0.0	9.9
	L		1.7	-5.8	0.0		-8.7	-15.6	0.0		-17.2	-23.6	0.0
	Total		6.1	-4.4	12.6		-9.1	-14.9	12.6		-20.4	-23.3	12.6

*H: Housing; S: Storage; L: Landspreading

Ammonia emission reduction at housing in relation to Reference Diet (RD) situation was 20% for Biphase Diet (BD) and 34% for Biphase+Aminoacids+Phytases Diet (BAPD) in average. These reductions are slightly higher than the ones reported by Canh et al. (1998) with a 10-12.5% lower NH_3 emission per 1% of CP decrease in pigs at 20ºC and ventilation rates of 55 m^3 h^{-1} pig^{-1}. Environmental conditions simulated at pig houses in the model were different (23.7ºC, 0.3-1.08 m^3 h^{-1} kg), which, together with other factors could explain differences found in results. Nevertheless, a previous study in Spain (MAGRAMA, 2014) showed an NH_3 reduction of 30% if CP was reduced from 16% to 13.6%.

If the whole farm scenario is considered, total NH_3 reduction achieved is reduced to 15% and 25% for Biphase Diet (BD) and Biphase+Aminoacids+Phytases Diet (BAPD) in comparison with Reference Diet (RD), that is, around 11% of NH_3 emission reduction per

1% of CP decrease. A similar result was obtained by Portejoie, Dourmad, Martinez and Lebreton (2004), who measured 13 % of NH_3 emission reduction per 1% of CP decrease from 16 to 12% during whole animal housing-storage-landspreading period.

4.3.1.3. Manure Management

According to the results, reducing outside storage duration decreases total farm NH_3 and CH_4 emission (Table 6). Ammonia volatilization in the storage is strongly influenced by the volatilization surface area (which is independent of the quantity stored), while CH_4 is more dependent of total organic matter stored. Thus, CH_4 emissions decreases in storage while as slurry removals increases from 2 (2MM) to 3 (3MM) periods. On the contrary, NH_3 emissions can even slightly increase. For a given climate, differences on NH_3 emissions during storage among managements will depend on ammoniacal nitrogen concentration; higher concentration promotes NH_3 emission (Figure 5).

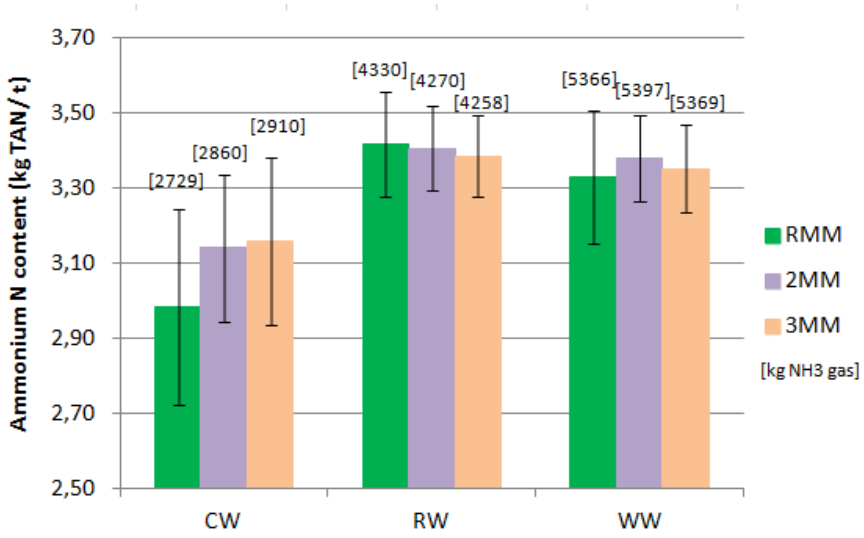
Table 6. Predicted gas emission under different climates and manure managements. Reference situation (RW and RMM, grey colour) in kg per year, rest expressed in % of variation respect reference situation values.

Climate	Stage*	Man.	Gaseous losses			Man.	Gaseous losses			Man.	Gaseous losses		
			NH_3	N_2O	CH_4		NH_3	N_2O	CH_4		NH_3	N_2O	CH_4
CW	H		-1.9	0.0	0.0		-1.9	0.0	0.0		-1.9	0.0	0.0
	S		-37.0	0.0	-28.7		-33.9	0.0	-47.8		-32.8	0.0	-56.5
	L		-9.7	5.7	0.0		-17.2	7.9	0.0		-15.6	8.5	0.0
	Total		-11.7	4.4	-25.3		-14.0	6.1	-42.2		-13.1	6.5	-49.9
RW	H		9494	121	5792		0.0	0.0	0.0		0.0	0.0	0.0
	S	RMM	4330	0	43631	2MM	-1.4	0.0	-31.8	3MM	-1.7	0.0	-46.2
	L		8576	396	0		-10.5	4.5	0.0		-8.7	6.2	0.0
	Total		22399	517	49423		-4.3	3.5	-28.0		-3.7	4.7	-40.8
WW	H		1.9	0.0	33.0		1.9	0.0	33.0		1.9	0.0	33.0
	S		23.9	0.0	9.9		24.7	0.0	-22.5		24.0	0.0	-36.7
	L		1.7	-5.8	0.0		-6.1	-1.2	0.0		-5.2	0.6	0.0
	Total		6.1	-4.4	12.6		3.3	-0.9	-16.0		3.5	0.5	-28.5

*H: Housing; S: Storage; L: Landspreading

In all cases NH_3 emission in landspreading is reduced when there are 2 (2MM) or 3 (3MM) periods of application in relation to reference scenario (RMM) with only one period. The reduction degree will depend on N available and emission factors in landspreading varying according to weather conditions in the month of application.

Fig. 5. Predicted annual average of TAN concentration in the slurry during storage under different climates and manure managements. Predicted NH_3 emissions in brakets.



4.3.1.4. Landspreading Equipment

Ammonia emissions during landspreading can be mitigated using equipments that reduce the contact between manure and atmosphere (Webb, Pain, Bittman, Morgan, 2010). Lowest NH_3 emission was achieved in SIL situation, followed by THL and the RL (Table 7). Values of NH_3 emission reduction during landspreading are in line with the ranges shown in EC (2013) and Bittman, Dedina, Howard, Oenema, and Sutton (2014), with 35% for THL and 82% for SIL situation in average.

Direct N_2O emission increases as more N reaches the soil, but can be compensated by practices which reduce NH_3 losses in landspreading due to lower indirect emission from redeposition of volatilised NH_3 and the reduction in N_2O emissions associated with chemical fertiliser N inputs offset.

Table 7. Predicted gas emission under different climates and landspreading equipment. Reference situation (RW and RL, grey colour) in kg per year, rest expressed in % of variation respect reference situation values.

Clim.	Stage*	Eq.	Gaseous losses			Eq.	Gaseous losses			Eq.	Gaseous losses		
			NH_3	N_2O	CH_4		NH_3	N_2O	CH_4		NH_3	N_2O	CH_4
CW	H		-1.9	0.0	0.0		-1.9	0.0	0.0		-1.9	0.0	0.0
	S		-37.0	0.0	-28.7		-37.0	0.0	-28.7		-37.0	0.0	-28.7
	L		-9.7	5.7	0.0		-41.5	-9.9	0.0		-84.1	-30.8	0.0
	Total		-11.7	4.4	-25.3		-23.9	-7.6	-25.3		-40.2	-23.6	-25.3
RW	H		9494	121	5792		0.0	0.0	0.0		0.0	0.0	0.0
	S	RL	4330	0	43631	THL	0.0	0.0	0.0	SIL	0.0	0.0	0.0
	L		8576	396	0		-34.7	-17.0	0.0		-80.9	-39.7	0.0
	Total		22399	517	49423		-13.3	-13.0	0.0		-31.0	-30.4	0.0
WW	H		1.9	0.0	33.0		1.9	0.0	33.0		1.9	0.0	33.0
	S		23.9	0.0	9.9		23.9	0.0	9.9		23.9	0.0	9.9
	L		1.7	-5.8	0.0		-34.7	-23.6	0.0		-83.2	-47.4	0.0
	Total		6.1	-4.4	12.6		-7.8	-18.1	12.6		-26.4	-36.3	12.6

*H: Housing; S: Storage; L: Landspreading

4.3.2. Basic quality check

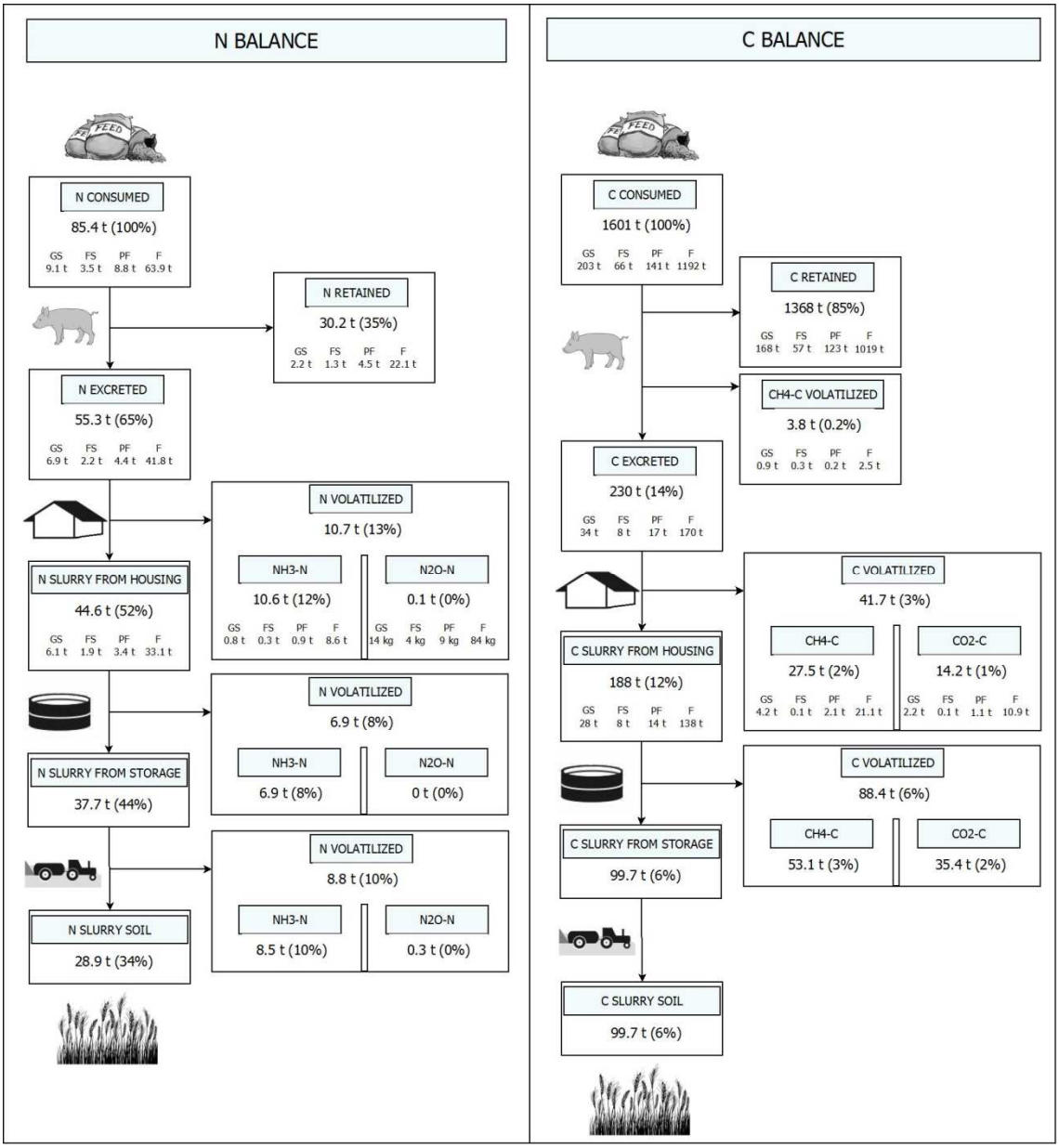
4.3.2.1. Mass balance

Nitrogen is a key element regarding its major role within production but its efficiency in farms, defined as the ratio between animal products and N input, is quite low (Van der Hoek, 1998). Predicted N retention in pig tissues in our model represents for the reference 35% of total N consumed by animals (Figure 6), similar to the 33% reported in Ajinomoto (2000).

The N that has not been retained by the animal is excreted and susceptible of being lost during manure handling in volatile form or in the form of nitrate leaching and runoff. Oenema, Oudendag, and Velthof (2007) indicates that maximally 52% of the N excreted in housing can be effectively recycled as plant nutrient, the rest is lost. Ajinomoto (2000) estimates that the 32% of total protein consumed by fattened pigs would remain in the manure spread on the soil. Our model predicts similar values (Figure 6): in the scenario simulated, 34% and 52% of total N consumed and excreted by animals, respectively, reaches the soil for potential uptake by the crops.

According to model predictions in the scenario proposed, most part of C intake (85%) is retained by the animals while the 14% is excreted at slurry pits and 8% will be volatilized afterwards during slurry storage. The estimated quantity of C that reaches the soil is a 6% of total C consumed (Figure 6).

Fig. 6. N and C farm balance, showing the percentage contribution of outputs over the total N or C feedstuff input. GS: Gestating sows, FS: Farrowing sows, PF: Prefattened piglets (5-22 kg), F: Fattened pigs (22-110 kg).



4.3.2.2. Comparison with literature data

Ammonia losses estimated at housing were in consonance with the ranges reported by EU member states inventories (Table 8). It can be seen that NH_3 emission range reported by the draft BREF document (EC, 2013) is especially wide for fattening pig facilities. This variability can be related to the different types of facilities and farming practices existing across Europe, and which have been considered for the national inventories. Arogo, Westerman and Heber (2003) reviewed NH_3 emission rates measured in EU for fattening buildings with slatted floor and deep pits (slurry removal frequency < 1 month) and the presented values were similar to the mean annual NH_3 emission estimated in this case study. Despite the fact that NH_3 emissions were measured in different climates and seasons, mean NH_3 rate was 2.5 kg NH_3 place^{-1} year^{-1}. Model simulation also estimated the monthly NH_3 losses, which ranged from 2.2 to 3.5 kg NH_3 place^{-1} year^{-1}. Similar values for fattening farms were found by Botermans, Gustafsson, Jeppsson, Brown and Rodhe (2010), from 2.4 to 3 kg NH_3 place^{-1} year^{-1}. Ammonia emission factor predicted by the model for prefattening pigs, 0.57 kg NH_3 place^{-1} year^{-1}, is also in line with Botermans et al. (2010) values, which ranged from 0.6 to 0.8 kg NH_3 place^{-1} year^{-1}and with EC (2013) values (Table 8).

Table 8. Comparison of NH_3 emission at housing predicted by the model and reported by EC (2013).

	kg NH_3-N Kg N excreted^{-1}		**Kg NH_3 place^{-1} year^{-1}**	
	Model	**EC (2013) Table 3.55**	**Model**	**EC (2013) Tables 3.55 3.56**
Sows [a]	0.12	0.14-0.17	2.84	3.01-5.5
Prefattening	0.22	0.2	0.57	0.21-0.78
Fattening	0.20	0.15-0.33	2.81	0.54-7

[a] Farrowing sows, gestating sows and gilts

Predicted NH_3 and CH_4 emission rates for sows were slightly under EC (2013) and Jungbluth, Hartung and Brose (2001) values (Table 8 and 9). Botermans et al. (2010), indicated emissions of 8.3-8.7 and 3.1-4.2 kg NH_3 place^{-1} year^{-1} for farrowing and gestanting sows, respectively, higher than those estimated by the model, 6.63 kg NH_3 place^{-1} year^{-1} for farrowing sows and 2.27 kg NH_3 place^{-1} year^{-1} for gestating. The model considers that the use of fogging coolers in gestating buildings and slurry frequencies of less than one month in farrowing sows buildings reduces gas emission. Indeed, if the example is run without fogging coolers, the emission predicted for sows is in the range of values shown in Tables 8

and 9. Results of N_2O and CH_4 at housing for prefattening and fattening pigs agreed reasonably well with data reported in Tables 9 and 10.

Table 9. Comparison of CH_4 emission at housing predicted by the model and reported in the literature.

	Kg CH_4 place^{-1} year^{-1}			g CH_4 animal^{-1} day^{-1}		
	Model	**Source**		**Model**	**Source**	
Sows	15.56	19	GrootKoerkamp and Uenk (1997) [a]			
Prefattening	1.53	3.5		0.81	0.75-1.03	Cabaraux et al. (2009)
Fattening	8.51	10		8.7	5.4 9.7	Guingand et al. (2010) [b] Philippe et al. (2014) [b]

[a] In Jungbluth et al. (2001) [b] In Philippe and Nicks (2015)

Table 10. Comparison of N_2O emission at housing predicted by the model and reported in the literature.

	Model	**Source**	
Prefattening mg N_2O animal^{-1} day^{-1}	3.64	0-10	Cabaraux et al. (2009)
Fattening Kg N_2O place^{-1} year^{-1}	0.04	0.02-0.15	EC (2013) Table 3.58

Regarding NH_3 emissions from storage, the model predicts an annual mean emission of 2.09 kg NH_3-N m^{-2} year^{-1}, while Sommer et al. (2006) showed values of 0.78±1.07 kg NH_3-N m^{-2} year^{-1} for lagoons. This difference could be attributed to the warmer and drier conditions considered in the simulation. EC (2013) reported different national NH_3 EF for storage of pig slurry, 14.4 kg NH_3-N place^{-1} year^{-1} for sows in full cycle, very similar to model calculations, 14.8 kg NH_3-N place^{-1} year^{-1}. Annual mean CH_4 emission predicted by the model, 684 μg m^{-2} s^{-1}, is higher than mean value measured by Park & Wagner-Riddle (2010), 422 μg m^{-2} s^{-1} under lower temperatures and higher precipitation rates. Similar values to those estimated by the model, 5.7 g NH_3-N m^{-2} day^{-1} and 49.4 g CH_4-C m^{-3} day^{-1} were found by Loyon, Guiziou, Beline, and Peu (2007), 6.7 g NH_3-N m^{-2} day and 49.8 g CH_4-C m^{-3} day^{-1}, in a raw slurry pit store without stirring. Liu, Powers, Murphy and Maghirang (2014) indicated a lower mean value, 3.8 g NH_3-N m^{-2} day^{-1} for several swine slurry lagoons in North America. Nitrous oxide emission predicted by the model is zero, as there was no surface cover or crust (IPCC, 2006).

In landspreading, according to model results 28% of TAN applied was volatilized as NH_3, which was in the low end of the range indicated by EC (2013) of 30-100%. Bittman

et al. (2014) reported that typical range was 40-60% of TAN applied in arable crops, although emissions outside this range could be also common. Misselbrook et al. (2000) showed values for NH_3 volatilization from 15% to 59% of applied TAN depending on DM content of pig slurry (<4% to >8%). In our case study, DM content of the slurry was below 4%, so according to this source even lower values than 28% would be reasonable. In relation to N_2O emissions, the model predicted that 0.81 % of TN applied was lost, which was in the range shown by EC (2013) of 0.12-2.95 % and IPCC (2006) of 0.3-3%.

Volume of slurry produced per head in housing was compared to Babot, Andrés, de la Peña and Chávez (2004) and EC (2013) (Table 11). The slurry produced by prefattening and fattening pigs was in the same range. The model predicted that 66% of the water was consumed during fattening period, accounting for 4.76 l animal^{-1} day^{-1}. Muhlbauer, Moody, Burns, Harmon and Stalder (2010) also found similar consumptions for fattening pigs using nipple drinkers in a cup. In the case of sows, model results were slightly higher or close to the upper end than Babot et al. (2004) and EC (2013) respectively (Table 11). This fact might be explained by the dilution coeficients used in the simulation for pigs, with typical values under WS climate (from 2.9 to 3.7 liters of water consumed kg feed^{-1}), which can be different from other climatic and production circumstances.

Table 11. Volume of slurry produced in housing.

	m^3 head^{-1} month^{-1}			**l head^{-1} day^{-1}**		
	Model	**EC (2013) Table 3.38**		**Model**	**Babot et al. (2004)**	
		Min	**Max**		**Min**	**Max**
Gestating sows	0.49 [a]	0.16	0.28	15.91 [a]	4.18	16.1
Gilts		0.11			6.84	10
Farrowing sows	0.62	0.43		20.24	10.26	19.7
Prefattening	0.05	0.04	0.05	1.55	1.14	2.03
Fattening	0.15	0.09	0.26	4.76	1.9	7.28

[a] Including gilts

Nutrients content (N, P, K) were near the values found by other authors (Table 12). It is important to consider that besides manure composition from housing, climate conditions and storage surface will impact on manure concentration values after outdoor storage, what might explain the differences found when comparing with bibliography (Table 12).

Table 12. Slurry nutrients content after outdoor storage (on wet matter basis).

	Kg t^{-1}		Kg m^{-3}		
	Model	**EC (2013) Table 3.45**	**Model**	**Babot et al. (2004) [a]**	
				Min	**Max**
Nitrogen	3.66	4.3	3.75	3.5	5.4
Phosphorus	1.19	1.7	1.22	1.4	2.7
Potassium	2.26	2.2	2.32	1.8	2.3

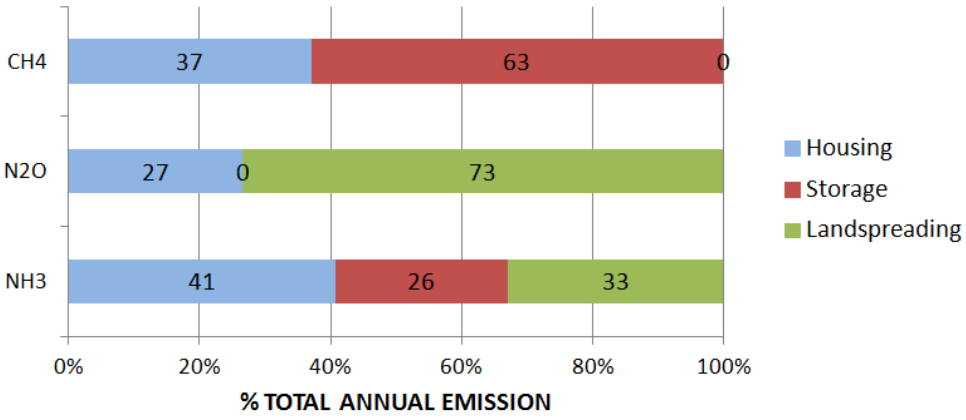
[a] Farrow to Finish farm

4.3.3. Application of abatement measures in alternative scenarios

According to model results, most of housing gas emissions is from fattening (81% of NH_3 and 76% of N_2O and CH_4), therefore fattening period seems to be the key stage to implement environmental techniques at housing. While NH_3 emission is produced in all stages, most of N_2O is volatilised during landspreading and CH_4 during storage (Figure 7). Highest farm emissions can be found in warmer months and slurry landspreading periods.

