UNIVERSITÀ DEGLI STUDI DI MILANO
DIPARTIMENTO DI SCIENZE E POLITICHE AMBIENTALI
DEPARTMENT OF ENVIRONMENTAL SCIENCE AND POLICY

EMISSIONS ARISING FROM MANURE MANAGEMENT IN DAIRY FARMS: AN EVALUATION USING LIFE CYCLE ASSESSMENT

Doctoral school in Environmental Sciences
XXIX PhD cycle
AGR 10

PhD candidate: Cecilia Baldini
R10570-R24

Tutor: prof. Marcella Guarino

Academic year 2016/2017

Contents

Abstract

The worldwide increasing meat and milk consumption has driven the shift in livestock farming methods from extensive to intensive, posing a number of significant challenges for animal welfare, environmental sustainability and food security. Livestock's contribution to greenhouse gases (GHG) and ammonia (NH_3) emissions is relevant and, reasonably, ruminants are accused of methane emissions caused by enteric fermentation. Thus, the identification of mitigation strategies for intensive livestock farming is of raising interest in recent years. Handling system can modify the physical and chemical properties of manure, influencing the emission levels of methane, nitrous oxide and ammonia. The aim of this thesis was to evaluate the environmental performances of manure removal systems largely spread in the Po Valley with a two-fold approach: (i) measuring the emissions and (ii) performing a Life Cycle Assessment applied to different housing solutions for dairy farms.

In the first step, a literature review about LCA studies was carried out to deeply understand strengths and weakness of LCA approach. Comparing LCA results related to milk production is difficult and a broader level of harmonization should be reached.

Thereafter, GHG and NH_3 emission levels were measured from farms equipped with different housing solutions. These data were then used as input parameters to compile the LCA inventory and compare the resulting potential impacts to those calculated using recommended emissions estimations. The results underlined the need of more adjustable emission factors, able to reflect more accurately the variability of farms conditions.

Finally manure sampling originated from the same farms were analysed to quantify their biomethane production potential.

The results in this thesis outlined that GHG and NH_3 emissions are influenced by manure management choices, which could play a key role in the reduction of livestock environmental impacts on air. Trade-offs among gases were observed, demonstrating that often a combination of measures is requested to control emissions.

Riassunto

Il crescente consumo di carne e latte a livello mondiale ha causato il cambiamento dei metodi di allevamento da sistemi di tipo estensivo a sistemi di tipo intensivo, e ha fatto emergere una serie di sfide significative nell'ambito del benessere animale, della sostenibilità ambientale e della sicurezza alimentare. Il contributo degli allevamenti alle emissioni di gas ad effetto serra (GHG) e ammoniaca (NH_3) è rilevante e, a ragione, i ruminanti sono accusati di emissioni di metano legate alle fermentazioni enteriche. Perciò l'interesse relativo all'identificazione di possibili strategie di mitigazione da applicare agli allevamenti intensivi è cresciuto negli ultimi anni. I sistemi di gestione possono modificare le caratteristiche fisico-chimiche dei reflui dell'allevamento, influenzando le emissioni di metano, protossido d'azoto ed ammoniaca che da essi hanno origine. Lo scopo di questa tesi è stato quello di valutare le performance ambientali di diversi sistemi di allontanamento delle deiezioni, ampiamente diffusi negli allevamenti della Pianura Padana, con un duplice approccio: (i) misurando le emissioni e (ii) applicando un'Analisi del Ciclo di Vita (LCA) a diverse soluzioni stabulative per stalle di vacche da latte.

Come primo passo, è stata condotta una revisione degli studi LCA per comprendere i punti di forza e le debolezze di questo approccio. Difficoltà sono state riscontrate nel paragonare i risultati di diversi studi relativi alla produzione di latte, facendo emergere la necessità di raggiungere un maggior livello di armonizzazione.

In seguito, sono stati misurati i livelli di emissione di GHG ed NH_3 da aziende con diversi tipi di gestione delle deiezioni. I dati ottenuti sono stati utilizzati come parametri per compilare l'inventario dell'analisi LCA e fare un paragone tra i potenziali impatti che da essa risultano e quelli che si ottengono utilizzando le equazioni raccomandate per la compilazione dell'inventario. Da questo confronto è emersa l'esigenza di disporre di fattori di emissione che rendano la stima più aderente alla variabilità di condizioni riscontrate nella stalla.