Alternative scenarios considered for gaseous emissions abatement are presented in Table 4. Most NH_3 and N_2O reduction is achieved in N scenario with a 37% and a 18% of reduction, respectively, in comparison with the reference situation (Figure 8). Nutritional techniques implemented in N scenario decrease the amount of N and P excreted. Subsequent BAT implemented throughout the whole farm process (housing, storage and landspreading) in this scenario, can avoid NH_3 emissions in the following phases, obtaining the highest N efficiency (64% of the N excreted reaches the soil) (Table 13). Reductions in water consumed and wasted by fattening pigs falls from 8780 m^3 in reference situation to 5827 m^3 per year in N scenario. The amount of slurry to be managed is reduced and nutrients are more concentrated than in other scenarios (Table 13). This can reduce manure management cost (less storage capacity required and slurry tanks to be applied), although the total amount of N and P available in soil is lower than S and T scenarios (Table 13). These savings related to water and slurry handling together with feed adjustment might offset part of the BAT implementation costs.

Fig. 7. Emission distribution in reference situation.



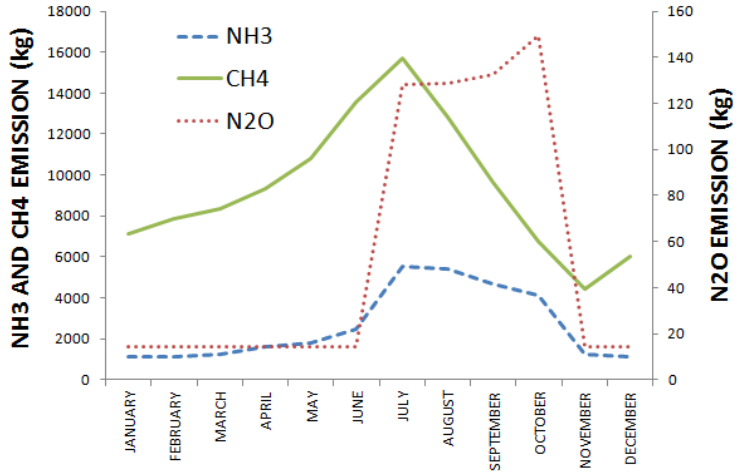


Fig. 8. Emission distribution in swine farm scenarios.

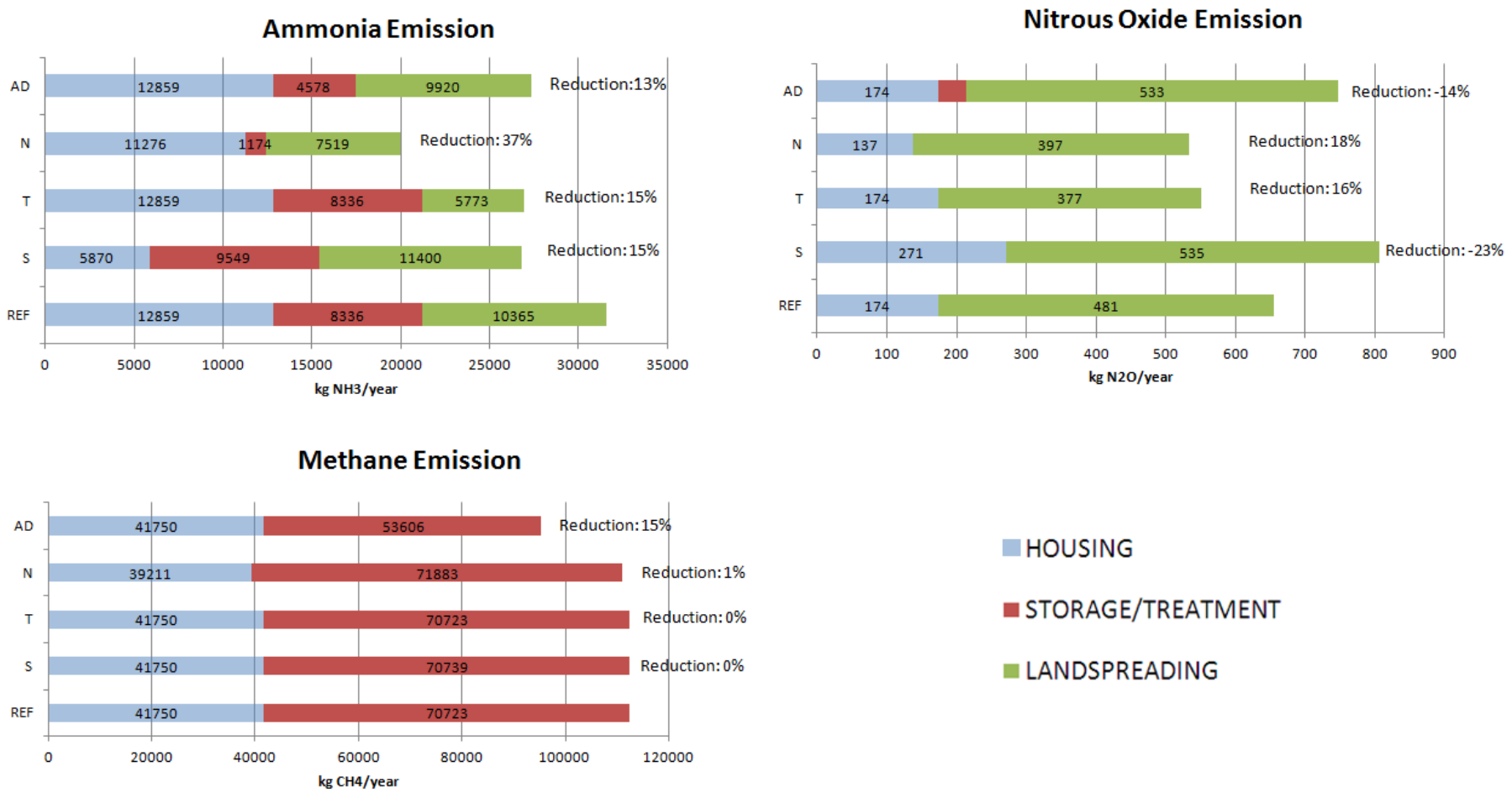


Table 13. Manure composition and nutrients in soil for the different swine farms scenarios

	PARAMETER	UNITS	REF	T	S	N	AD* (Liquid Phase)	AD* (Solid Phase)
MATERIAL BEFORE LANDSPREADING	Quantity	tons	10317	10317	11131	7421	13724	414
	Dry Matter	% TM	2.0	2.0	1.9	2.9	1.8	36.7
	Organic Matter	% TM	1.7	1.7	1.5	2.4	1.3	24.9
	Nitrogen	kg/t TM	3.7	3.7	3.8	4.8	4.5	9.9
	Ammoniacal Nitrogen	kg NH_4-N/t	3.0	3.0	3.1	4.0	3.7	5.1
	Phosphorus	kg P/t	1.2	1.2	1.1	1.2	0.8	7.9
	Potassium	kg K/t	2.3	2.3	2.1	3.1	1.6	2.7
	Copper	g/t TM	11.9	11.9	11.0	16.5	8.2	24.1
	Zinc	g/t TM	45.0	45.0	41.7	62.5	30.2	119.4
NUTRIENTS IN SOIL AFTER LANDSPREADING	**PARAMETER**	**UNITS**	**REF**	**T**	**S**	**N**	**AD***	
	N	Kg/year	28802	32584	32594	29424	57088	
	N in soil from N excreted	%	52	59	59	64	66	
	P_2O_5	Kg/year	27983	27983	27982	19616	32826	
	K_2O	Kg/year	28110	28110	28109	28109	28110	

*Addition of coproduct (4500 t of fruits and vegetables wastes into the reactor)

Trailing hose (T scenario) as the last element in the process chain, is an efficient individual abatement measure. Reducing NH_3 losses during landspreading, increases the amount of N in the soil and the fertilized potential of the slurry.

Air scrubber (S scenario) although technically effective, can lose its efficiency if applied without any further measures in downstream farming activities. If emissions are reduced in housing, more NH_3 reaches the storage, increasing the emissions there. Consequently, part of the mitigation effect in housing is lost. Moreover, S scenario promotes N_2O emissions, 23% higher than reference situation due to nitrification-denitrification processes in scrubber water and indirect emissions on landspreading. The model assumes that discharged water of the scrubber goes to slurry pit, producing an increase on slurry volume to be managed. Nutrients content in soil after application and NH_3 reduction are similar to T scenario, but probably with higher cost per kg of NH_3 abated.

AD scenario reduces CH_4 emission, but NH_3 emission abatement is similar to T and S scenarios with a 13% of reduction and even increases N_2O emissions comparing to reference (Figure 8). In this scenario, the quantity of ammoniacal nitrogen susceptible to be

volatilized in landspreading is higher than in other situations. Therefore, implementing techniques that strongly decrease NH_3 losses during landspreading, like injectors or rapid incorporation, are of great interest in AD scenario. Indeed, incorporing the digestate into the soil 4 hours after spreading with trailing hose, would reduce 41% and 16% respectively of total NH_3 and N_2O emissions, comparing to the reference situation. The interest of including coproducts in anaerobic digestion, is to increase the biogas produced in the reactor and in some occasions to perceive remuneration for waste management. According to model calculations and considering an energy consumption of 983 kWh/sow and year (Marcon, 2008), AD scenario would produced enough energy for 1347 sows. In this example, the addition of coproduct into the digester, has increased total energy production by 180%.

Regarding to FIOs evaluation, trailing hose use for landspreading and anaerobic digestion treatment (thermophilic conditions) were found to have a positive effect on pathogens mitigation.

4.3.4. Model Usability

Before being released for regular use, the software has been tested by final users (farmers and advisors). Firstly, a software prototype was delivered to 8 farm advisers (agronomists and veterinarians) to be used during four weeks. This was followed by round table discussions on insights and reactions around the themes of usability, quality and reliability of results to help us improving the model. Then, a software workshop addressed to regional stakeholders (farmers, advisors and environmental inspectors) was organized in Navarre (Spain). After demonstrating main tool functions, the 36 attendees were asked to score the practicality and the user-friendlyness of the software, reaching an average score of 4.33 and 4 (range 1 to 5), respectively.

4.4. Discussion and conclusion

The model behaviour study conducted seems to demonstrate that it responds to those factors having an important impact on volatilisation, this is in consonance with current knowledge and constitutes a confirmation of model coherence. However, further validation would be required to evaluate its reliability and enable broaden uses, not only for

scenario comparisons but also for the calculation of absolute farm emissions. This aspect will be addressed in the future, testing model results against empirical data and comparing it with similar tools.

The model can also be used to compare N and C efficiency of different farm scenarios (constructing whole farm balance sheets), leading both to an economic profit and an environmental benefit.

The application scenarios illustrated well model flexibility and its potential as a tool for management purposes. Furthermore, this information has to be integrated with current economic data to help formulate cost-effective strategies to reduce farm pollution.

The model offers a number of mitigation techniques to be simulated, including measures for housing, storage/treatment and incorporation of manure. These measures, however, largely differ with regard to their effectiveness and related costs. As has been shown in the application scenarios, the scale of operation as well as the sequence of measures from feeding and housing to storage/treatment and application of manure may strongly influence the efficiency and needs to be considered in the assessment of abatement measures.

The effect of emission reduction at the production point influences the N quantity in the following step and therefore also the quantity of potential NH_3 emissions. While trailing hose is an efficient individual abatement measure, air scrubber loses its efficiency if applied without any further measures in downstream farming activities. When the amount of N increases in the slurry (e.g. after the adition of coproduct in anaerobic digestion) the reduction measures in the following stage become more cost-effective. Through a combination of mitigation techniques in different farming activities, maximum emission abatement is achieved. Nutritional strategies not only have the advantage of possible benefits due to saved feeding costs and manure management, but also reduce the N available for downstream losses of NH_3. In order to model these interrelationships, farms must be approached as a holistic process.

A challenge faced by the model is its data-intensive nature. Although different levels of database accessibility have been designed to develop a widely used and versatile tool, the availability and the quality of data will constrain model parameterization for specific farms

conditions. The degree of model adjustment and the quality of results will depend on users' knowledge and data availability. The end user of the model must ensure that the input data describing the system is consistent.

In conclusion, farms are complex systems with several interacting subsystems. The large number of available techniques, the diversity of farming systems and of socioeconomic conditions, dificults decision making process for BAT selection. This whole-farm model considers relevant interactions between farm components and is useful for integrated scenario development and evaluation. It is intended that the model outputs will be used to engage stakeholders, providing more evidence based information encouraging them to explore possible farm mitigation options in their operational, tactical and strategic management.

4.5. Acknowledgements

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4.6. References

Aarnink A.J.A., Wagemans M.J.M. and Keen, A. (1993). Factors affecting ammonia emission from housing for weaned piglets. Proceedings of the Congress on Nitrogen Flow in Pig production and Environmental Consequences, EAAP Publishers, Wageningen, The Netherlands, 286-294.

Aguilar M., Arriaga H., Dupard P., Lalor S., Fragoso R., Pahl O., Abaigar A., Cordovín L., Boyle M., Lanigan G., Loyon L., Duarte E., Merino P. (submitted for

publication). A whole farm model for Ammonia and Greenhouse Gas Emissions from Livestock Operations. Part 1: Model description.

Ajinomoto Animal Nutrition (2000). Prevention of nitrogen pollution from pig husbandry through feeding measures. Booklet 22, Ajinomoto Eurolysine, Paris, 12 pp.

Arogo J., Westerman P. W. and Heber A. J. (2003). A review of ammonia emissions from confined swine feeding operations. Transactions of the ASAE, 46(3), 805-817.

Babot D., Andrés N., de la Peña L. and Chávez E.R. (2004). Técnicas de Gestión Medioambiental en Producción Porcina. Proyecto TRAMA (LIFE). Edición de la Fundación Catalana de Cooperación, 135 pp.

Berthiaume P., Bigras-Poulin M. and Rousseau A.N. (2007). Sensitivity Analysis and Application of a Dynamic Simulation Model of Nitrogen. Fluxes in Pig Housing and Outdoor Storage Facilities. Biosystems Engineering 96 (4), 455-470.

Bittman S., Dedina M., Howard C.M., Oenema O. and Sutton M.A. (2014). Options for Ammonia Mitigation: Guidance from the UNECE Task Force on Reactive Nitrogen. Centre for Ecology and Hydrology, Edinburgh, UK, 83 pp.

Botermans J., Gustafsson G., Jeppsson K-H, Brown N. and Rodhe L. (2010). Measures to reduce ammonia emissions in pig production. Review. Landscape Horticulture Agriculture Report series, Swedish University of Agricultural Sciences, Report 2010:12, 63 pp.

Cabaraux J-F, Philippe F-X, Laitat M., Canart B., Vandenheede M. and Nicks B. (2009). Gaseous emissions from weaned pigs raised on different floor systems. Agriculture, Ecosystems and Environment 130, 86-92.

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

EC (2003).Reference Document on Best Available Techniques for intensive Rearing of Poultry and pigs.Integrated Pollution Prevention and Control (IPPC), July 2003, 341 pp.

EC (2013). Best Available Techniques (BAT) Reference Document for the Intensive Rearing of Poultry and Pigs. Integrated Pollution Prevention and Control (IPPC), Draft 2-August 2013, 839 pp.

Hutchings N.J., Sommer S.G., Andersen J.M. and Asman W.A.H., (2001). A detailed ammonia emission inventory for Denmark. Atmospheric Environment, 35, 1959-1968

IPCC (2006). IPCC guidelines for national greenhouse gas inventories, prepared by the national greenhouse gas inventories programme. Eds H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, K. Tanabe, Hayama, Japan: IGES, 87 pp.

Jungbluth T., Hartung E. and Brose G. (2001). Greenhouse gas emissions from animal houses and manure stores. Nutrient Cycling in Agroecosystems 60, 133-145.

Liu, Z., Powers W., Murphy J. and Maghirang R. (2014). Ammonia and Hydrogen Sulfide Emissions from Swine Production Facilities in North America: a Meta-Analysis. Journal of Animal Science 92 (4), 1656-1665.

Loyon L., Guiziou F., Beline F., Peu P. (2007). Gaseous Emissions (NH_3, N_2O, CH_4 and CO_2) from the aerobic treatment of piggery slurry - Comparison with a conventional storage system. Biosystems Engineering, 97(4), 472-480

MAGRAMA (2014). Evaluación de Técnicas de Reducción de Emisiones en Ganadería. Sectores de Porcino y Avicultura de Carne y Puesta. Ministerio de Agricultura, Alimentación y Medio Ambiente, Madrid, 114 pp.

Marcon M. (2008). L'énergie dans les bâtiments d'élevage porcin. 8ème édition de la Journée des Productions Porcines et Avicoles, 42 pp.

MARM (2010). Caracterización de sistemas de gestión de deyecciones. Sector porcino intensivo. Ministerio de Medio Ambiente y Medio Rural y Marino, Madrid, 41 pp. http://www.magrama.gob.es/es/ganaderia/publicaciones.

Meda B., Hassouna M., Aubert C., Robin P. and Dourmad J.Y. (2011). Influence of rearing conditions and manure management practices on ammonia and greenhouse gas emissions from poultry houses. World's Poultry Science Journal 67, 441-455.

Menzi H., Bonjour C., Zaucker F., Leuenberger C. and Reidy B. (2009). Agrammon: A New Internet Based Model for the Differentiated Estimation of Ammonia Emissions from Individual Farms or at the Area-Wide Scale. Ecology and Future-Bulgarian Journal of Ecological Science, 8(1), 28-32.

Misselbrook T.H., Van Der Weerden T.J, Pain B.F., Jarvis S.C., Chambers B.J., Smith K.A., Phillips V.R., Demmers T.G.M. (2000). Ammonia emission factors for UK agriculture. Atmospheric Environment 34, 871-880.

Muhlbauer R.V., Moody L.B., Burns R.T., Harmon J. and Stalder K. (2010). Water consumption and conservation techniques currently available for swine production. Research Report Environment. National Pork Board: Des Moines, LA, 32 pp.

Oenema O., Oudendag D. and Velthof G.L. (2007). Nutrient losses from manure management in the European Union. Livestock Science 112 (3), 261-272.

Park K.H. and Wagner-Riddle C. (2010). Methane Emission Patterns from Stored Liquid Swine Manure. Asian-Aust. J. Anim. Sci. 23 (9), 1229-1235.

Philippe F.X. and Nicks B. (2015). Review on greenhouse gas emissions from pig houses: Production of carbon dioxide, methane and nitrous oxide by animals and manure. Agriculture, Ecosystems and Environment 199, 10-25.

Philippe F.X., Cabaraux J.F. and Nicks B. (2011). Ammonia emissions from pig houses: Influencing factors and mitigation techniques. Agriculture, Ecosystems and Environment, 141, 245-260.

Portejoie S., Dourmad J.Y., Martinez J. and Lebreton Y. (2004). Effect of lowering dietary crude protein on nitrogen excretion, manure composition and ammonia emission from fattening pigs. Livestock Production Science 91, 45-55.

Rigolot C., Espagnol S., Robin P., Hassouna M., Béline F., Paillat J. M. and Dourmad J.Y. (2010). Modelling of manure production by pigs and NH_3, N_2O and CH_4 emissions. Part II: effect of animal housing, manure storage and treatment practices. Animal 4 (8), 1413-1424.

Schils R.L.M., Olesen J.E., del Prado A. and Soussana J.F. (2007). A review of farm level modelling approaches for mitigating greenhouse gas emissions from ruminant livestock systems. Livestock Science 112, 240-251.

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

Sommer S.G. and Hutchings N.J. (2001). Ammonia emission from field applied manure and its reduction–invited paper. European Journal of Agronomy 15 (1), 1-15.

Van der Hoek K.W. (1998). Nitrogen efficiency in global animal production. Environmental Pollution 102, 127-132.

Webb J, Broomfield M, Jones S, Donovan B (2014). Ammonia and odour emissions from UK pig farms and nitrogen leaching from outdoor pig productin. A review. Science of the Total Environment. 470-471: 865-875.

Webb J., Pain B., Bittman S. and Morgan J. (2010). The impacts of manure application methods on emissions of ammonia, nitrous oxide and on crop response: a review. Agriculture, Ecosystems & Environment 137, 39-46.

Webb J., Ryan M., Anthony S.G., Brewer A., Laws J., Aller M.F. and Misselbrook T.H. (2006). Cost-effective means of reducing ammonia emissions from UK agriculture using the NARSES model. Atmospheric Environment 40, 7222–7233.