Infine, alcuni campioni di refluo provenienti dalle stesse stalle sono stati utilizzati per quantificare la produzione di metano potenzialmente ottenibile.

I risultati di questa tesi sottolineano come la scelta della strategia di gestione degli effluenti sia fondamentale nella riduzione dell'impatto ambientale dell'allevamento,

poiché in grado di influenzare le emissioni di GHG ed NH_3. Inoltre, dal momento che il controllo delle emissioni necessita compromessi tra i diversi gas, spesso è richiesta una combinazione di misure per l'efficace controllo delle emissioni.

E così, scegliere
che ci sia luce nel disordine,
è un racconto oltre le pagine
spingersi al limite,
non pensare sia impossibile
camminare sulle immagini
e sentirci un po' più liberi
se si può tremare e perdersi
è per cercare un'altra via nell'anima,
strada che si illumina,
la paura che si sgretola,
perché adesso sai la verità:
questa vita tu vuoi viverla
vuoi viverla

E così, sorridere
a quello che non sai comprendere
perché il mondo può anche illuderci
che non siamo dei miracoli
e se ci sentiamo fragili
è per cercare un'altra via nell'anima,
strada che si illumina,
e la paura che si sgretola,
perché adesso sai la verità:
questa vita tu vuoi viverla
vuoi viverla

E vivi sempre
Ogni istante

-Elisa-

CHAPTER 1

1 Introduction

1.1 Environmental impacts of livestock production

Global food systems play a pivotal role in anthropogenic environmental change. In particular, the livestock sector is a key contributor to a range of critical environmental problems, such as habitat change and loss of biodiversity, land use and soil degradation, climate change, water use and pollution, water scarcity, eutrophication of water bodies, and toxic emissions (Notarnicola et al., 2015; Pelletier and Tyedmers, 2010). In figures: twenty billion animals make use of 30% of the terrestrial land area for grazing, one-third of global cropland area is devoted to producing animal feed, and 32% of freshwater is used to provide direct livelihood and economic benefits to at least 1.3 billion producers and retailers. As an economic activity, livestock contributes up to 50% of agricultural Gross Domestic Product (GDP) globally (Herrero et al., 2016).

Growing populations, incomes and urbanization have driven the unprecedented increase of livestock products' demand observed over the past few decades, and are projected to drive increases in the consumption of milk and meat over the next 20 years (Dangal et al., 2017; Herrero et al., 2016). These trends, if continued, will exacerbate pressures on ecological systems. Indeed, the intensification of production overlooked sustainability and overall efficiency of the farms, disrupting a finely balanced system, in which animals pull ploughs and carts, and fertilize with their manure crops, which supply post-harvest residues to livestock (Eisler et al., 2014).

Thus, the global livestock sector is now facing with a three-fold challenge: (i) the still ongoing need to increase production to meet demand, (ii) adapt to a changing and increasingly variable economic and natural environment and (iii), at the same time, improve its environmental performance (Opio et al., 2013).

1.1.1 Livestock's contribution to GHG emissions

Carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O) and chlorofluorocarbons are gases responsible of the greenhouse effect. Their molecular structures enable them to trap a fraction of the energy received from the sun in the atmosphere, increasing the temperature of our planet.

The atmospheric concentrations of these greenhouse gases (GHG) augmented in an alarming way since the start of the industrial era: +40%, +150% and +20% respectively for CO_2, CH_4 and N_2O (Tian et al., 2016). The climate changes resulting from this increase pose a serious threat to the environment, economy and well-being of both human and animals (Sejian et al., 2015).

According to IPCC (2014):

- CO_2 accounts for around three-quarters of the warming impact of current human GHG emissions. The key source of CO_2 is the burning of fossil fuels such as coal, oil, and gas, though deforestation is also a very significant contributor.
- CH_4 accounts for around 16% of the impact of current human GHG emissions. Key sources of this gas include agriculture (especially livestock and rice fields), fossil fuel extraction, and the decay of organic waste in landfill sites. Methane doesn't persist in the atmosphere as long as CO_2, though its warming effect is much more potent for each gram of gas released (Global Warming Potential 34 of CO_2 eq in 100 years-time horizon, according to IPCC (2013)).
- N_2O accounts for around 6% of the warming impact of current human GHG emissions. Key sources are agriculture (especially nitrogen-fertilized soils and livestock waste) and industrial processes. N_2O is even more potent per gram than methane (Global Warming Potential 298 of CO_2 eq in 100 years-time horizon, according to IPCC (2013)).