Performance and costs of two biotrickling filters to remove gaseous pollutants from a pig house

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Operational factors affect the performance and costs of two biotrickling filters to remove gaseous pollutants from the exhaust air of a pig house

Abstract

Air scrubbers have demonstrated to be effective to reduce NH_3 emissions in livestock houses. Nevertheless, the effect of operational parameters and potential nitrous oxide (N_2O) emissions has been identified as improvement aspects. In this study, two identical biotrickling filters installed in a pregnant sows facility in Northern Spain were monitored. Two tests were conducted (Test 1: 45 days during 2009; Test 2: 254 days during 2012 and 2013) changing the removal frequency of washing water, from no discharge to two weekly water discharges. Ammonia, carbon dioxide (CO_2), methane (CH_4) and N_2O were continuously monitored before and after the scrubber using a photoacoustic gas monitor. Washing water nitrogen species, pH and electric conductivity were determined, and the installation and operation costs were estimated. The removal efficiency of NH_3 varied between scrubbers and was affected by washing water management (from 17% to 81% in Scrubber 1, and from 91% to 92% in Scrubber 2). The development of biofilm differed among scrubbers and this was considered to originate NH_3 removal differences. Water discharge management also affected the concentration of the different nitrogen species in water. Both scrubbers were found to be a net source of N_2O, increasing the outlet concentration from 14% to 74% with respect to the concentration of the inlet air. The scrubber did not affect CH_4 and CO_2 concentrations. Costs ranged from 29.34 to 32.16 Eur per sow and year. The results evidence how similar scrubbers may have very different performances in practice due to management and operation.

Keywords

Scrubber, Ammonia, Nitrous Oxide, Cost Analysis, Nitrification, Denitrification, Nitrogen Balance, Biofilm.

5.1. Introduction

Intensive livestock production is the main source of ammonia (NH_3) emissions to the atmosphere (EEA, 2016). These emissions are associated to impaired indoor air quality for animals and farmers, and when released to the atmosphere they constitute a major contributor to acidification of soils and eutrophication of water (Behera, Sharma, Aneja, Balasubramanian, 2013). Furthermore, NH_3 emitted to the atmosphere is a precursor of secondary particulate matter, and therefore reducing these emissions also contribute to improve environmental air quality (Backes, Aulinger, Bieser, Matthias, Quante, 2016; Cambra-López, Aarnink, Zhao, Calvet, Torres, 2010). For all these reasons, countries in the European Union are committed to lower their emissions through the establishment of ever-more stringent emission ceilings. To achieve these ceilings, mitigation strategies for ammonia emissions are required, which focus on the livestock sector as a major source.

Pig production is the largest livestock sector in terms of meat quantity worldwide (FAOSTAT, 2016). In the European Union, abating ammonia emissions in the pig sector has received a major attention in recent years because it is affected by the Directive 2010/75/EU on industrial emissions, and farms over certain sizes are committed to implement the best available techniques (BAT). Recently, a draft document was reported by the IPPC Bureau collecting these techniques (EC, 2015). To minimize emissions, major efforts should be conducted to increase the efficiency of the production process, reducing nitrogen (N) inputs and considering the global process (housing, manure storage and land application). However, most of the measures conducted at the housing level (e.g. nutritional strategies, housing design or manure management inside the house) are of limited mitigation potential (Philippe, Cabaraux, Badouin, 2011) and the only way to achieve further NH_3 emission reductions (up to more than 70%) is the use of end of pipe techniques such as air scrubbing systems (Melse, Ogink, Rulkens, 2009a).

Air scrubbers are widely used in densely populated European regions (e.g. the Netherlands, Northern Germany or Denmark) because of local regulations on ammonia reductions, and in some cases they are used in combination with biofilters to reduce odours (Melse et al., 2009a). Standardized measurement protocols have also been recently developed (VERA, 2016). As reviewed by Van der Heyden, Demeyer and Volcke (2015), different scrubbing systems can be found in practice, depending on the use of cleaning

water without or with addition of acid, the bed materials and thickness, the disposition in the farm (horizontal or vertical flow direction), or the use or not of biological processes to remove the excess nitrogen. A number of studies on the evaluation of different types of scrubbers can be found in the literature, particularly in pig production (Chou & Wang, 2007; Dumont, Hamon, et al., 2014; Estellés, Melse, Ogink, Calvet, 2011; Melse, Ploegaert, Ogink, 2012).

Among air scrubbers, biotrickling filters or bioscrubbers are relatively simple in management because they only use cleaning water, but for this reason these systems also have relatively lower efficiencies (around 70% NH_3 reduction) than acid scrubbers (typically around 90% reduction) (Melse, Ogink, Rulkens, 2009b; Van der Heyden et al., 2015). Biotrickling filters rely on nitrification and denitrification processes to remove the excess nitrogen. However, this converts this technology in an unintentional source of nitrous oxide (N_2O) (Melse & Mosquera, 2014). This is a greenhouse gas with a high global warming potential, and therefore recent research is focusing on quantifying and reducing these emissions (De Vries & Melse, 2014; Dumont, Lagadec, Landrain, Landrain, Andrès, 2014; Frutos, Arvelo, Pérez, Quijano, Muñoz, 2015).

It is also remarkable that air scrubbing systems are mostly used in European temperate areas where intensive animal production is located in densely populated areas. In Southern Europe there are also highly intensive producing areas, but scrubbing systems are infrequent probably as a consequence of lower pressure for environmental protection. Also, ventilation systems for pigs may differ as a consequence of the warmer environmental conditions. For example, natural ventilation is used more frequently for pig production. For these reasons, very limited information is available on the performance and costs of air scrubbing systems in warmer conditions.

Therefore, the aim of this study was to evaluate the performance and cost-effectiveness of a biotrickling filter in Southern Europe from a wide perspective in order to assess potential use and limitations. Particularly, the effect of washing water management was investigated.

5.2. Materials and methods

5.2.1. Livestock facility

The experiments were conducted in a commercial pig farm located in Zurucuain (Navarre, Spain). Two biotrickling filters were installed at the exhaust air system of a commercial building for pregnant sows accomplishing with the Council Directive 2008/120/EC laying down minimum standards for the protection of pigs. The building had dimensions of 78 x 28 m and housed between 900 and 1000 sows which were housed in collective pens of 12 to 14 animals each. The real number of sows was taken from the farmer records and used to provide results per sow in the farm. Sows were fed conventional restricted nutrition for pregnant sows and had permanent access to water. The floor was fully slatted and manure accumulated underneath it for about three to four months. The building was equipped with mechanical pit ventilation, which means that air entered the building through lateral windows and was exhausted from the pits below the animals (Figure 1).

Airflow was forced by means of variable speed fans, and airflow rate was set automatically to maintain temperatures between 21 and 28 ºC. To determine ventilation rates, the methodology described by Calvet, Cambra-López, Blanes-Vidal, Estellés and Torres (2010) was followed: firstly, fans were calibrated at the beginning and at the end of each test and during the experiment fan speed variation was continuously monitored from the fan control system. Calculated ventilation rates ranged between 22 and 130 m^3/h per sow. Indoor and outdoor temperature and relative humidity were registered every 5 minutes using HOBO data loggers (U12-013, Onset Computer Corporation, Bourne MA, USA).

5.2.2. Scrubbing system design and operation

The scrubbing system consisted in two identical packing beds (Scrubber 1 and Scrubber 2) which operated in parallel, each treating approximately half of the airflow exhausted from the building (Figure 1). Similar to the scrubbers described by Melse et al. (2012), biotrickling filters operated in counter-current, with airflow forced in upward direction and nozzles irrigating the packing bed from the top. Recirculation of water was continuous but discharge of trickling water was modified depending on the trial, as

described below. Evaporated water was added automatically to keep level when the water in the tank reached a minimum of 1.39 m^3. Discharged water was conducted to the slurry pits inside the building. A summary of characteristics of the biofilters is detailed in Table 1.

Fig. 1. Disposition of the pens in the farm and air circulation (up) and detail of the scrubber (down)

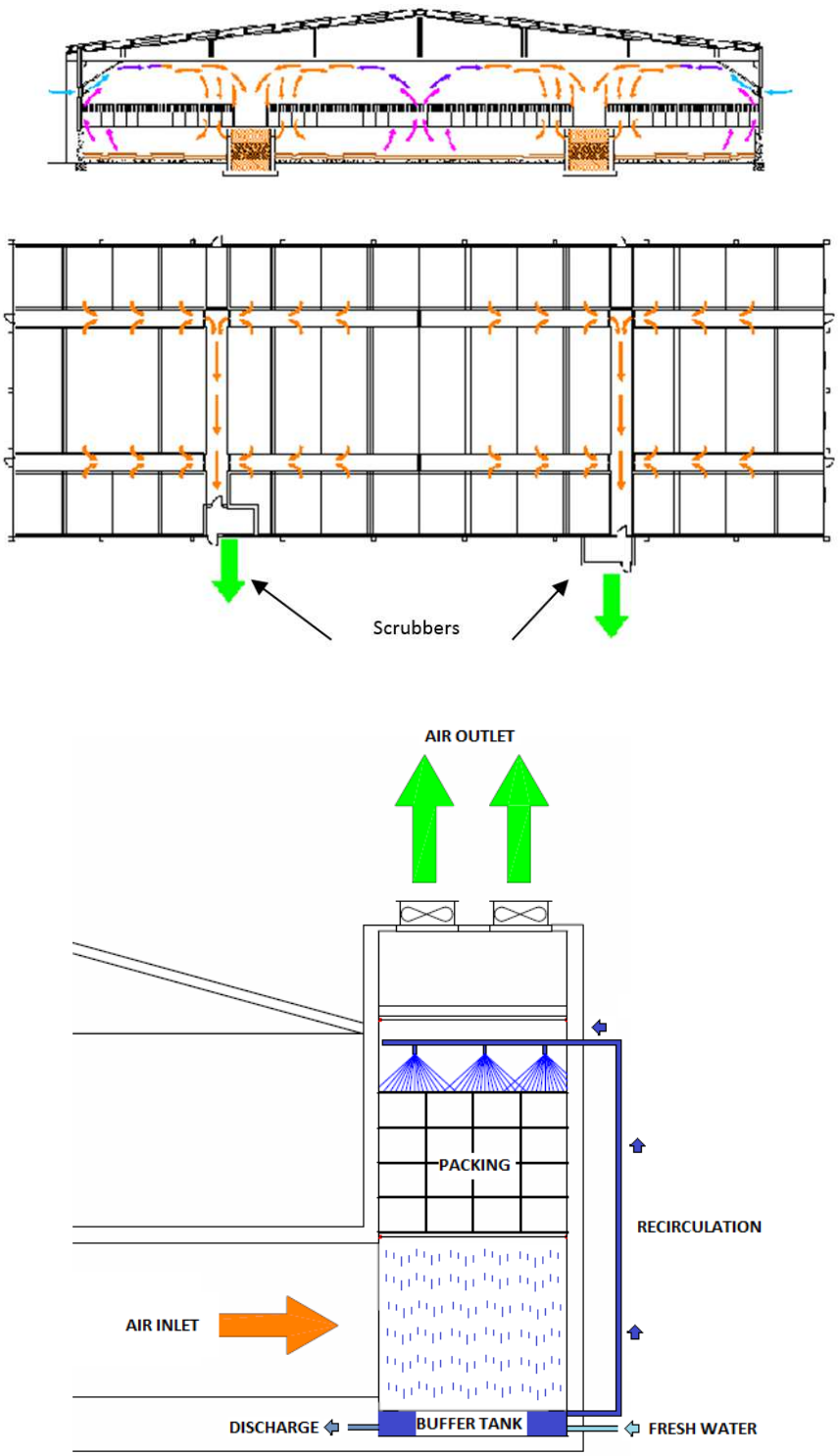

Table 1. Specifications and operation conditions of each biotrickling filter

Specification	Description of value
Packing bed material	Inorganic (Polypropylene)
Packing bed dimensions	2.40 x 4.50 x 1.8 m (wide x long x high)
Packing bed volume	19.44 m^3
Specific surface area	80 m^2m^{-3}
Water tank volume (min-max)	1.39-3.64 m^3
Water recirculation rate	25 m^3 h^{-1}
Empty bed retention time at 100% ventilation	1.04 s (Test 1) – 1.38 s (Test 2)

5.2.3. Experimental design

The research was carried out in two main periods in 2009 (Test 1) and 2012-2013 (Test 2). Different management of biotricking filters were tested regarding water discharge frequency, and therefore different phases were established in each test. These two tests were conducted as summarized in Table 2. In both tests, an identical pre-experimental phase was conducted, since they were previously out of operation. This phase lasted five weeks and included an initial cleaning of filters and water tank, after which the scrubber operated without water replacement for four weeks in order to generate the nitrifying sludge. Finally, water was partially replaced every day during 1 week before the experimental tests started.

Test 1 was conducted for 45 days in from October to November 2009, and aimed to evaluate the effect of water discharge frequency. During this test, emissions from Scrubber 1 were monitored. In phase 1A, Scrubber 1 operated with one partial water discharge per week (renovation of about 64% of the water tank) for 28 days, and then no discharge was done for 17 days (phase 1B). Scrubber 2 was operated with no discharge of biotrickling water during phases 1A and 1B, and water was only added to compensate evaporation.

Test 2 was conducted during several periods between August 2012 and May 2013 and both scrubbers were operated under the same conditions during the whole test (Table 2). After the same pre-experimental phase previously described, scrubbers were operated with two weekly water discharges during 24 days (phase 2A). Then, after five days operated with daily water discharges, phase 2B consisted in only one weekly discharge during 42 days. Finally scrubbers were operated with no water discharge during the 7 following

months (phase 2C), in order to evaluate long-term performance under these management conditions (Table 2).

Table 2. Management of scrubbers 1 and 2 regarding water discharge frequency[1] and measurements conducted (in parenthesis)[2]

Test	Phase	Duration	Operation conditions	
			Scrubber 1	Scrubber 2
Test 1	Phase 1A	28 days (start 2nd October 2009)	1WR (G, W)	NR (W)
	Phase 1B	17 days (start 30th October 2009)	NR (G, W)	NR (W)
Test 2	Phase 2A	24 days (start 7th August 2012)	2WR (G, W)	2WR (G, W)
	Phase 2B	42 days (start 4th September 2012)	1WR (G, W)	1WR (G, W)
	Phase 2C	188 days (start November 2012[3])	NR (G, W)	NR (G, W)

[1] Water management: NR: no renovation; 1WR: 1 renovation per week, 2WR: 2 renovations per week.
[2] Measurements conducted each phase: G-gas concentrations; W: water analysis.
[3] Measurement period during this phase 9-14 May.

5.2.4. Measurements

Ammonia (NH_3), nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2) concentrations were measured continuously and simultaneously by duplicate at five locations: incoming air to the building, exhaust air before and after Scrubber 1, and exhaust air before and after Scrubber 2 (only in Test 2). Gas samples were taken by means of heated Teflon® conductions and analysed in the farm using an infra-red photoacoustic system (1412 Photoacoustic Field Gas Monitor, Innova LumaSense Technologies). This analyser is considered by the VERA test protocol for air cleaning technologies (VERA, 2016). Considering the limitations of the photoacoustic analysers regarding the time response to changes in NH_3 concentrations, repeated measurements were taken at each location and data were considered for analysis after the stabilisation of readings. According to the technical specification of the INNOVA 1412, the detection limit of the measurement is 0.2 ppm for NH_3, 0.4 ppm for CH_4, 0.03 ppm for N_2O and 1.5 ppm for CO_2. The Analyser was calibrated before the start of the trial. Similar to the procedure indicated by Dumont, Hamon et al. (2014), the initial measurements of each measurement point were discarded to account for the response time for NH_3. These authors found no significant differences between photoacoustic measurements and the reference system (acid trapping).

Water characteristics and consumption in each scrubber was monitored weekly to perform a nitrogen balance. Evaporation and discharge water volumes were quantified by

measuring the depth of water in the washing water tank and by mean of water counters. Water samples were collected for further laboratory analyses of ammonia nitrogen (NH_3-N) by direct distillation, nitrates (NO_3^-) and nitrites (NO_2^-) using ion chromatography with conductivity detector, and total Kjeldahl nitrogen. Electric conductivity and pH were also measured every week using Crison equipments CM2202 and GLP 22, respectively.

5.2.5. Scrubber efficiency and related parameters

For descriptive and comparative purposes, emissions from the animals during the tests (without considering the effect of the scrubbers) were calculated by means of an air balance, multiplying the difference of concentrations between the outlet (before the scrubber) and the building inlet fresh air, times the ventilation rate, on an hourly basis. Emissions from each outlet (scrubber) were calculated separately considering the corresponding concentrations and ventilation rates, and then added to obtain the total house emissions. Average emissions for each phase were then calculated and expressed per year and sow using the corresponding number of animals in the farm.

The removal efficiency (RE) was calculated as the percentage of the incoming concentration removed by each scrubber for each gas, according to Equation 1:

$$RE(\%) = [(C_{sc\ inlet} - C_{sc\ outlet}) \div C_{sc\ inlet}] \times 100 \quad (1)$$

Where RE is expressed as percentage and $C_{sc\ inlet}$ and $C_{sc\ outlet}$ are the scrubber inlet and outlet gas concentrations, respectively, expressed in mg m^{-3}.

A Student's t-test was conducted to evaluate whether differences existed between scrubber inlet and outlet concentrations, thus involving a significant absorption of a gas by the scrubber. This analysis was conducted using SPSS software.

A water-air balance was also conducted using water volumes and concentrations of different dissolved N species (NH_3-N, NO_2^- and NO_3^-). This balance was conducted in order to determine the fate of the retained ammonia nitrogen in the scrubbers.

5.2.6. Cost analysis

Costs were determined by comparison with a reference building for pregnant sows with mechanical ventilation using automated wall windows and ceiling exhaust fans

according to EC (2003). A cost analysis of the system was conducted considering real costs of investment and operation costs. Investment costs were calculated assuming a life of 20 years for the civil construction and 10 years for the equipment. Operational costs were estimated from real measurements of energy and water consumption recorded from the corresponding meters installed at the farm, as well as from and real prices for the farmer in 2015. The management of discharge water was accounted for as an equivalent cost of slurry treatment according to local average prices (0.80 Eur m^{-3} for storage and 2.5 Eur m^{-3} for slurry application). Three scenarios (1 and 2 weekly water discharge and no water discharge) were considered. Costs were expressed both per sow place and per kg of NH_3 abated.

5.3. Results

5.3.1. Farm operational parameters

Average air temperature, relative humidity, ventilation rates, gas concentrations and gas emissions are shown in Table 3. Ambient temperature was affected by seasonal and daily variations and ranged between 4.6 and 27.7ºC in Test 1 and from 7.2 to 40.8 ºC in Test 2. Indoor air temperature was more stable, ranging between 20.0 and 29.7ºC. Ambient relative humidity also varied daily and seasonally (from 14% to 97%), whereas indoor relative humidity varied in a narrower range (from 39 to 84%). As ventilation rate was established to maintain appropriate indoor temperature, it was also affected by seasonal variations, and ranged from 22.1 to 129.7 m^3 h^{-1} per sow.

Indoor gas concentrations were on average for the two tests 10.8 mg m^{-3} NH_3, 3108 mg m^{-3} CO_2, 84.3 mg m^{-3} of CH_4 and 1.3 mg m^{-3} of N_2O. Outlet gas concentrations before the scrubber followed an opposite variation pattern to ventilation flow, with maximum concentrations in the early morning and minimum during the afternoon. Gaseous emissions were consistent among the different experimental periods, as shown in Table 3. On average, gaseous emissions throughout the experiment were 6.1 kg NH_3 per sow and year, 1399.3 kg CO_2 per sow and year, 49.5 kg CH_4 per sow and year and 0.29 kg N_2O per sow and year.

Table 3. Building operation conditions of ambient and indoor temperature ($T_{ambient}$ and T_{indoor}, respectively), ambient and indoor relative humidity ($RH_{ambient}$ and RH_{indoor}, respectively), ventilation rate (V), outlet gas concentrations before the scrubbing system (NH_3, CO_2, CH_4, N_2O; Scrubber 1 / Scrubber 2), and house emissions of each gas (E NH_3, E CO_2, E CH_4 and E N_2O). Standard deviations are expressed in parenthesis.

	Test 1		Test 2		
	Phase 1A	**Phase 1B**	**Phase 2A**	**Phase 2B**	**Phase 2C***
$T_{ambient}$ (ºC)	18.0 (4.4)	17.0 (3.8)	25.0 (6.0)	18.9 (5.7)	13.0 (3.8)
T_{indoor} (ºC)	25.5 (1.0)	24.2 (1.1)	27.2 (1.4)	25.7 (1.4)	**
$RH_{ambient}$ (%)	55.0 (15)	66.0 (7.9)	53.9 (18.4)	64.5 (18.2)	69.6 (16.5)
RH_{indoor} (%)	61.3 (8.9)	64.0 (7.7)	66.5 (8.6)	64.2 (8.8)	**
V (m^3/h per sow)	53.5 (5.6)	46.4 (5.0)	78.2 (7.2)	70.7 (5.7)	90.5 (4.1)
NH_3 (mg/m^3)	12.80 / NA (2.94 / NA)	11.75 / NA (1.92 / NA)	7.59 / 11.39 (2.13 / 2.40)	7.73 / 10.81 (2.51 / 3.31)	21.66 / 24.27 (4.35 / 9.81)
CO_2 (mg m^{-3})	3826 / NA (904 / NA)	4281 / NA (811 / NA)	2527 / 2774 (593 / 726)	2822 / 3060 (748 / 687)	4074 / 4125 (1079 / 1194)
CH_4 (mg m^{-3})	100.3 / NA (22.0 / NA)	142.7 / NA (29.1 / NA)	58.41 / 98.70 (16.48 / 23.59)	61.88 / 82.57 (29.05 / 25.13)	146.4 / 96.8 (26.89 / 27.80)
N_2O (mg m^{-3})	0.67 / NA (0.12 / NA)	0.62 / NA (0.07 / NA)	1.41 / 1.48 (0.22 / 0.15)	1.34 / 1.47 (0.20 / 0.20)	1.02 / 1.77 (0.31 / 0.36)
E NH_3 (kg $year^{-1}$ sow^{-1})	5.9 (1.52)	4.7 (0.91)	5.9 (1.03)	5.4 (1.44)	16.6 (4.90)
E CO_2 (kg $year^{-1}$ sow^{-1})	1425 (321)	1425 (274)	1231 (394)	1316 (395)	2439 (915)
E CH_4 (kg $year^{-1}$ sow^{-1})	46 (6.97)	56 (7.66)	50 (10)	43 (14)	90 (18)
E N_2O (kg $year^{-1}$ sow^{-1})	0.11 (0.06)	0.06 (0.03)	0.42 (0.11)	0.37 (0.10)	0.66 (0.22)

* Values reported for 9-14 of May; ** No measured data available; NA: concentration values not measured for Scrubber 2

5.3.2. Removal efficiency in the air balance

Removal efficiency was calculated for each gas from the continuous measurements of gaseous concentrations (see Appendix A, supplementary material), and results are shown in Table 4. The removal efficiency of NH_3 was very variable depending mainly on the scrubber performance, management of washing water and inlet concentration. During Test 1 gas concentrations were only monitored in Scrubber 1 and progressive changes in removal efficiency were found according to washing water management. Before phase 1A, it was observed that scrubber efficiency increased from less than 30% to more than 70% when daily water replacements were conducted after four weeks without replacing the water (see supplementary material). During phase 1A (one weekly water discharge), the highest efficiencies were observed (74% on average), which were reduced during phase 1B, in which no water discharges were conducted. At the end of this phase, removal efficiencies lower than 30% were found again in the scrubber. Nitrous oxide was generated in all phases, leading to negative efficiencies and outlet scrubber concentrations about 74% higher than the inlet concentrations. No significant effect of the scrubber was found for CO_2 or CH_4.

During the second test (2012-2013) the behaviour of both scrubbers was similar for all gases except for NH_3. Also, visual differences between the two scrubbers were appreciated: whereas in Scrubber 2 a biofilm growth was present, this was absent in Scrubber 1 (see Appendix B, supplementary material). Similar to Test 1, no influence on CO_2 and CH_4 emissions was detected, whereas both scrubbers were net generators of N_2O, but in a lower amount than in Test 1. However, the difference between scrubbers 1 and 2 was evidenced according to NH_3 removal efficiencies (significantly higher in Scrubber 2 than in Scrubber 1). The increase of water replacement frequency increased NH_3 removal efficiency in Scrubber 1, whereas no differences were detected for Scrubber 2 because it was high for all phases (higher than 90% on average).

5.3.3. Nitrogen mass balance

The characteristics of the washing water in each test are presented in Table 5. Electric conductivity rose up to values ranging 40 to 70 dS m^{-1} as water evaporated and nitrogen accumulated, and this was coincident with lower NH_3 removal efficiencies. On

the contrary, pH dropped slightly from about 7.5 to 6.8-7 as the N content in the water increased probably due to H^+ liberation during nitrification.

Table 4. Average removal efficiency of each gas during the different phases in Scrubber 1 and Scrubber 2. Standard deviations are given in parenthesis.

Gas	Scrubber	Test 1		Test 2		
		Phase 1A	Phase 1B	Phase 2A	Phase 2B	Phase 2C[1]
NH_3	1	74.2 (8.6)*	54.4 (14.0)*	81.1 (13.3)*	69.2 (11.4)*	17.1 (14.7)*
	2	NA	NA	91.1 (7.5)*	92.0 (6.0)*	90.9 (2.5)*
CO_2	1	0.4 (4.0)	1.0 (4.4)	0.9 (3.7)	-0.1 (3.5)	-1.3 (5.5)
	2	NA	NA	-1.3 (3.7)	-1.3 (2.8)	-0.5 (5.4)
CH_4	1	1.0 (4.9)	1.0 (5.4)	-3.9 (13.8)	0.1 (8.1)	-0.5 (5.9)
	2	NA	NA	-2.1 (9.1)	-0.2 (6.5)	0.9 (5.6)
N_2O	1	-73.8 (31.8)*	-73.0 (27.5)*	-19 (7.1)*	-23.4 (8.2)*	-24.6 (12.4)*
	2	NA	NA	-25.7 (6.4)*	-19.6 (5.7)*	-14.0 (5.7)*

* The removal (positive values) or increased emission (negative values) was statistically significant ($p<0.05$).
NA: Scrubber 2 was not measured during Test 1
[1] Values reported for the period from 9th to 14th of May.

Table 5. Washing water characteristics at the end of each test. Water volume in the tank, ammonium nitrogen (NH_4-N), nitrite (NO_2^-) and nitrate (NO_3^-) concentrations are shown. Electrical conductivity (EC) and pH are also reported.

	Scrubber	Test 1		Test 2		
		Phase 1A	Phase 1B	Phase 2A	Phase 2B	Phase 2C
Water volume in the tank (m^3)	1	1.65	3.40	2.81	2.94	3.03
	2	3.32	3.67	3.02	3.55	3.13
NH_4 (kg m^{-3})	1	9.33	4.79	1.33	2.99	7.45
	2	8.19	7.88	0.45	1.47	0.94
NO_2^- (kg m^{-3})	1	11.98	4.59	0.40	7.59	18.63
	2	3.64	2.99	0.39	3.42	1.03
NO_3^- (kg m^{-3})	1	32.99	18.02	4.65	0.53	1.36
	2	37.03	37.32	1.33	0.60	3.20
EC (dS m^{-1})	1	68.60	37.50	9.56	17.99	41.5
	2	58.40	56.20	3.77	9.19	7.68
pH	1	6.83	6.94	7.03	7.22	7.13
	2	6.84	6.82	7.25	7.15	6.91

Final nitrogen accumulated in the air and washing water for the different phases in this study is shown in Figure 2, more detailed graphs of the evolution of the N balances can be found in the supplementary material (Appendix C). In the first test, Scrubber 1 accumulated a 40% more of nitrogen in the washing water than Scrubber 2. Both ammonium and nitrate were accumulated in similar proportions in both scrubbers, although the amount of nitrogen retained as nitrite was higher in Scrubber 1. In the second test, differences between scrubbers can be observed both using one or two water discharges per week. It can be observed that Scrubber 2, which had developed a biofilm on the packing material surface, was more efficient in capturing the NH_3, but also in transforming to gaseous nitrogen by means of nitrification-denitrification. On the contrary, Scrubber 1, which did not develop any biofilm, was less efficient and accumulated higher amounts of nitrogen species in the washing water. As a consequence, reducing water discharge frequency from 2 to 1 times per week had a detrimental effect on NH_3 removal efficiency in Scrubber 1 (from 81% to 69%) whereas no effect was appreciated on Scrubber 2 (settled around 90%).

In all cases N_2O-N production represents around the 3% of total NH_3-N inlet in the scrubbers.

5.3.4. Estimation of costs

The estimation of costs is detailed in Table 6, and ranged between 29.30 and 32.16 Eur per sow and year depending on scrubber management. Investment costs, both construction and equipment, constituted a major share of total costs for this system. These constituted about 60% of total costs. Operational costs were mainly related to extra energy costs with respect to a conventional system. These energy costs included the energy required for water recirculation and also the extra energy needed by the fans to overcome the extra pressure drop caused by the packing material (about 80-100 Pa at maximum ventilation capacity).

When costs were expressed per kg of NH_3 abated, the efficiency of the scrubber become a major driving force, and these costs ranged from 5.7 Eur kg^{-1} NH_3 abated for the most efficient management (no water discharge in Scrubber 2, which had developed a biofilm) to 9.8 Eur kg^{-1} NH_3 for the most inefficient (no water discharge, in Scrubber 2, which had no visible biofilm).

Fig. 2. Nitrogen balance expressed as final N accumulated in the washing water or emitted to the air during the tests, expressed as a percentage of NH_3-N content in the scrubber inlet air (S1_1A: Scrubber 1, Phase 1A; S1_2A: Scrubber 1, Phase 2A; S1_2B: Scrubber 1, Phase 2B; S2_2A: Scrubber 2, Phase 2A; S2_2B: Scrubber 2, Phase 2B).

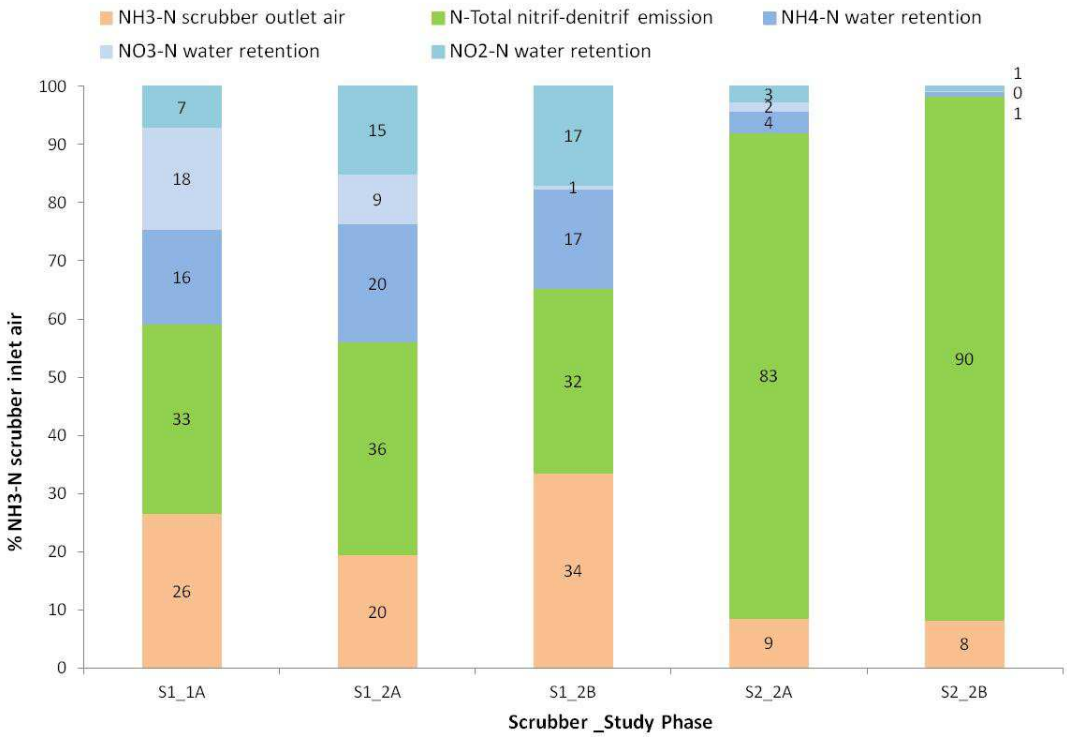

Table 6: Summary of costs (Eur) expressed per sow and year, and per kg NH_3 abated

		Test 1		Test 2		
Water management		**NR**	**1WR**	**NR**	**1WR**	**2WR**
Scrubber		**2**	**1**	**2***	**1**	**1**
Investment	Construction	9.87	9.87	9.87	9.87	9.87
	Equipment	8.03	8.03	8.03	8.03	8.03
	Total investment	17.91	17.91	17.91	17.91	17.91
Operation	Water consumption	0.41	0.40	0.49	0.27	0.48
	Energy consumption	10.53	10.53	12.05	11.81	11.81
	Discharged water management	0.10	0.55	0.00	0.66	1.56
	Maintenance	0.40	0.40	0.40	0.40	0.40
	Total operation	11.44	11.88	12.94	13.14	14.25
Total costs per sow and year		29.34	29.79	30.85	31.04	32.16
Total cost per kg NH_3 abated		9.78	6.62	5.71	7.39	6.70

NR: no renovation of water; 1WR: one water renovation per week; 2WR: two water renovations per week
*Visible and efficient nitrifying biofilm

5.4. Discussion

This study provides information on the performance and costs of a bioscrubbing system in a house for pregnant sows under warm conditions. The results obtained evidence higher environmental temperatures than those reported in other studies of similar characteristics in Europe. In general, measured gaseous emissions were within the range of those reported in the literature (Groot Koerkamp et al., 1998). The warmer conditions reflected in the amount of ventilation and thus on gas concentrations: ventilation rates were in general terms higher than those found in the literature (Seedorf et al., 1998), whereas gas concentrations were in the lower range of those previously reported in colder conditions (Groot Koerkamp et al., 1998; Van der Heyden et al., 2015). The scrubber system, however, was dimensioned accordingly, and the empty bed retention time was also within the range of previously reported values (Van der Heyden et al., 2015). This is of critical importance, since scrubber efficiency is affected by the empty body retention time (Dumont, Hamon, et al., 2014; Melse et al., 2012), and therefore higher packing volumes are required for higher ventilation rates to achieve the same efficiency.

Ammonia outlet gas concentrations before the scrubber followed an opposite variation pattern to ventilation flow. As temperature and ventilation rates rise, the efficiency of the scrubber retaining NH_3 decreases due to lower contact time between air and liquid phase. These daily and seasonal patterns in scrubbers' removal performance have been also described by Melse et al. (2012) and Van der Heyden, Brusselman, Volcke and Demeyer (2016). However, the different removal efficiency of NH_3 between scrubbers and the influence of water discharge frequency demonstrates the relevance of chemical and biological reactions acting in this process.

As reviewed by Van der Heyden et al. (2015), biotricking filters convert the ammonia in the air to nitrite and then nitrate by means of nitrifier microorganisms, which must be present in immobilised form in a biofilm on the packing material surface. However, N contents must be monitored to prevent the inhibition of ammonia oxidizing bacteria (AOB) such as *Nitrosomonas sp.* and nitrite oxidizing bacteria (NOB) such as *Nitrobacter sp.* These bacterial communities must be installed in the biotrickling filter in order to reduce the excess of dissolved free ammonia and nitric oxide, which are reported to inhibit NOB at concentrations as low as 0.1-1 mg L^{-1} and 0.2-2.8 mg L^{-1},

respectively (Anthonisen, Srinath, Loehr, Prakasam, 1976). As AOB are inhibited at much higher concentrations (e.g. 10-150 mg L^{-1} of free NH_3), nitrite accumulation is characteristic of inhibited nitrification, thus leading to an improper biotricking filter operation. As reported by Melse et al. (2012), conductivity is also an excellent indicator of accumulation of nitrogen species and therefore can be used for monitoring in practice. In this study, inhibition of NH_3 removal by the scrubbers was detected with conductivity about 40 mS/cm and N content of 10 kg N/m^3, measured at maximum water tank level. pH also plays an essential role for inhibitions, since they are mainly affected by free NH_3 and HNO_2. Most frequent among scrubbers may be the inhibition by HNO_2, since nitrification processes tend to lower pH, which also enhances NH_3 removal (Jun & Wenfeng, 2009). In this study, although Scrubber 1 did not show low pH values, it had an important NO_2^- accumulation, particularly at low or no washing water replacement. This was interpreted as a signal of nitrification inhibition. Water recirculation rate may also be of importance. On the one hand, excessive recirculation flow can cause biofilm erosion and sloughing (Van der Heyden et al., 2015). On the other hand, water recirculation must ensure oxygen supply, since nitrification is an aerobic process which needs at least 1-2% of dissolved oxygen (Béline, Martinez, Chadwick, Guiziou, Coste, 1999).

As evidenced in this study and also in previous works on biotrickling filters (Chou & Wang, 2007; Dumont, Lagadec, et al., 2014; Maia, Day, Gates, Taraba, Coyne, 2012; Melse et al., 2012) N_2O is also formed in the biological processes occurring in biotrickling filters. The nitritation, this is, the conversion of ammonium to NO_2^-, is a potential source of N_2O, particularly as a by-product of the intermediate reduction reaction of hydroxylamine to NO_2^- by the enzyme hydroxylamine reductase (Butterbach-Bahl, Baggs, Dannenmann, Kiese, Zechmeister-Boltenstern, 2013). However, denitrification processes occurring at anaerobic conditions, typically at dissolved O_2 concentration lower than 1 mg/L, (Béline et al., 1999) also may release N_2O as an intermediate product. This pathway has also been widely described in wastewater or slurry aerobic reactors as potentially emitting relevant amounts of N_2O (Béline et al., 1999; Cheng & Liu, 2001; Jun & Wenfeng, 2009). The anaerobic conditions required for denitrification and thus for enhanced N_2O generation may be produced both in anaerobic sectors of the packing material and the water tank. These N_2O emissions may be enhanced at high NO_3^- concentrations (Wrage, Velthof, van Beusichem, Oenema, 2001).

Although not intentioned, a different development of biofilm was observed during Test 2 (see supplementary material), which is almost certainly the cause of the observed differences between the scrubbers. As a consequence, the origin of the N_2O in the two scrubbers may also correspond to different origins. In Scrubber 1, nitrogen was mostly accumulated in the washing water, and therefore denitrification was expected to occur at a low rate. For this scrubber, removal of washing water was the main source of eliminating the excess nitrogen, and N_2O would probably be associated mostly to the nitrification process. On the contrary, in Scrubber 2 the different nitrogen species were kept at low concentrations, thus suggesting an important denitrification activity. Therefore, both nitrification and denitrification would be expected to occur in Scrubber 2.

The concentrations of CO_2 and CH_4 were not affected by the scrubber, as reported by previous research (Van der Heyden et al., 2015). Recent research focuses on developing effective scrubbers also to remove pollutant gases other tan NH_3. For N_2O, a reduction of concentration was obtained using a denitrifying off-gas bioscrubber using methanol as electric donor (Frutos et al., 2015; Frutos, Quijano, Pérez, Muñoz, 2016). On the other hand, the potential removal of CH_4 has also been analyzed (Estrada et al., 2014; Girard, Ramirez, Buelna, Heitz, 2011; Lebrero, Hernández, Pérez, Estrada, Muñoz, 2015). These authors identified limitations for the removal of these gases, such as the low gas solubility. Consequently, effective abatement of these gases can only be obtained at the moment with high initial concentrations and high empty bed retention times. For this reason, further developments are essential in order to obtain practical abatement scrubbers for these gases.

Assuming that biofilm development is the cause of different efficiencies among scrubbers, it will also affect the operational cost of scrubbers and the optimal water management. Where there was no visible biofilm, one programmed water renovation per week seemed to be advisable in the case of the scrubber studied. Thus, if a healthy bacterial community is well established in the filters, no renovation seems to be required. In any case, electrical conductivity evolution must be used as an indicator for water discharge control. In this sense, many different inhibition thresholds have been reported in the literature suggested a maximum value of 20 dS/m (Van der Heyden et al., 2015).

Regarding to costs, large bacterial community relies on water to stay stable, thus increasing water consumption. Moreover biofilm growth increases the pressure drop in the filters and the energy required by the ventilation fans. However, these costs seem to be compensated by the lower amount of water discharged and higher NH_3 removal efficiency, thus avoiding increased costs of slurry management.

As the washing water is discharged into the slurry pit, the amount of N in the slurry and the risk of N volatilization increases if any further measures are implemented in downstream farming activities. Considering that most of the N in the water is in oxidised forms (nitrates and nitrites), losses via denitrification in slurry pits under anaerobic condition can be significant. Therefore, to ensure effective emission abatement, a combination of mitigation techniques in different farming activities will be needed.

5.5. Conclusions

The management of washing water affected the NH_3 removal efficiency of two bioscrubbers installed in a commercial sow building. The two bioscrubbers had also different performance as a consequence of the different implantation of biofilm. The efficiency was higher for the biofilter where the biofilm was present (over 90% in all phases), whereas it was lower for the scrubber without visible biofilm ranged from 17% to 81% depending on water discharge frequency.

Nitrous oxide was generated in by the scrubbers, increasing from 14% to 74% the initial concentration. Different implantations of biofilm were found between the scrubbers, although management of scrubbers was similar. Biofilm implantation affected the nitrogen balances and probably the pathways for N_2O formation. The concentrations of CO_2 and CH_4 were not affected by the scrubbing system.

Costs per sow and year of the scrubbing system ranged from 29.34 to 32.16 Eur. The major share of these costs was associated to the installation and energy costs, representing about the 60% and 40%, respectively of total costs. When expressed per kg of NH_3 abated, costs varied largely depending on scrubber performance (from 5.71 to 9.78 Eur per kg of NH_3 abated).

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The authors would like to thanks the support of SAT URRA farmers and INTIA technicians.

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5.7. References

Anthonisen A.C., Srinath E.G., Loehr R.C. and Prakasam T.B.S. (1976). Inhibition of nitrification and nitrous acid compounds. Journal of the Water Pollution Control Federation 48, 835-852.

Backes A.M., Aulinger A., Bieser J., Matthias V. and Quante M. (2016). Ammonia emissions in Europe, part II: How ammonia emission abatement strategies affect secondary aerosols. Atmospheric Environment 126, 153-161.

Behera S.N., Sharma M., Aneja V.P. and Balasubramanian R. (2013). Ammonia in the atmosphere: a review on emission sources, atmospheric chemistry and deposition on terrestrial bodies. Environmental Science and Pollution Research 20, 8092-8131.

Béline F., Martinez J., Chadwick D., Guiziou F. and Coste C.M. (1999). Factors affecting Nitrogen Transformations and Related Nitrous Oxide Emissions from Aerobically Treated Piggery Slurry. Journal of Agricultural Engineering Research 73, 235-243.

Butterbach-Bahl K., Baggs E.M., Dannenmann M., Kiese R. and Zechmeister-Boltenstern S. (2013). Nitrous oxide emissions from soils: how well do we understand the processes and their controls? Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences 368 (1621), 20130122.

Calvet S., Cambra-Lopez M., Blanes-Vidal V., Estellés F. and Torres A.G. (2010). Ventilation rates in mechanically ventilated commercial poultry buildings in Southern

Europe: Measurement system development and uncertainty analysis. Biosystems Engineering 106, 423-432.

Cambra-López M., Aarnink A.J.A., Zhao Y., Calvet S. and Torres A.G. (2010). Airborne particulate matter from livestock production systems: A review of an air pollution problem. Environmental Pollution 158(1), 1-17.

Cheng J. and Liu B. (2001). Nitrification/Denitrification in Intermittent Aeration Process for Swine Wastewater Treatment. Journal of Environmental Engineering 127 (8), 705-711.

Chou M.S. and Wang C.H. (2007). Treatment of Ammonia in Air Stream by Biotrickling Filter. Aerosol and Air Quality Research 7, 17-32.

De Vries J.W. and Melse R.W. (2014). Comparing the environmental impact of a nitrifiying biotrickling filter with or without denitrification for ammonia abatement at animal houses. Wageningen UR Livestock Research Report 717. Wageningen, 25 pp.