Depending on the accounting approaches and scope of emissions covered, estimates by various sources (IPCC, FAO, USEPA or others) place livestock contribution to global anthropogenic GHG emissions at between 7 and 18% (Hristov et al., 2013; O'Mara, 2011). Although it accounts for only 9% of global CO_2, the livestock sector generates 65% of human-related nitrous oxide (N_2O) and 35% of CH_4 (Sejian et al., 2015).

Among livestock, ruminants are the primary emitters contributing to the largest anthropogenic source (25%–40%) of CH_4 emission, with cattle representing the 65% of the livestock sector's emissions (Dangal et al., 2017; Gerber et al., 2013). Enteric fermentation and manure management are the main processes driving CH_4 emissions from cattle species. Enteric emissions from cattle represent 46% and 43% of the total emissions in dairy and beef supply chains, respectively (FAO, 2016). The CH_4 emissions from enteric fermentation constitute a physiological by-product of the digestive process of ruminants, and are influenced by feed quantity and quality, body weight, feeding level and the activity and health of livestock (Dangal et al., 2017). The CH_4 emission from manure depends on the decomposition process, which is influenced by climate, and the way in which manure is collected and stored before its application (Chadwick et al., 2011).

Apart from being a strong GHG with a residence time of 130 years in the atmosphere, N_2O is also the largest anthropogenic stratospheric ozone-depleting substance. Main sources of N_2O emissions are manure management and the application and deposition of manure. Manure-derived nitrous oxide (N_2O) accounts for 44% of total anthropogenic N_2O emissions (Zhang et al., 2017). Furthermore, volatilization losses of NH_3 and NOx from manure management systems and soils lead to indirect N_2O emissions.

The manure handling system determines the moisture content and oxygen availability in the manure, influencing the emission levels with liquid systems producing predominantly CH_4 and solid system producing both CH_4 and N_2O (Hristov et al., 2013).

1.1.2 Livestock's contribution to NH_3 emissions

Ammonia (NH_3) is the most abundant alkaline compound in the atmosphere (Behera et al., 2013). It has many negative effects on ecosystems function and health, and on air quality. Deposition of NH_3 and NH_4^+ ions significantly contributes to soil acidification, eutrophication of natural ecosystems, and nitrate leaching (Yang et al., 2017). Furthermore, NH_3 is a chemically active gas and readily reacts with sulfuric acid and nitric acid in the lower atmosphere to form secondary inorganic particulate matter with diameters ≤ 2.5 µm ($PM_{2.5}$), which have been implicated in human respiratory problems and other environmental effects (Xu et al., 2017). Since NH_3 abundance is a key element in the $PM_{2.5}$ formation, recent studies have shown that the reduction of its emission in the future could be a cost-effective strategy for air quality control compared to further abatement of sulfur dioxide, and nitrogen oxides emissions (Backes et al., 2016; Xu et al., 2017).

The agricultural sector is currently responsible for the vast majority of NH_3 emissions in the European Union. About 94% of the global anthropogenic NH_3 emissions is associated to agriculture practices, and Italy is one of the Member States with the highest contribution in 2014 (EEA, 2017).
Emissions of NH_3 mainly occur from feces and urine in housing and manure storage systems, from excreta of grazing animals voided on pastures and from agricultural land following application of manure and mineral N fertilizers (Velthof et al., 2014). In the EU-28, the principal key categories for NH_3 emissions are (i) cattle manure management - 31% of NH_3 emissions; (ii) inorganic N-fertilizers - 21% of NH_3 emissions; (iii) swine manure management - 13% of NH_3 emissions; (iv) animal manure applied to soil - 10% of NH_3 emissions (EEA, 2017).

1.2 The Po Valley context

Livestock farming has an outstanding role in the economies of the northern Italian regions. In 2010, Italy recorded one of the highest values among EU-27 Member State as number of agricultural holdings and in terms of Utilized Agricultural Area (UAA). Although the agricultural sector showed a marked reduction in the period 2000-2010 (-32.4% agricultural holdings, -14.4% labour force), the Italian livestock population remained rather constant (-0.6%) over the inter census decade: 10 million Livestock Unit (LSU) were recorded in 2000 while 9.9 were surveyed in 2010 (EUROSTAT, 2012).