Dumont E., Hamon L., Lagadec S., Landrain P., Landrain B. and Andres Y. (2014). NH_3 biofiltration of piggery air. Journal of Environmental Management 140, 26-32.

Dumont E., Lagadec S., Landrain P., Landrain B. and Andrès Y. (2014). N_2O generation resulting from piggery air biofiltration. Chemical Engineering Journal, 248, 337-341.

EC (2003). Reference Document on Best Available Techniques for intensive Rearing of Poultry and pigs. Integrated Pollution Prevention and Control (IPPC), July 2003, 341 pp.

EEA (2016). European Union emission inventory report 1990–2014 under the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP). Technical report No 16/2016, 122 pp.

Estellés F., Melse R. W., Ogink N.W.M. and Calvet. S. (2011). Evaluation of the NH_3 removal efficiency of an acid packed bed scrubber using two methods: A case study in a pig facility. Transactios of the ASABE, 54, 1905-1912.

Estrada J.M., Lebrero R., Quijano G., Perez R., Figueroa-Gonzalez I., Garcia-Encina P. A., and Muñoz R. (2014). Methane abatement in a gas-recycling biotrickling filter: Evaluating innovative operational strategies to overcome mass transfer limitations. Chemical Engineering Journal 253, 385-393.

FAOSTAT (2016). Food and Agriculture Organization of the United Nations. Statistical Division. Retrieved June 12, 2016, from http://faostat3.fao.org/home/E

Frutos O.D., Arvelo I.A., Pérez R., Quijano G. and Muñoz R. (2015). Continuous nitrous oxide abatement in a novel denitrifying off-gas bioscrubber. Applied Microbiology and Biotechnology 99(8), 3695-3706.

Frutos O.D., Quijano G., Pérez R. and Muñoz R. (2016). Simultaneous biological nitrous oxide abatement and wastewater treatment in a denitrifying off-gas bioscrubber. Chemical Engineering Journal 288, 28-37.

Girard M., Ramirez A.A., Buelna G. and Heitz M. (2011). Biofiltration of methane at low concentrations representative of the piggery industry-Influence of the methane and nitrogen concentrations. Chemical Engineering Journal 168, 151-158.

Groot Koerkamp P.W.G., Metz J.H.M., Uenk G.H., Phillips V.R., Holden M.R., Sneath R.W., Short J.L., White R.P.P., Hartung J., Seedorf J., Schröder M., Linkert K.H., Pedersen S., Takai H., Johnsen J.O. and Wathes C.M. (1998). Concentrations and Emissions of Ammonia in Livestock Buildings in Northern Europe. Journal of Agriculture Engineering Research 70, 79-95.

EC (2015). Best Available Techniques (BAT) Reference Document for the Intensive Rearing of Poultry or Pigs. Final Draft - August 2015, European Commision, 911 pp.

Jun Y. and Wenfeng X. (2009). Ammonia biofiltration and community analysis of ammonia-oxidizing bacteria in biofilters. Bioresource Technology 100(17), 3869-3876.

Lebrero R., Hernández L., Pérez R., Estrada J.M. and Muñoz R. (2015). Two-liquid phase partitioning biotrickling filters for methane abatement: Exploring the potential of hydrophobic methanotrophs. Journal of Environmental Management 151, 124-131.

Maia G.D.N., Day V.G.B., Gates R.S., Taraba J.L. and Coyne M.S. (2012). Moisture effects on greenhouse gases generation in nitrifying gas-phase compost biofilters. Water Research 46, 3023-3031.

Melse R. W. and Mosquera J. (2014). Nitrous oxide (N_2O) emissions from biotrickling filters used for ammonia removal at livestock facilities. Water Science and Technology 69, 994-1003.

Melse R.W., Ogink N.W.M. and Rulkens W.H. (2009a). Air Treatment Techniques for Abatement of Emissions from Intensive Livestock Production. The Open Agriculture Journal 3, 6-12.

Melse R.W., Ogink N.W. M. and Rulkens W.H. (2009b). 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, 289-298.

Melse R.W., Ploegaert J.P.M. and Ogink N.W.M. (2012). Biotrickling filter for the treatment of exhaust air from a pig rearing building: Ammonia removal performance and its fluctuations. Biosystems Engineering 113 (3), 242-252.

Philippe F.X., Cabaraux J.F. and Badouin N. (2011). Ammonia emissions from pig houses: Influencing factors and mitigation techniques. Agriculture, Ecosystems & Environment 141 (3-4), 245-260.

Seedorf J., Hartung J., Schröder M., Linkert K.H., Pedersen S., Takai H., Johnsen J.O., Metz J.H.M., Groot Koerkamp P.W.G., Uenk G.H., Phillips V.R., Holden M.R., Sneath R.W., Short J.L.L., White R.P. and Wathes, C.M. (1998). A Survey of Ventilation Rates in Livestock Buildings in Northern Europe. Journal of Agricultural Engineering Research 70, 39-47.

Van der Heyden C., Demeyer P. and Volcke E.I.P. (2015). Mitigating emissions from pig and poultry housing facilities through air scrubbers and biofilters: State-of-the-art and perspectives. Biosystems Engineering 134, 74-93.

Van der Heyden C., Brusselman E., Volcke E.I.P. and Demeyer P. (2016). Continuous measurements of ammonia, nitrous oxide and methane from air scrubbers at pig housing facilities. Journal of Environmental Management 181, 163-171.

VERA. (2016). Test Protocol for Air Cleaning Technologies. Verification of environmental technologies for agricultural production. Retrieved June 12, 2016, from http://www.vera-verification.eu/en/technology-manufacturers/test-protocols/

Wrage N., Velthof G.L., van Beusichem M.L. and Oenema O. (2001). Role of nitrifier denitrification in the production of nitrous oxide. Soil Biology and Biochemistry 33, 1723-1732.

APPENDIX A: Gaseous concentrations

Water management: NR: no renovation; 1WR: 1 renovation per week, 2WR: 2 renovations per week; FR: frequent removal, more than 2 renovations per week.

Fig. A.1. Ammonia concentration before and after the scrubber. Test 1 (2009).

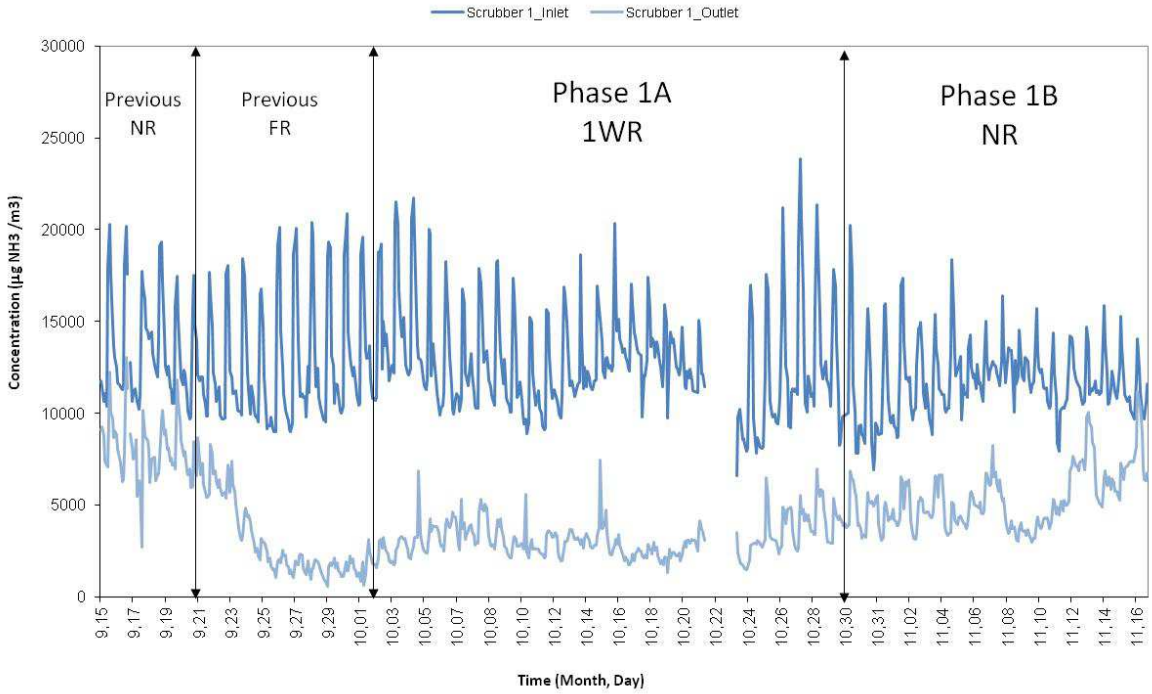


Fig. A.2. Nitrous oxide concentration before and after the scrubber. Test 1 (2009).

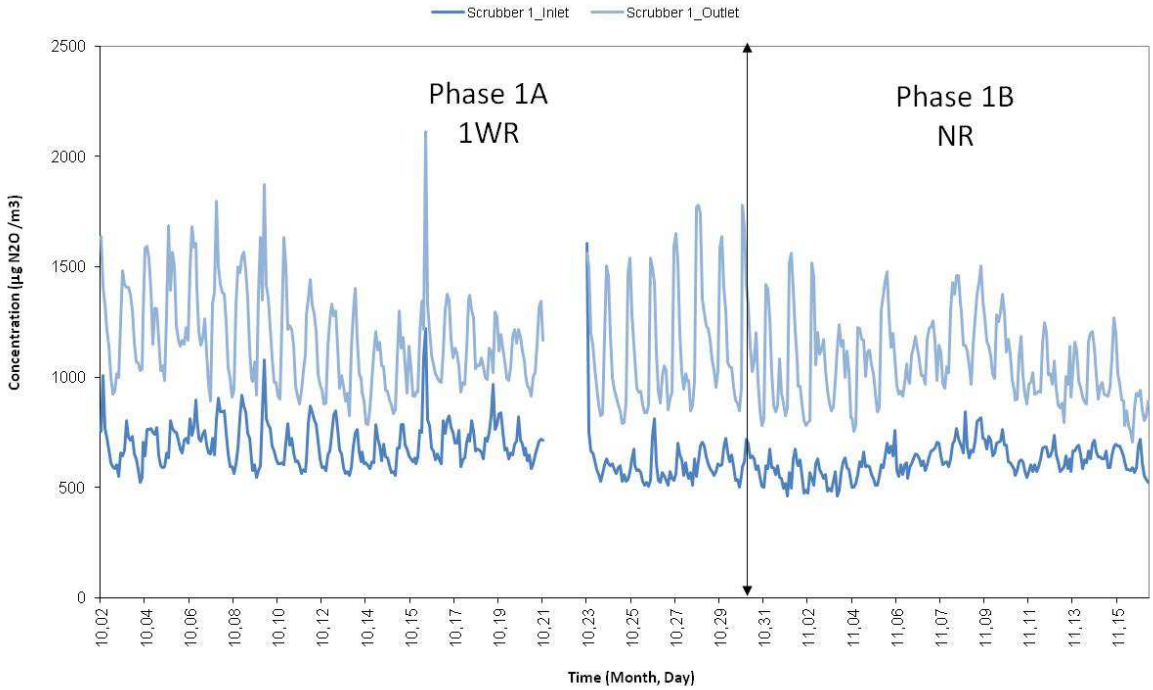


Fig. A.3. Methane concentration before and after the scrubber. Test 1 (2009).

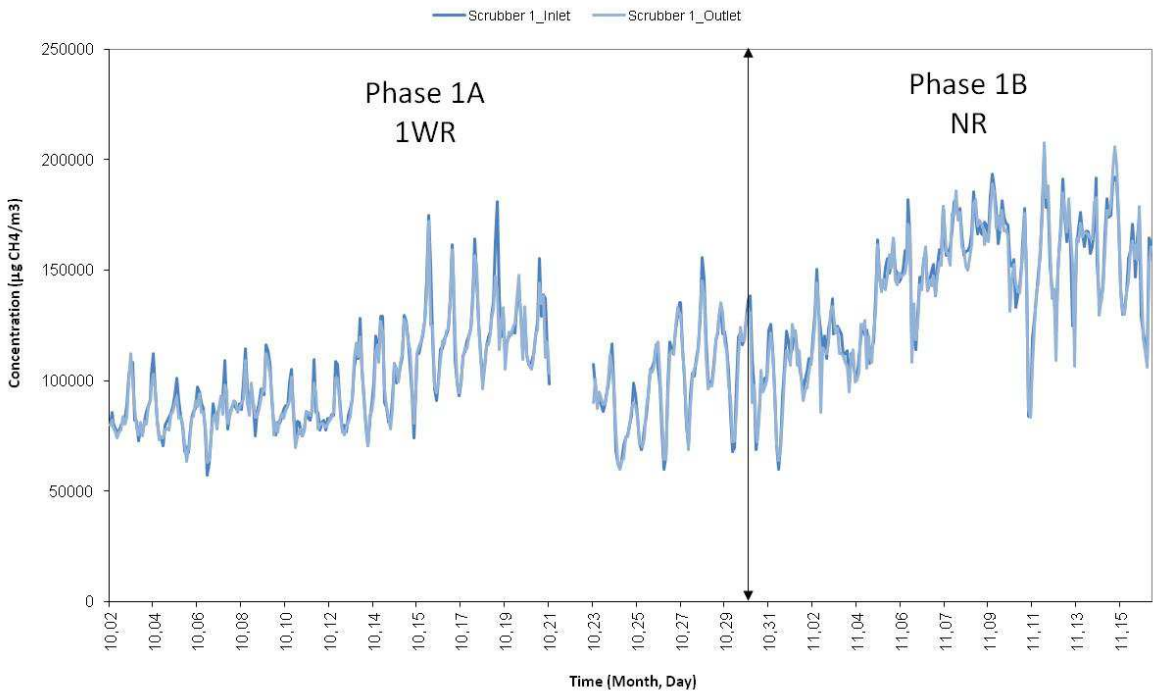


Fig. A.4. Carbon dioxide concentration before and after the scrubber. Test 1 (2009).

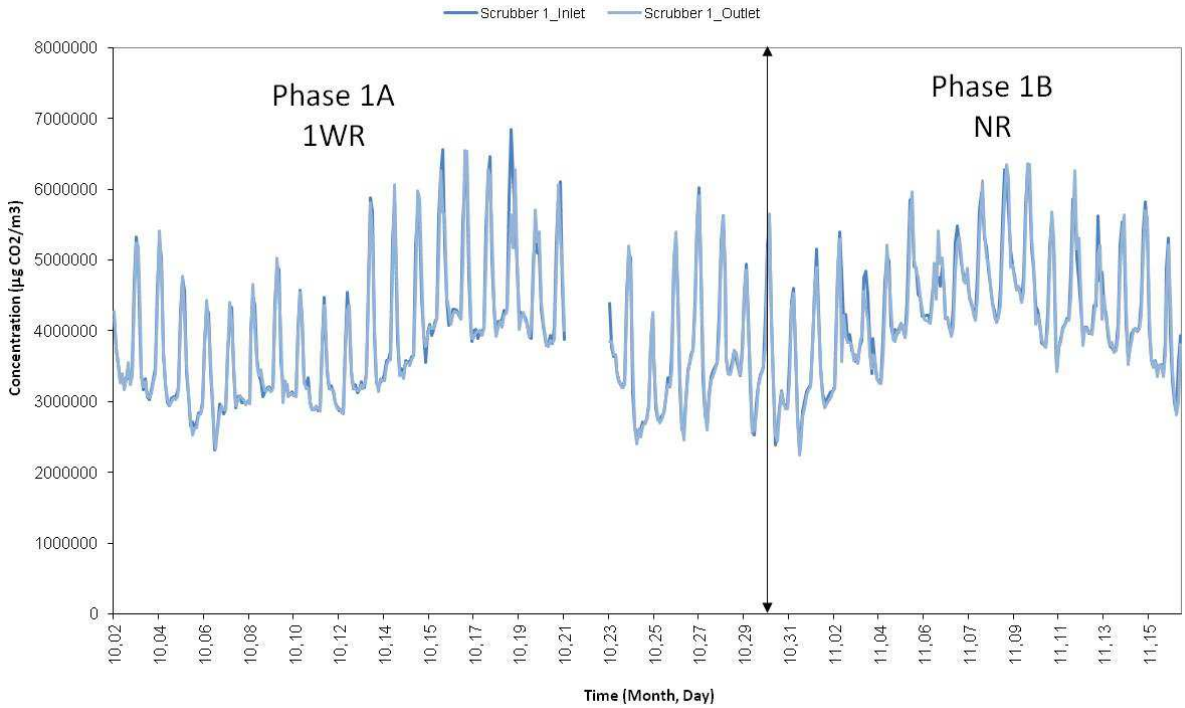


Fig. A.5. Ammonia concentration before and after the scrubber. Test 2 (2012).

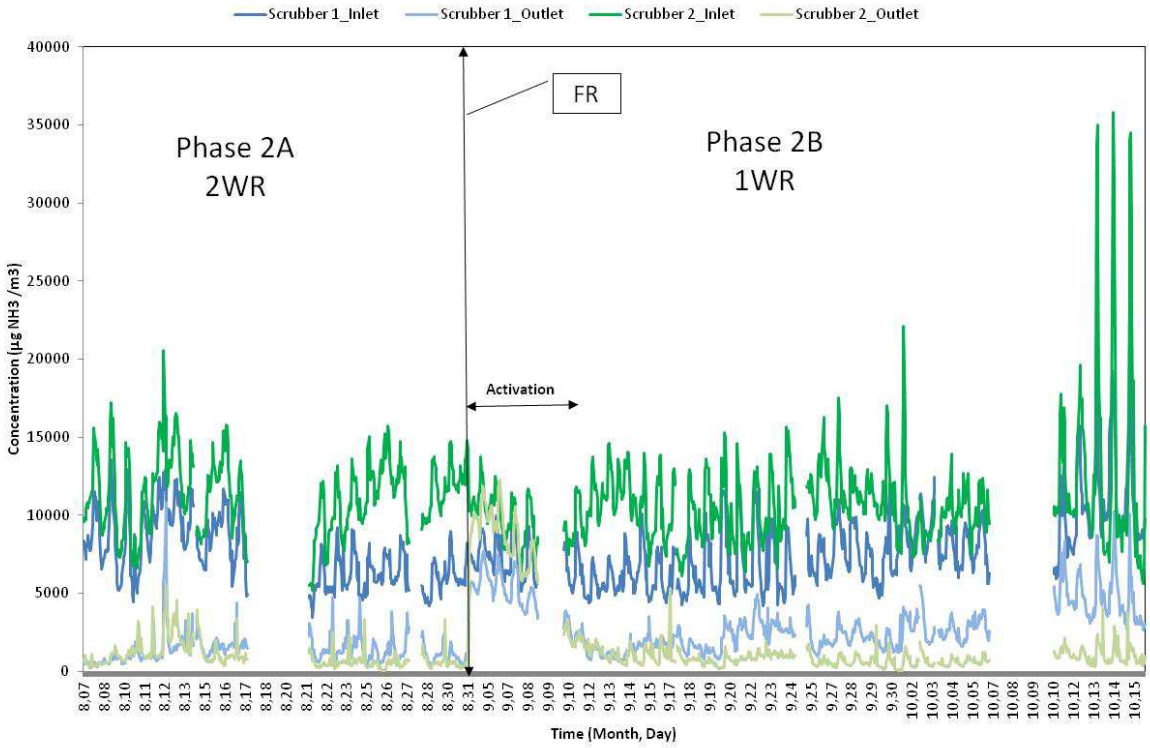


Fig. A.6. Ammonia concentration before and after the scrubber. Test 2 (2013).

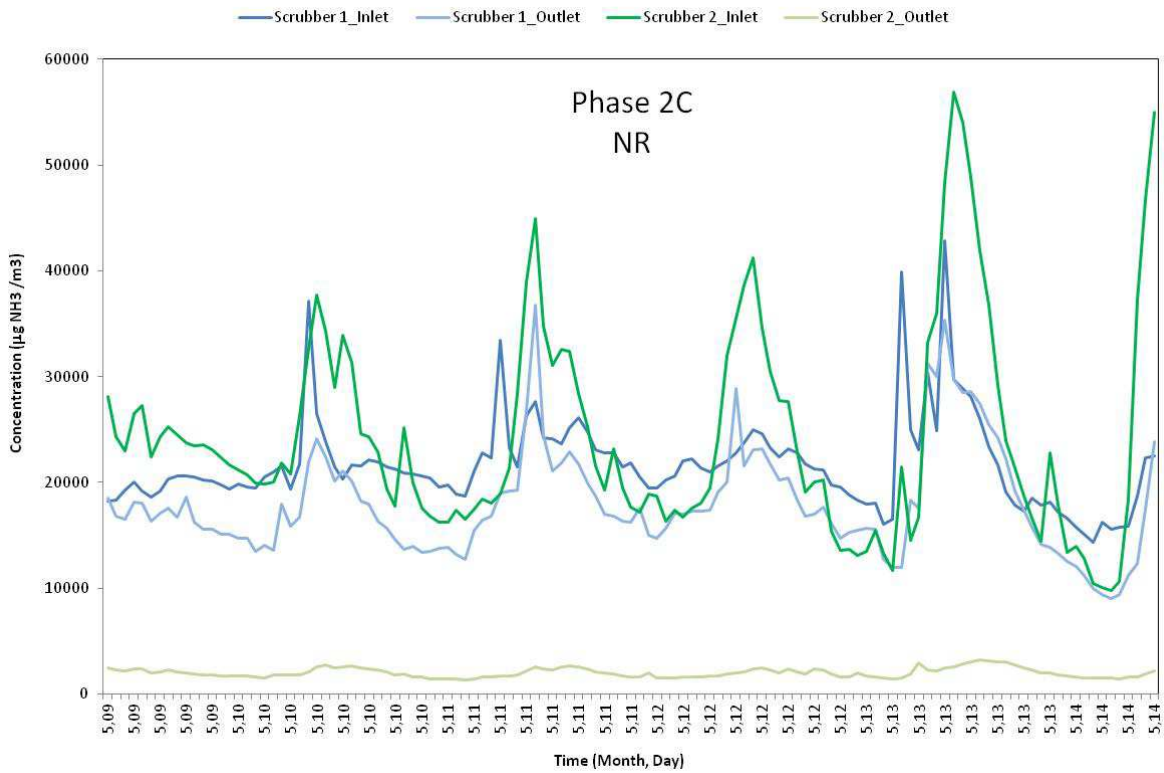


Fig. A.7. Nitrous oxide concentration before and after the scrubber. Test 2 (2012).

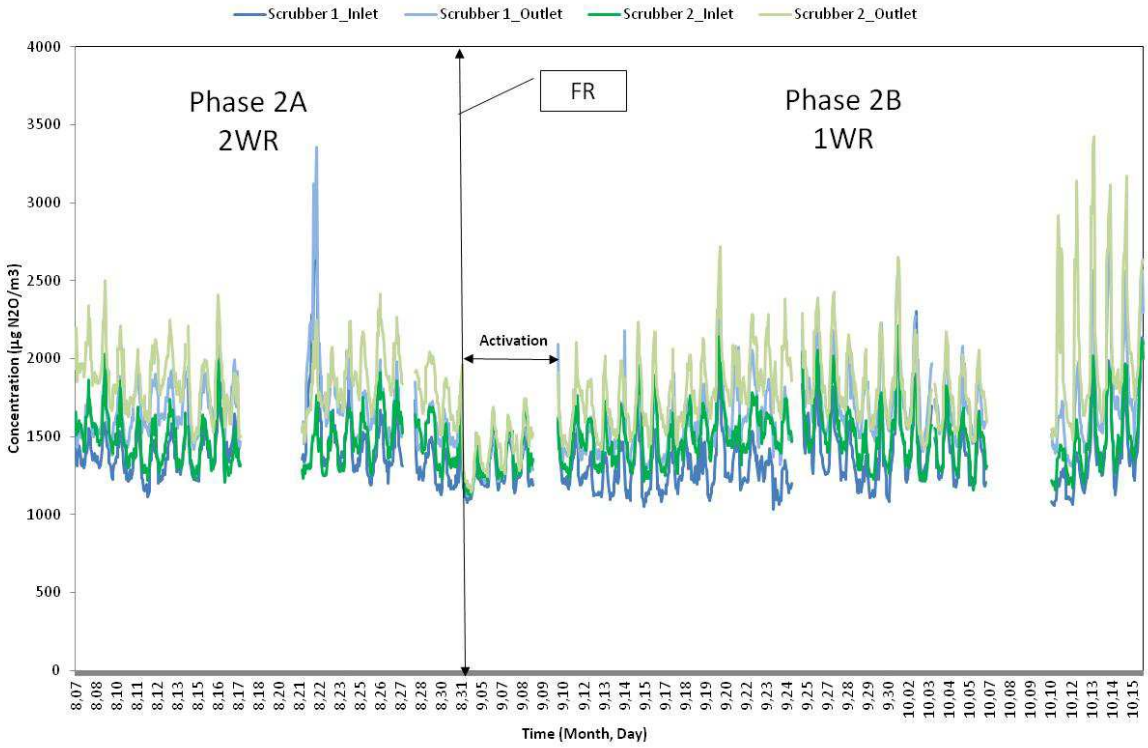


Fig. A.8. Nitrous oxide concentration before and after the scrubber. Test 2 (2013).

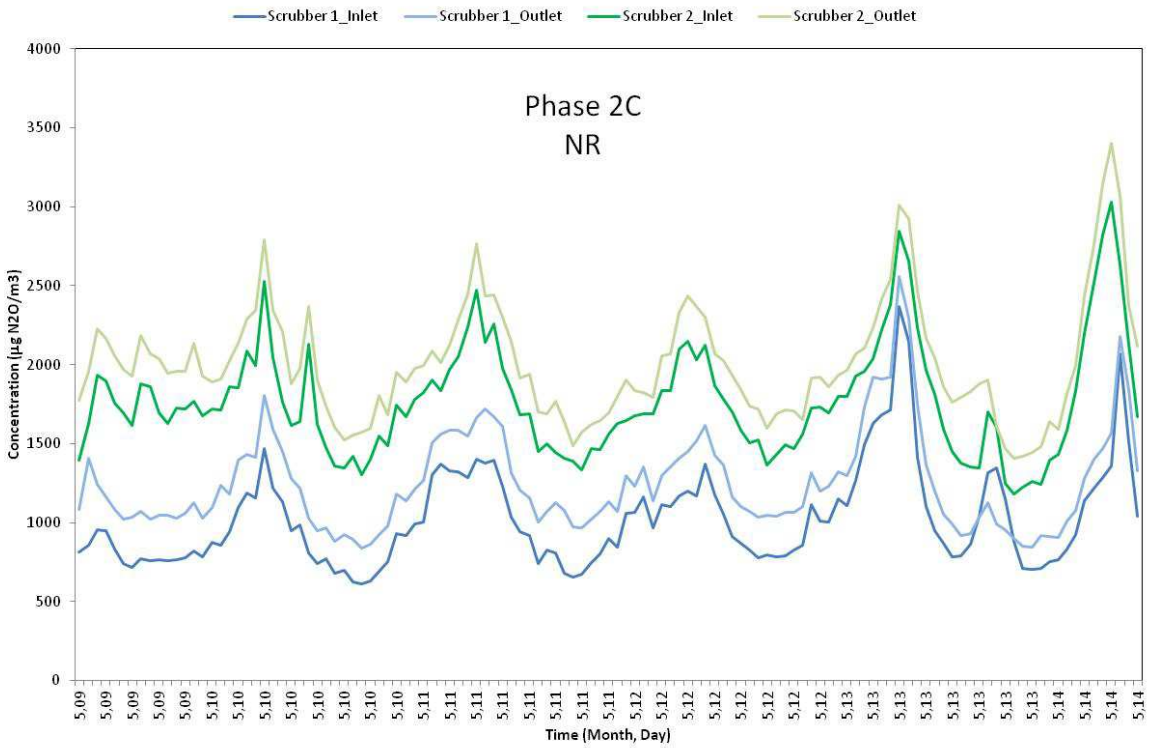


Fig. A.9. Methane concentration before and after the scrubber. Test 2 (2012).

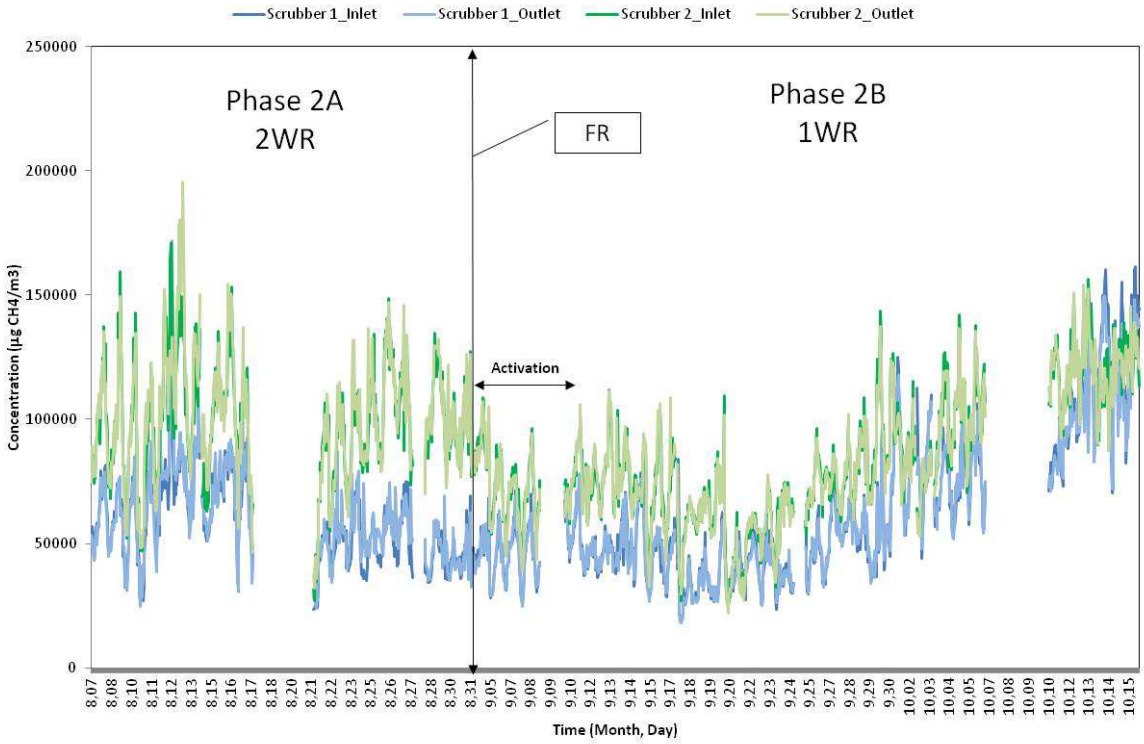


Fig. A.10. Methane concentration before and after the scrubber. Test 2 (2013).

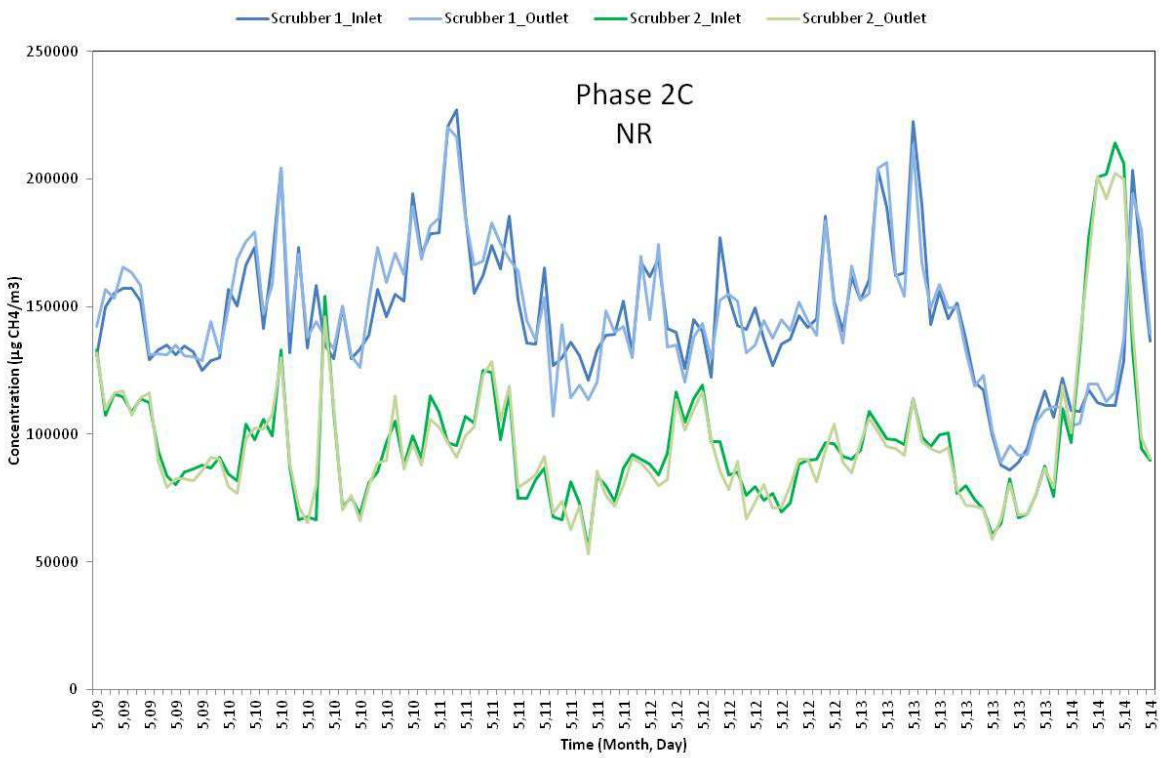


Fig. A.11. Carbon dioxide concentration before and after the scrubber. Test 2 (2012).

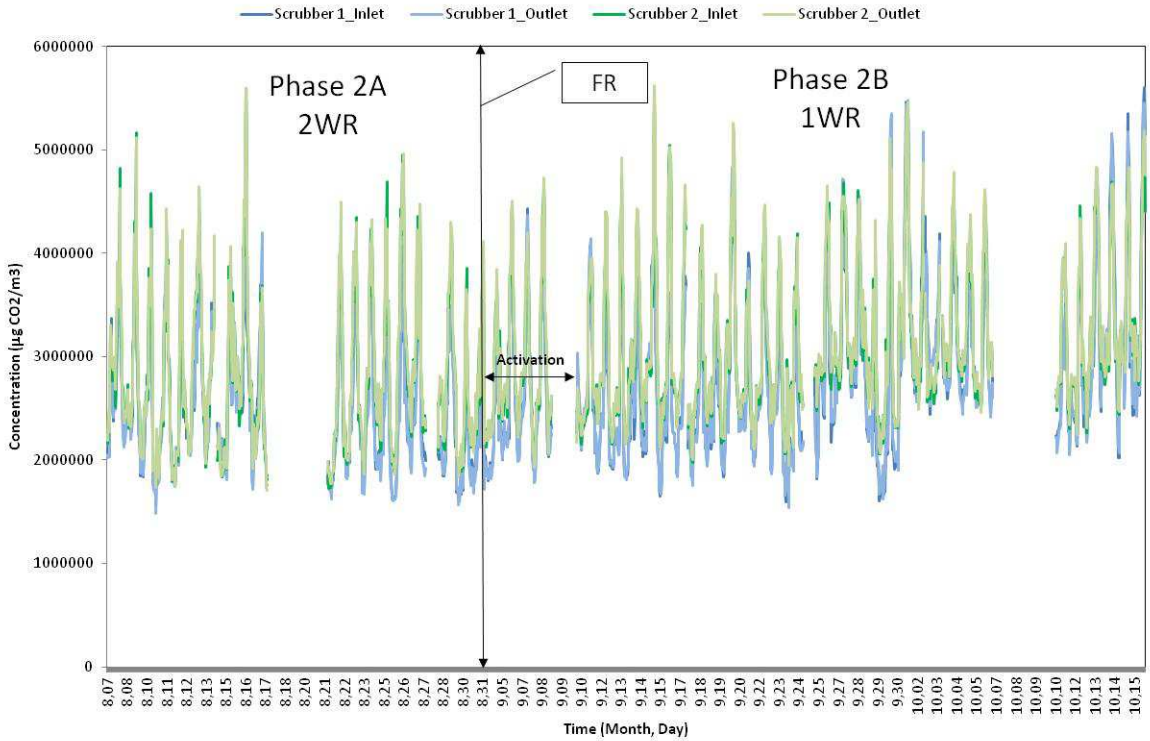
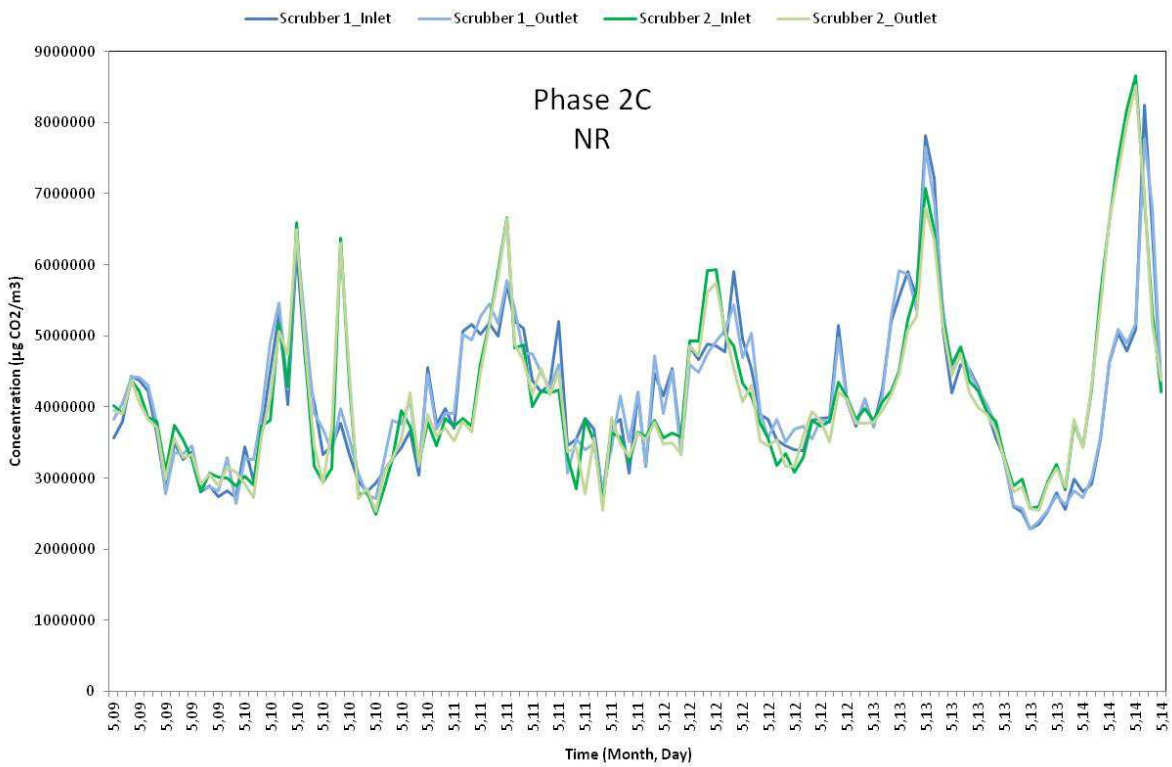


Fig. A.12. Carbon dioxide concentration before and after the scrubber. Test 2 (2013).



APPENDIX B: Scrubbers' Biofilm (photos and microbiology analysis)

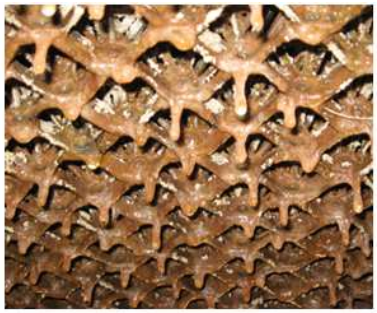
PHOTOS

Test: 2

Study phase: 2C

Date: 14th May 2013

Scrubber	1	2
Water management	124 days NR	188 days NR
NH_3 scrubber retention efficiency	17±15%	91±2%
Water conductivity	41.5 mS cm^{-1}	8 mS cm^{-1}
Water consumption (evaporation)	0.40 m^3 day^{-1}	1.61 m^3 day^{-1}



FILTERS SCRUBBER 2

(Visible biofilm)

FILTERS SCRUBBER 1

(No visible biofilm)

MICROBIOLOGY ANALYSIS

Test: 2

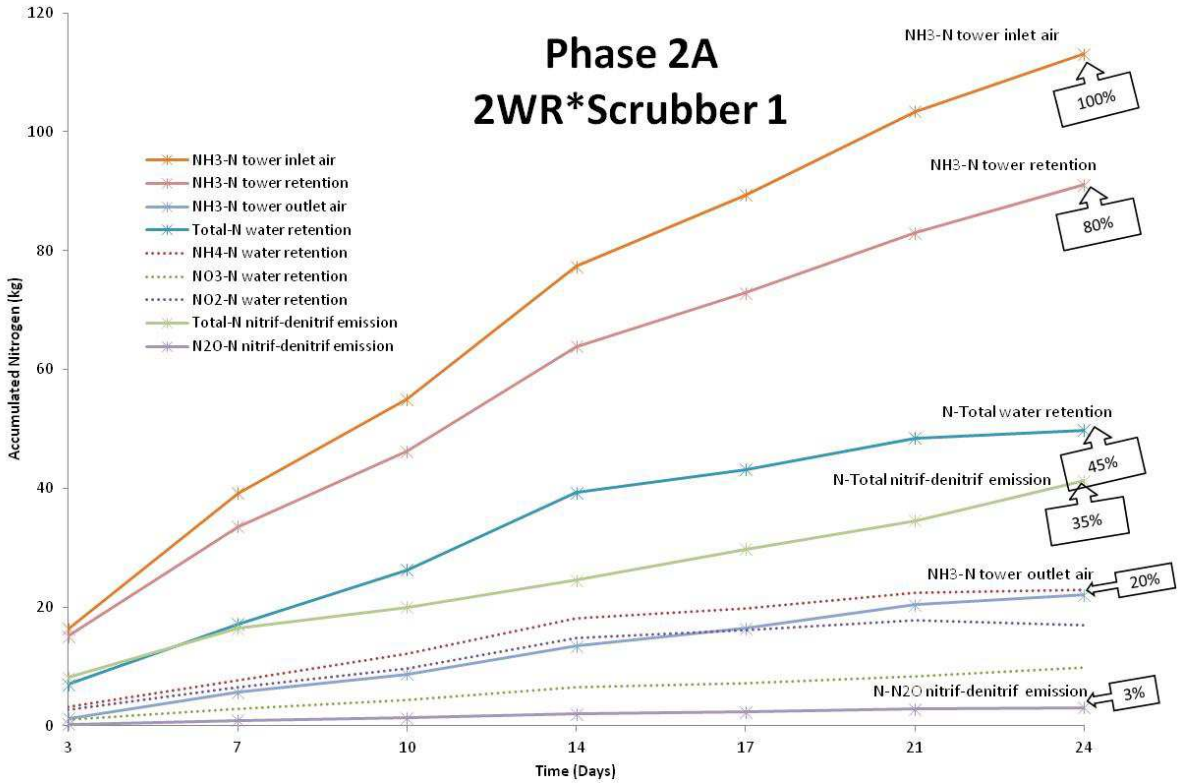
Study phase: 2C

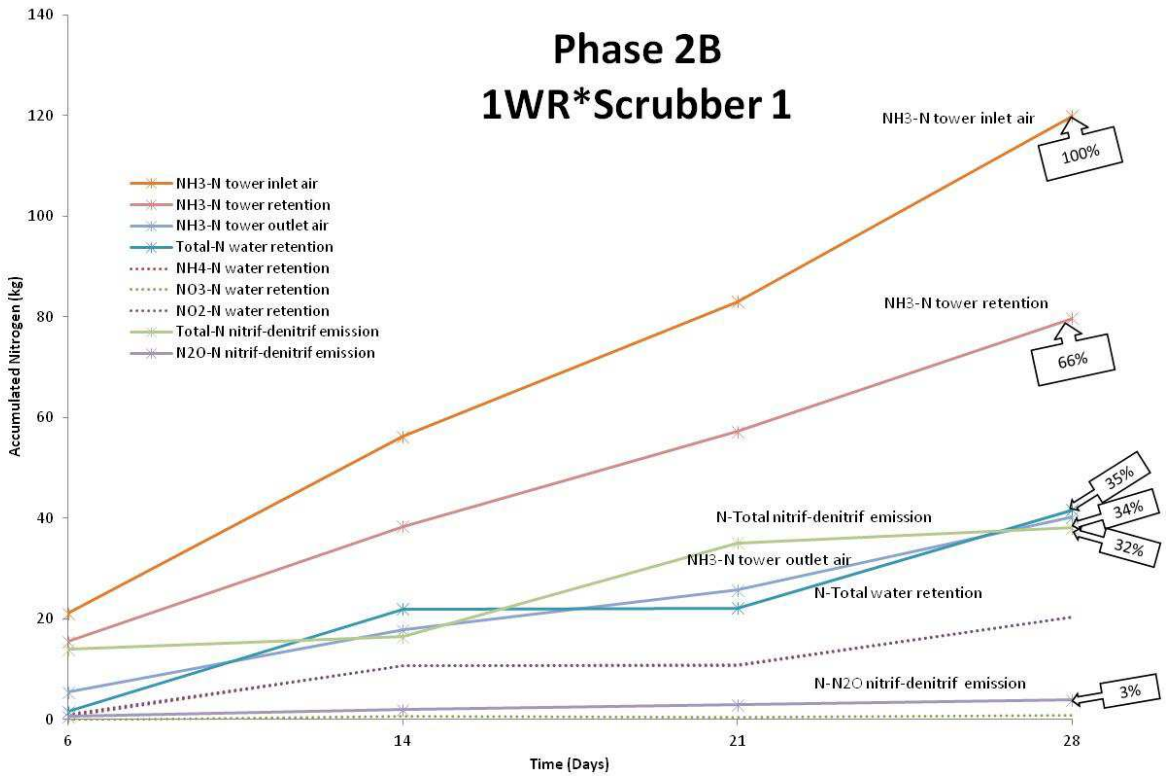
Date: 20th November 2012

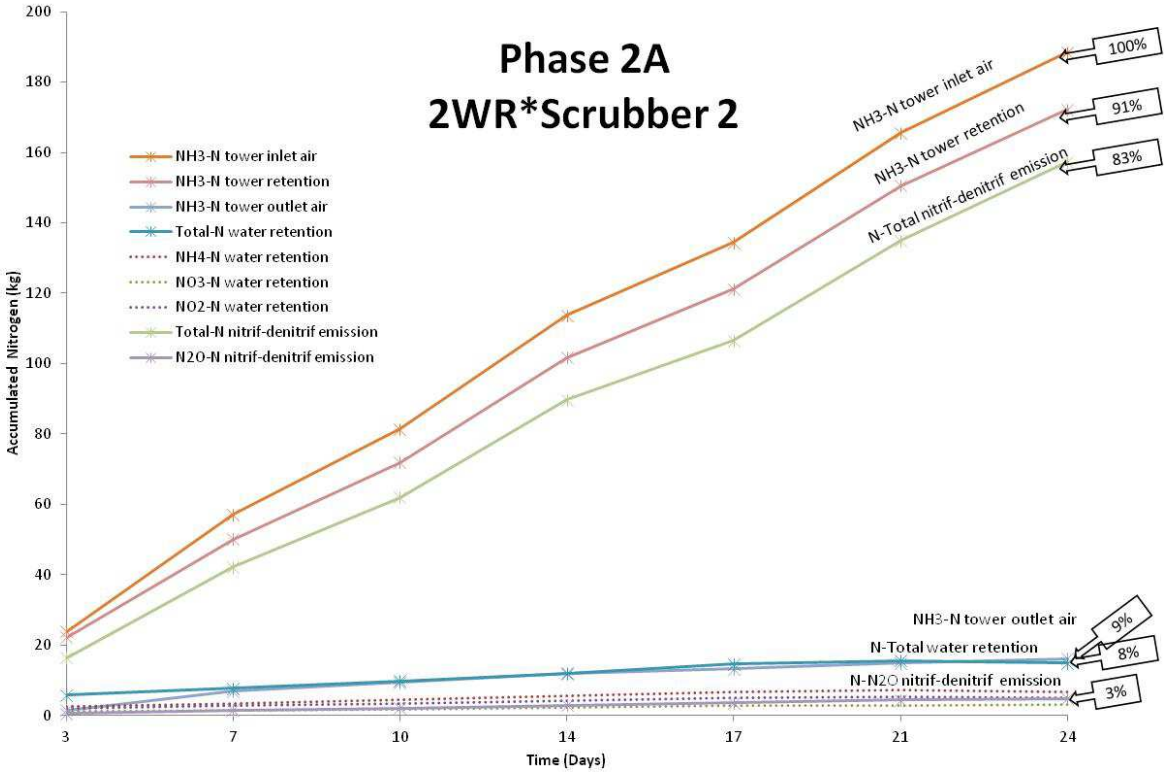
Scrubber	1	2
Sediment content (water tank) (g l^{-1})	0.2	1.3
Nitrification potential (mg NO_2 Kg $sediment^{-1}$ h^{-1})	**	307
Total Viable Count (CFU ml^{-1})	312000	507500

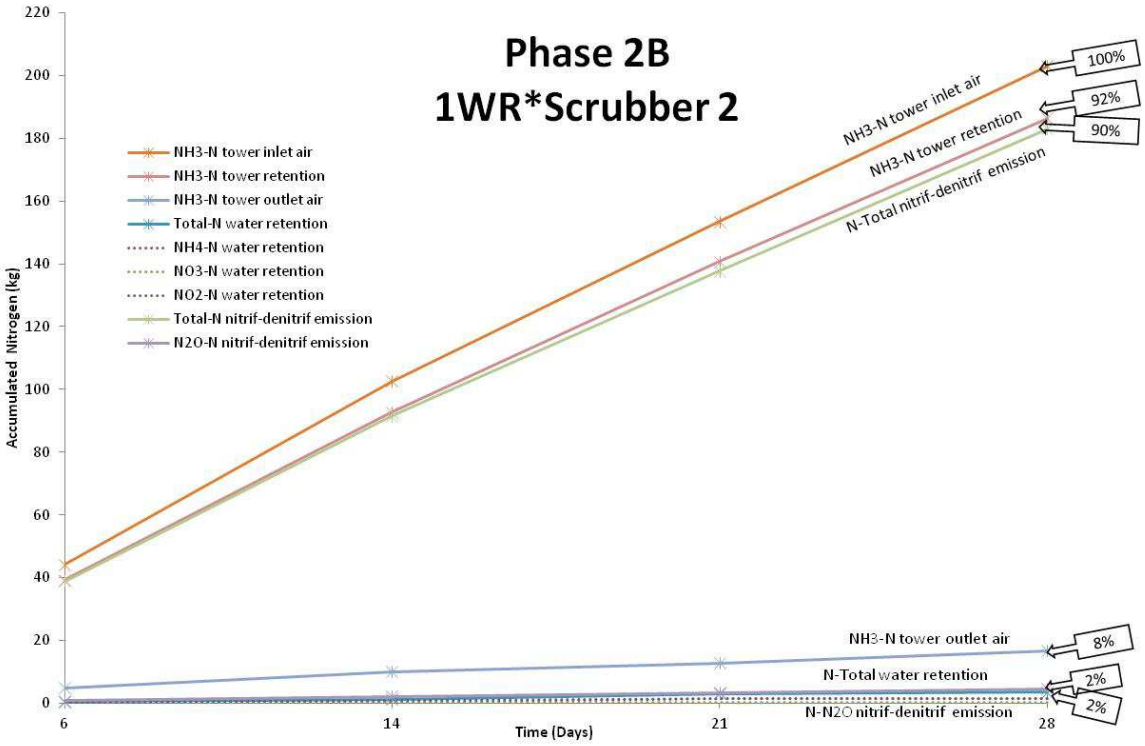
** No measured in Scrubber 1

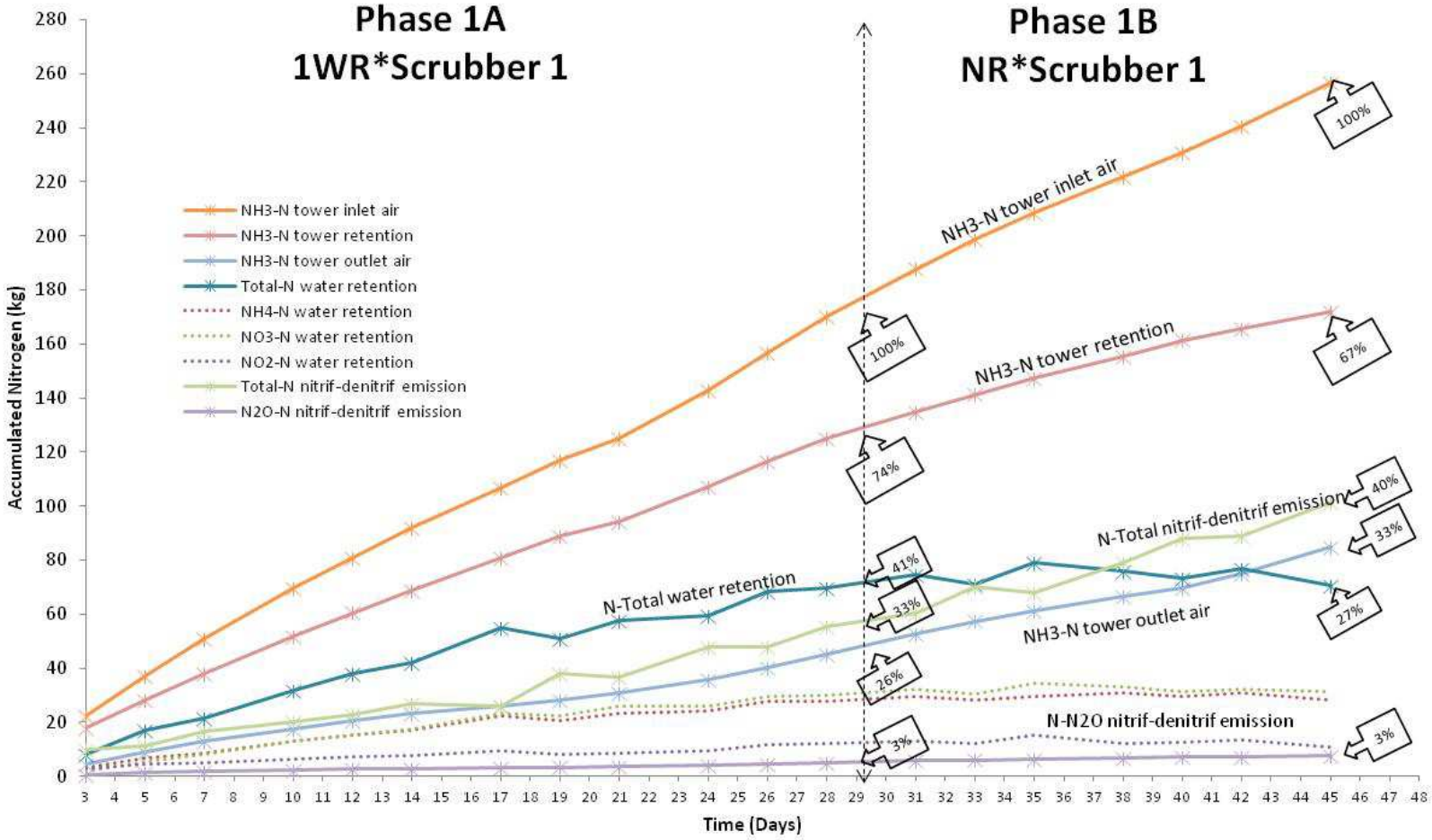
APPENDIX C: Evolution of the nitrogen accumulated in the air and washing water by N species during the tests



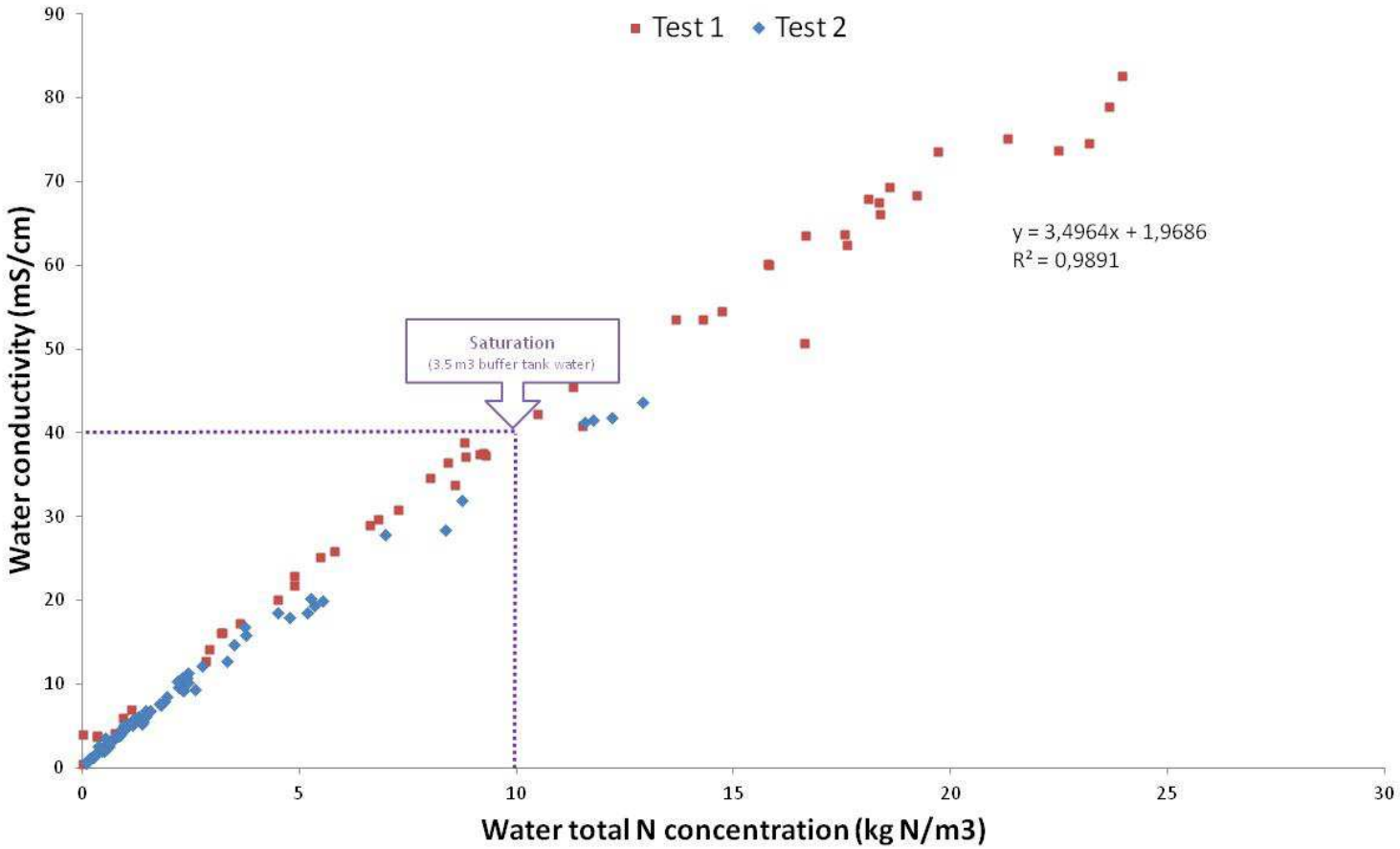



Phase 2A
2WR*Scrubber 2
NH3-N tower inlet air
NH3-N tower retention
NH3-N tower outlet air
Total-N water retention
NH4-N water retention
NO3-N water retention
NO2-N water retention
Total-N nitrif-denitrif emission
N2O-N nitrif-denitrif emission
Accumulated Nitrogen (kg)
Time (Days)
100%
91%
83%
9%
8%
3%


Phase 2B
1WR*Scrubber 2
NH3-N tower inlet air
NH3-N tower retention
NH3-N tower outlet air
Total-N water retention
NH4-N water retention
NO3-N water retention
NO2-N water retention
Total-N nitrif-denitrif emission
N2O-N nitrif-denitrif emission
Accumulated Nitrogen (kg)
Time (Days)
100%
92%
90%
8%
2%
2%


Phase 1A
1WR*Scrubber 1
Phase 1B
NR*Scrubber 1
NH3-N tower inlet air
NH3-N tower retention
NH3-N tower outlet air
Total-N water retention
NH4-N water retention
NO3-N water retention
NO2-N water retention
Total-N nitrif-denitrif emission
N2O-N nitrif-denitrif emission
NH3-N tower inlet air
NH3-N tower retention
N-Total water retention
N-Total nitrif-denitrif emission
NH3-N tower outlet air
N-N2O nitrif-denitrif emission
100%
67%
74%
40%
33%
41%
33%
27%
26%
3%
3%
Accumulated Nitrogen (kg)
Time (Days)

APPENDIX D: Water conductivity- Water N concentration



General Discussion & Conclusions

6.1. Introduction

As introduced in Chapter 1, this thesis aims to assess abatement techniques in the context of commercial farms using a combination of methodologies including emission factors, empirical equations and process-oriented mechanisms. This work has been developed throughtout the 4 previous chapters (Chapters 2 to 5), in which we have tried to analyse main challenges for an effective implementation of BAT and to bring new procedures and tools for BAT decision support, conducting innovation-driven research in collaboration with main stakeholders.

This final chapter is intended to provide a general discussion and main conclusions of the thesis.

6.2. General discussion

The implementation of IPPC (IED) has had significant implications not only for environmental performance, but also for the economic and social dimensions of sustainability (Pellini & Morris, 2001). Depending on existing circumstances and practices on a particular installation, IPPC (IED) could mean additional capital expenditure and changes in operating costs with respect to labour, raw materials, utilities, and time spent on record keeping and environmental management (Pellini & Morris, 2001).

A rational farmer can be expected to select mitigation activities by their cost effectiveness. Win-win techniques which fit well into farming operation and implying decreasing or no additional cost remain most popular (Franks & Hadingham, 2012). However, as discussed by Loyon et al. (2016) (Thesis Chapter 2), despite the abundance of possible techniques and the huge amount of published material and research carried out, there is a low level of BAT implementation in practice, unlikely to be enough to fulfill the environmental targets set.

The diversity of the farming systems and of socioeconomic conditions, the perceived high cost joined to the lack of demonstration on the benefits and the environmental impact of these techniques in commercial farms, may be some of the reasons (Loyon et al., 2016) (Thesis Chapter 2).

Farms in the EU are not a homogenous group; they produce very different products with different technologies in different environmental conditions and with different farm structures. This all implies that the level of uptaken technologies to protect the environment is variable across the territory and a 'one-size-fits-all' solution is unlikely to be successful, as it is the case of the BAT. It is impossible to classify concrete actions in advance as being 'innovative' or not as it depends on the state of development (EU SCAR, 2013).

Despite huge research carried out during the last years studying BAT, research results are often insufficiently exploited and taken up in practice, and innovative ideas from practice are not captured and spread (Loyon et al., 2016) (Thesis Chapter 2).

The scientific contribution to improving technology uptake is mainly related to the evaluation of the actual performance of BAT in commercial farm systems developing consistent procedures to this aim. Much of the confusion often comes down to the lack of a clear objective against which a system can be measured (Loyon et al., 2016) (Thesis Chapter 2). Simulation modeling clearly has a role but nowadays the need to provide robust and user-friendly decision support tools that can be optimised for specific farms is one of the main challenges of this topic. In this sense, this thesis (Chapters 3 and 4) has provided a framework for future research in support and knowledge transfer models designed for farm-scale operations. Besides, direct measurements on farm are required to develop emission factors for up-scaling farm level processes in modeling work. At this regard, Thesis Chapter 5 deals with the evaluation of the performance of a BAT (wet scrubber) following science-based protocols in a commercial pig farm.

The treatment of the exhaust air from animal houses is a method that has lately gained importance when intensive farming needs to comply with stricter regulations and emission limits and alternative emission reduction principles (such as feed management and adaptation of housing system design) are not sufficient to meet the standards.

Wet scrubbers are required in some Member States in order to comply with acceptable emission levels for ammonia, PM10 and odour, particularly in densely populated areas, in regions characterised by high animal density or in vulnerable natural protected areas (e.g. Natura 2000 sites) (EC, 2015).

Currently, in Southern Europe countries, air cleanning is not a common technique. High animal density areas and odour problems to nearby populations, although existing in some regions (e.g. Catalonia), are less frequent in this case. According to Eurostat database, animal density in 2013 in Spain was 0.62 LU/ha, while Germany, Ireland, Norway, Luxembourg and Denmark were all over 1 LU/ha. For countries like Belgium and Netherlands it can almost be considered a national issue, with 2.74 and 3.57 LU/ha, respectively. Moreover, pits in buildings have less capability than in Nothern countries; the slurry is usually stored in big lagoons during more than 6 months, so end of pipe techniques in housing have less interest as a considerable part of emission take place during outside storage. Besides, buildings with natural ventilation are more common in Southern countries and scrubbers require dynamic ventilation systems. It is also remarkable that decreased benefits in livestock production and lack of public funding have reduced farmers' possibilities to undertake new investments, especially costly techniques like scrubbers. Nevertheless, end-of-pipe air treatment technologies could be interesting mitigation measures under more restrictive NH_3, dust and odour emission scenarios, also for certain farm situations in Southern countries.

However, bioscrubbers often experience operational problems (due to poor process control) which have not been addressed adequately by the moment (Melse, Ogink, Rulkens, 2009; Van der Heyden, Demeyer, Volcke, 2015). This fact has also been corroborated in our study. Continuous measurements of ammonia removal over an extended period have provided insights into variations and fluctuations in removal performance. Variation in ventilation rates, biofilm growth and NH_3 concentration in the inlet air during the different tests carried out have affected to scrubber operation and efficiency. Therefore optimal scrubber management has also to be adjusted to each particular circumstance. At this regard, control of the full-scale installations is recommended to ensure proper operation and maintenance of these systems. Discharge of the washing water can be controlled by electrical conductivity which serves as an indication of the total nitrogen content. Many different inhibition thresholds have been reported in the literature suggesting a maximum value of 20 dS/m (Van der Heyden et al., 2015). In this study, inhibition of NH_3 removal by the scrubbers was detected with conductivity about 40 mS/cm and N content of 10 kg N/m3, measured at maximum water tank level.

Maximum ventilation capacity is of great importance for the cost of air cleaning, since the size of the air scrubber is proportional. This is especially remarkable for Southern

European countries with higher ventilation rates. Adjustable trickling density to keep a sufficiently high liquid-to-gas ratio at high ventilation rates and partial air cleaning systems can be interesting options to reduce installation costs under warmer conditions (Van der Heyden et al.; 2015; Melse, Wagenberg, Mosquera, 2006). More efficient pumps and fans at higher backpressure levels and increasing air inlet surface areas to reduce thickness of the packing and pressure drop, can reduce scrubbers energy use although these solutions have not yet been sufficiently verified by independent research (Melse et al., 2009).

The formation of nitrous acid in air scrubbers, mainly in biological ones, is likely and warrants further research to elucidate the underlying mechanisms. The N_2O emission level registered in our study is in line with the values obtained in literature (Van der Heyden et al., 2015; Hahne & Vorlop, 2004), representing a 2-3% of total inlet NH_3-N in the scrubbers but probably with different pathways of formation due to varying biofilm implantation.

Air scrubbers although technically effective in NH_3 reduction, can lose its efficiency if applied without any further measures in downstream farming activities. If emissions are reduced in housing, more NH_3 reaches the storage, increasing the emissions there. Consequently, part of the mitigation effect in housing is lost. In order to model these interrelationships, animal husbandry and cost-efficiency of BAT must be shown as an entire process chain. Whole-farm models can play an important role at this regard for integrated scenario development and evaluation. The incorporation of new experimental data with known protocols and context conditions (e.g. outdoors climatic conditions, housing characteristics or animal management) enables further simulation adjustment to that particular condition and estimating the effect in the whole farm system (which is practically impossible to measure). This is even more important with costly techniques (as scrubbers) under new environmental conditions (warmer climates). Modelization will facilitate to calculate the cost-efficiency of scrubbers implementation in farm emissions and comparing it with other alternatives before undertaking such important investment.

At this regard, the BATFARM model calculates the emissions status of existing farms and allows simulation of the effect of several farm management strategies, on emissions and consumptions, through the comparison of different scenarios defined by users. It can be helpful to identify the key stages giving rise to farm emission, which would most benefit from implementing environmental techniques, or to set up regional reference situations for farm comparisons.

This farm-scale tool is based on existing models of emissions of atmospheric pollutants from on-farm sources within the manure management system, and accounts for the flows of nutrients and pollutants. The newsworthiness of the tool lies primarily in the link of flows and emissions at the farm scale which provides new insights into the knock-on and side effects of the implementation of single or multiple emission reduction measures, in particular:

- The linking of the different on-farm sources of individual compounds or elements. This enables the user to account for the effect of measures taken to reduce emissions of a given compound or element from one source, on emissions from other sources within the manure management system.

- The linking of the flows and emissions of different compounds or elements. This enables the user to account for the synergistic or antagonistic effects of measures targeted at one compound or element, on the flows and emissions of other compounds or elements.

The model has been designed bearing in mind the farming community and related stakeholders as potential users. Then clear communication between science providers and science users (farming community, extension officers and policy makers) has been and still being primordial to maximize utility of model products to both parties. Using models with farmers and advisors influences the way researchers understand farmers' reality and identify knowledge gaps. At the same time it helps to promote subsequent software uptake to solve real problems, increasing its usability by checking graphical user interface and guidance documents (Le Gal, Dugué, Faure, Novak, 2011; Oliver et al., 2012). The model has been tested by final users (farmers and advisors) in Navarre and Basque Country regions (Spain) before being readily available and accessible to the public. In any case, professional advice with overall farm vision, including economic aspects, should be sought on the interpretation of the results provided by the tool.

The model presented is a big effort in collating the results of different researches and existing models whose individual validity has been assessed in other scientific publications, but it is true that there is no evidence that absolute farm emissions provided by BATFARM model are reliable. However, while it is theoretically feasible to test farm-scale models or tools at the farm scale, in practice, the logistical and financial challenges mean that is practically impossible. The practical alternative is to show that the emissions

estimated from the individual sources are within the range found by empirical experiments, when used within the context of the tool. This does not demonstrate that the tool is accurate, rather that the component models have probably been incorporated into the tool consistently and that the results are not wildly inaccurate. This type of alternative model testing has been carried out in the Chapter 3 of this thesis for typical swine farms under Navarre region conditions as case study. Nevertheless, further research and validation is needed to evaluate model reliability and enable broaden uses, not only for scenario comparisons but also for the calculation of absolute farm emissions. These aspects are expected to be addressed in the future, conducting detailed sensitivity analysis, testing model results with empirical data and comparing it with other similar tools.

There are also other features missing in the initial version of the BATFARM model. For example the current version lacks crop nutrient balance after landspreading and it does not consider litter based systems in swine and laying hen farms. It does not yet consider aqueous streams that pollute the soil, surface and ground water. It would be also be desirable to deepen the additional subroutines methods (cost calculator and energy balance), as they are important aspects of BAT selection. In particular, the model could be completed with cost-effectiveness calculations, considering farm emissions and the cost-benefit throughout the whole production process. It is expected that future developments will overcome these limitations and will also incorporate more types of environmental strategies enriched with new knowledge as it becomes available.

Although environmental considerations are in the top of the decision criteria for BAT selection, other aspects like animal welfare, agronomic, social and health, and economic factors, are of great interest, both for farm community and for the rest of society. Even for the environmental factors the importance of each one can vary in each particular farm. Because of this, a balance needs to be found between these different decision-making considerations when one has to select a specific type of environmental techniques and to implement it. Giner-Santoja et al. (2012) propose a decision-making process for assessing BAT based on Multi-Criteria Decision Analysis applied to ceramic industry. In the case of farms, decision criteria including environmental (ammonia, GHG gases, dust), animal welfare, agronomic (nutrients for plant uptake), social (odours), health (pathogens), and economic factors, are a challenge for future model development. Users would be allowed to vary the weight of the different factors to suit site-specific considerations, so the model would not only be able to evaluate the environmental

performance of the indicated technology, but will also could present the best option according to the site-specific requirements and user-defined criteria, providing very useful information for the stakeholders.

A common challenge faced by models is its data-intensive nature. Due to the uncertainties in pollutant modeling, the evolving evidence base on mitigation method impacts and the variation in costs of implementation over time, the modeling process should be flexible and adjustable (Voinov & Bousquet, 2010; Gooday et al. 2014). To this aim BATFARM model interface has been designed to allow users to improve the parameterisation of existing mitigation techniques and overwrite the baseline emission factors should other references be considered to be more appropriate. However, the availability and the quality of data will constrain model parameterization for specific farms conditions. Emission factors, expressed in units of animals or area, are generalizations given the variability induced by geography and meteorology, methodology of the measurement, and the variety of animal and management types that exist (McGinn, 2006). On this point, continuing refinement of the model is expected to be maintained where possible to accommodate new knowledge and understanding. A periodic revision and update of the model's methods and database, especially of regionalizable values (zootechnical data, climatic information and emission factors) should be carried out to improve estimations at regional scale.

At this regard, a lack of information related to environmental friendly practices or techniques is still found in Spain. In this sense, more work is needed to develop practical and economically-viable mitigation techniques that can be widely practised. Integrated work of different agents working close with farmers to demonstrate the environmental and economic advantages of the implementation of environmental strategies at farm level would be advisable.

6.3. General conclusions

- The large number of available techniques, the diversity of farming systems and of socioeconomic conditions, difficult decision making process for BAT selection. Main scientific challenge at this regard is developing consistent procedures to ensure a fair and trustworthy evaluation of listed techniques in the context of the whole farm, including modelization.

- In this sense, the model developed in this thesis intends to provide a framework for future research in support and knowledge transfer models designed for farm-scale operations. Results produced by the model aims to identify the key stages giving rise to farm emissions, which would most benefit from implementing environmental techniques and predicts farm emission variation under different scenarios. For the moment, results must be interpreted as indicative of the relative emission reduction achieved due to implementation of mitigation practices rather than absolute values. Further research and validation is needed to extend its use. Despite these limitations, the model can be used to assist in decision making regarding BAT selection that are best suited to the region and farm circumstances, offering more evidence based information to stakeholders to encourage them to explore possible farm mitigation options in their operational, tactical and strategic management.

- Wet scrubbers seems to be, also in our climatic conditions, very effective for NH_3 reduction if an adequate washing water management is carried out according to particular circumstances. At this regard biofilm implantation seems to be quite relevant, affecting to N balances, NH_3 retention and probably the pathways for N_2O formation. The efficiency in NH_3 retention was higher for the biofilter where the biofilm was present (over 90%), whereas it was lower for the scrubber without visible biofilm ranged from 17% to 81% depending on water discharge frequency. Nitrous oxide was generated by the scrubbers, increasing from 14% to 74% the initial concentration. Air cleaning may not be a generally applicable technique due to the high implementation cost (5.71-9.78 Eur per kg of NH_3 abated). Besides it can lose its efficiency if applied without any further measures in downstream farming activities.

6.4. References

EC (2015). Best Available Techniques (BAT) Reference Document for the Intensive Rearing of Poultry or Pigs. Final Draft - August 2015, European Commision, 911 pp.

EU SCAR (2013). Agricultural knowledge and innovation systems towards 2020 – an orientation paper on linking innovation and research. Brussels, 204 pp.

Franks J.R. and Hadingham B. (2012). Reducing greenhouse gas emissions from agriculture: Avoiding trivial solutions to a global problem. Land Use Policy 29, 727-736.

Giner-Santonja G., Aragonés-Beltrán P. and Niclós-Ferragut J. (2012). The application of the analytic network process to the assessment of best available techniques. Journal of Cleaner Production 25, 86-95.

Gooday R., Anthony S., Chadwick D., Newell-Price P., Harris D., Duethmann D., Fish R., Collins A. and Winter M. (2014). Modelling the cost-effectiveness of mitigation methods for multiple pollutants at farm scale. Science of the Total Environment 468-469, 1198-1209.

Hahne J. and Vorlop K-D (2004). Waste Air Scrubbers Useful in Reducing Ammonia Emissions from Piggeries?. Landtechnik 59, 56-57.

Le Gal P.-Y., Dugué P., Faure G. and Novak, S. (2011). How does research address the design of innovative agricultural production systems at the farm level? A review. Agricultural Systems 104, 714-728.

Loyon L., Burton C.H., Misselbrook T., Webb J., Philippe F.X., Aguilar M., Doreau M., Hassouna M., Veldkamp T., Dourmad J.Y., Bonmati A., Grimm E. and Sommer S.G. (2016). Best available technology for European livestock farms: availability, effectiveness and uptake. Journal of Environmental Management 166, 1-11.

McGinn, S. M. (2006). Measuring greenhouse gas emissions from point sources in agriculture. Canadian Journal of Soil Science 86, 355-371.

Melse R. W., Ogink N. W. M. and Rulkens W. H. (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, 289-298.

Melse R. W., van Wagenberg A. V. and Mosquera J. (2006). Size reduction of ammonia scrubbers for pig and poultry houses: use of conditional bypass vent at high air loading rates. Biosystems Engineering 95(1), 69-82.

Oliver D.M., Fish R.D., Winter M., Hodgson C.J., Heathwaite A.L. and Chadwick D.R. (2012). Valuing local knowledge as a source of expert data: Farmer engagement and the design of decision support systems. Environmental Modelling & Software 36, 76-85.

Pellini T. and Morris J. (2001). A framework for assessing the impact of the IPPC directive on the performance of the pig industry. Journal of Environmental Management 63, 325-333.

Van der Heyden C., Demeyer P., and Volcke E. I. P. (2015). Mitigating emissions from pig and poultry housing facilities through air scrubbers and biofilters: State-of-the-art and perspectives. Biosystems Engineering 134, 74-93.

Voinov A. and Bousquet F. (2010). Modelling with stakeholders. Environmental Modelling&Software 25, 1268-1281.

List of publications

Refereed journal papers

Loyon L., Burton C.H., Misselbrook T., Webb J., Philippe F.X., **Aguilar M.**, Doreau M., Hassouna M., Veldkamp T., Dourmad J.Y., Bonmati A., Grimm E. and Sommer S.G. (2016). Best available technology for European livestock farms: availability, effectiveness and uptake. Journal of Environmental Management 166, 1-11.

Aguilar M., Cenoz I., Estellés F., Merino P. and Calvet S. Operational factors affect the performance and costs of two biotrickling filters to remove gaseous pollutants from the exhaust air of a pig house. . (Submitted to Biosystems Engineering Journal. Currently under revision).

Conference papers

International Conferences

Aguilar M., Abaigar A., Merino P., Estellés F. and Calvet S. (2010). Effect of a bioscrubber on NH_3, N_2O, CH_4 and CO_2 emissions from a pig facility in Spain. International Conference on Agricultural Engineering, Clermont-Ferrand, France, 6th-8th September. Oral communication.

Aguilar M., Abaigar A., Merino P., Estellés F., Montalvo G., Piñeiro P. and Calvet S. (2010). Effect of water scrubbing on ammonia emissions from a gestating sows building in the South of Europe. 14th Ramiran International Conference, Lisbon, 12nd-15th September. Oral communication.

Aguilar M., Irujo E., Abaigar A., Merino P. and Calvet S. (2012). Practical experiencies with 3 floating covers for slurry storages in Spain. International Conference of Agricultural Engineering, Valencia, Spain, 8th -12nd Julio. Poster.

Aguilar M., Abaigar A., Merino P., Estellés F. and Calvet S. (2012). Study of two energy saving techniques in a broiler farm in Spain. Preliminary results. International Conference of Agricultural Engineering, Valencia, Spain, 8th -12nd Julio. Poster.

Aguilar M., Arriaga H., Dupard P., Lalor S., Fragoso R., Pahl O., Abaigar A., Cordovín L., Boyle M., Lanigan G., Loyon L. and Merino P. (2013). BATFARM

SOFTWARE: A support tool in the selection of environmental strategies in livestock operations in the Atlantic Region. European workshop: Reconciling the environment with livestock management, Rennes, France, 19th -20th March. Oral communication.

Arriaga H., **Aguilar M.**, Abaigar A., Cordovín L. And Merino P. (2013). BATFARM software: A support tool to mitigate gaseous losses at dairy housing in the Atlantic region. Greenhouse Gases & Animal Agriculture (GGAA) 2013 Conference, Dublin, 23rd-26th June. Poster.

Aguilar M., Arriaga H., Dupard P., Lalor S., Fragoso R., Pahl O., Abaigar A., Cordovín L., Boyle M., Lanigan G., Loyon L. and Merino P (2013). BATFARM SOFTWARE: A support tool in the selection of environmental strategies in livestock operations in the Atlantic Region. 15th Ramiran International Conference, Versailles, France, 3rd-5th June. Oral communication.

Arriaga H., **Aguilar M.**, Abaigar A., Cordovín L. and Merino P. (2013). A novel support tool to mitigate nutrient and gaseous losses at dairy housing from EU-Atlantic Region. 15th Ramiran International Conference, Versailles, France, 3rd-5th June. Poster.

Aguilar M., Arriaga H., Dupard P., Lalor S., Fragoso R., Pahl O., Abaigar A., Cordovín L., Boyle M., Lanigan G., Loyon L., Hunter C. and Merino P (2013). BATFARM SOFTWARE: A support tool in the selection of environmental strategies in livestock operations in the Atlantic Region. British Grassland Society 11th research conference, Dumfries, Scotland, 2nd-3rd September. Oral communication.

Aguilar M., Arriaga H., Dupard P., Lalor S., Fragoso R., Pahl O., Abaigar A., Cordovín L., Viguria M., Boyle M., Lanigan G., Loyon L. and Merino P. (2014). A Novel Tool to Assess the N_2O, CH_4 and NH_3 Mitigation Potential of environmental Techniques in intensive Livestock Operations. The conference on Livestock, Climate Change and Food Security, Madrid, Spain, 19th-20th May. Poster.

Aguilar M., Arriaga H., Dupard P., Lalor S., Fragoso R., Pahl O., Abaigar A., Cordovín L., Boyle M., Lanigan G., Loyon L. and Merino P. (2014). A decision support system for environmental strategies: Effect in farm N balance. 18th Nitrogen Workshop, Lisbon, 30th June-3rd July. Poster.

Arriaga H., **Aguilar M.**, Abaigar A., Cordovín L. and Merino P. (2014). BATFARM software: A support tool to mitigate nitrogen losses at dairy housing. 18th Nitrogen Workshop, Lisbon, 30th June-3rd July. Poster.

Aguilar M., Arriaga H., Dupard P., Lalor S., Fragoso R., Pahl O., Abaigar A., Cordovín L., Boyle M., Lanigan G., Loyon L. and Merino P. (2014). Selection of Environmental Strategies in intensive livestock operations. International livestock modelling and research colloquium, Bilbao, Spain, 14th-16th October. Oral communication.

National Conferences

Aguilar, M. (2011). Programa de ayudas a la implantación de MTD (Mejores Técnicas Disponibles). Jornada Cambio Climático y Mundo Rural. CRANA, Pamplona, Spain, 29 September. Oral communication.

Irujo E., **Aguilar M.** and Abaigar A. (2011). Seguimiento de procesos de compostaje en explotaciones ganaderas de Navarra. II Simposio de compostaje: Compostaje, salud del suelo y uso de energía en el mundo rural, Ribadeo, Lugo, Spain, 27-29 July. Poster.

Mangado J.M., **Aguilar M**., Abaigar A., Intxaurrandieta J.M., Eguinoa P., Maeztu F.A. (2012). Experiencias en INTIA sobre la mitigación de la emisión de GEIs en el sector ganadero navarro. I Workshop Remedia, Bilbao, Spain, 8-9 March. Poster.

Aguilar M., Cenoz I., Irujo E., Cordovín L., Merino P., Estellés F. And Calvet S. (2013). Performance of heat exchangers in a broiler farm in Spain. Preliminary results. II Workshop Remedia, Zaragoza, Spain, 11-12 April. Poster.

Aguilar M. (2013). Cálculo de la producción de Nitrógeno en la granja. Jornadas sobre Deyecciones Ganaderas, Barcelona, Spain, 12 November. Oral communication.

Aguilar M., Arriaga H., Dupard P., Lalor S., Fragoso R., Pahl O., Abaigar A., Cordovín L., Boyle M., Lanigan G., Loyon L. and Merino P (2014). BATFARM SOFTWARE: A support tool in the selection of environmental strategies in livestock operations. III Workshop Remedia, Valencia, Spain, 10-11 April. Oral communication.

Aguilar M., Irujo E., Cenoz I., Abaigar A. (2014). Comparison of biomass and gasoil boilers in a swine farm in Spain: effects on emissions and costs. III Workshop Remedia, Valencia, Spain, 10-11 April. Poster.

Arriaga H., **Aguilar M**., Abaigar A., Cordovín L. and Merino P. (2014). BATFARM SOFTWARE: A Support Tool to Mitigate Gaseous Losses at Dairy Housing from EU-Atlantic Region. III Workshop Remedia, Valencia, Spain, 10-11 April. Poster.

Technical papers

Aguilar M., Abaigar A., Merino P., Estellés F. and Calvet S. (2010). Estudio de un lavador de aire en una nave de gestación porcina. Navarra Agraria 183, 41-45.

Abaigar A., **Aguilar M.**, Irujo E., Gárriz I. and Pérez B. (2010). Estudio de MTD en fase de definición en Navarra. Navarra Agraria 179, 41-51.

Aguilar M., Irujo E., Abaigar A. (2010). Cubiertas Flotantes en balsas de purines. Navarra Agraria 182, 48-52.

Aguilar M., Irujo E., Abaigar A. (2013). Alternativas al manejo de la gallinaza en fresco: secado y compostaje. Navarra Agraria 197, 29-36.

Ministerio de Agricultura, Alimentación y Medio Ambiente (2014). Evaluación de Técnicas de reducción de emisiones en ganadería. Sectores porcino y avicultura de carne y puesta. Ficha técnica: Lavador de aire y ventilación centralizada, Fase de Gestación, Porcino. Ministerio de Agricultura, Alimentación y Medio Ambiente, Secretaría General Técnica, Centro de Publicaciones, Madrid, 128 pp.

Aguilar M., Arriaga H., Dupard P., Lalor S., Fragoso R., Pahl O., Abaigar A., Cordovín L., Boyle M., Lanigan G., Loyon L. and Merino P. (2014). BATFARM SOFTWARE: Una herramienta de apoyo en la selección de MTD en explotaciones ganaderas intensivas. Navarra Agraria 207, 13-18.

Aguilar M., Abaigar A., Mangado J.M., Irañeta I., Cordovín L., Iñigo J.A. and Bildarratz G. (2014). Mejores Técnicas Disponibles para la gestión agrícola de estiércoles y purines. Revista Vida Rural 383, 50-58.

Aguilar M., Irujo E., Abaigar A. (2014). Gallinaza. Alternativas a la gestión en fresco. AviNews 6, 42-49.

Aguilar M., Cordovín L., Íñigo J.A. (2015). Claves para la selección de MTD: en explotaciones ganaderas intensivas. Navarra Agraria 213, 14-18.

Aguilar M., Cenoz I., Merino P., Estellés F. and Calvet S. (2016). Lavado de aire de una nave de gestación porcina: últimos resultados. Navarra Agraria 219, 29-36.

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