In 2010, 90% of the Italian livestock population consisted of cattle (44%), pigs (24.8%) and poultry (21.6%). In absolute terms, cattle accounted for 4.4 million LSU and recorded a decrease of 3.6% over the Farm Structure Survey (FSS) 2000 value; pigs indicated the value of 2.5 million LSU and a +6.7 % growth compared to 2000; poultry recorded 2.1 million LSU, and a fall of 2.5 % compared with 2000.

In terms of animal livestock, the northern regions of Lombardia, Emilia-Romagna, Piemonte and Veneto accounted together for 64% of the Italian LSU in 2010. In particular, the Lombardia region hosts a significant part of Italian livestock population: 28% of LSU in 2010, +6.7% compared to the FSS 2000 (EUROSTAT, 2012). It accounts for 7.9% of the Italian territory and is characterised by an intensively managed agriculture with one of the highest livestock density in the world (Zucali et al., 2017). In 2016 the total number of dairy cows reared in the region was 478 881 (the 26% of national consistency) with an average production of 9 793 kg of milk cow^{-1}. The amount of milk delivered to the dairy industry was 4 887 200 tons (+3.97%) about the 43% of national milk production (CLAL, 2016).

Important Protected Designations of Origins (PDO) products are made by dairy industry of the Po Valley regions (e.g. Grana Padano, Parmigiano Reggiano, Gorgonzola, Taleggio, etc.) and constitute an essential cultural heritage of this area.

These figures underline the relevance of the livestock sector in contributing to environmental pressure in this area.

The INEMAR database (INEMAR, 2017) provides the emission inventories for the pollutants responding to the Convention on Long-Range Transboundary Air Pollution (CLRTAP) for several Italian regions. The latest inventory, referred to 2014 period, confirmed the high contribution of the agricultural sector in the emission of CH_4, N_2O and NH_3 in Lombardia, accounting respectively for the 56.8%, 81.4% and 97.8% (Figure 1.1).

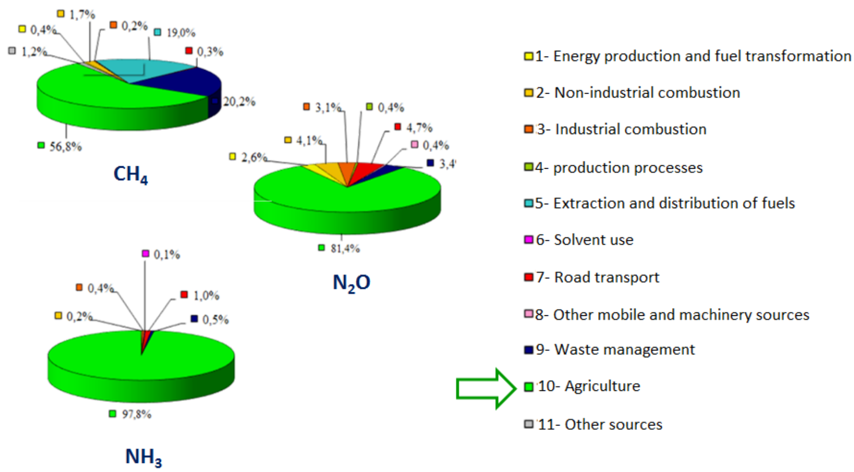


Figure 1.1. Sectorial emissions of CH_4, N_2O and NH_3 in Lombardia for the year 2014 (INEMAR, 2017).

1.3 Life cycle assessment

Life Cycle Assessment (LCA) is a holistic approach for evaluating environmental impacts of products, processes and services throughout their "life cycle", from production, to use, end-of-life and waste management (ISO, 2006a). Life cycle assessment (LCA) is now recognized as one of the most complete and widely used methodology frameworks developed to assess the environmental impact of products and processes and can be used as a decision support tool within environmental management.

Its long history started in 1960s, when the first studies about environmental implications of alternative sources of energy were performed. The study carried out by Harry Teasley and others at the Coca-Cola Company in 1969 is certainly the most famous among "proto-LCA" studies (Hunt and Franklin, 1996). The research was aimed at identify the best packaging in terms of environmental releases and laid the foundation for the current method of life cycle inventory analysis. In 1970s and 1980s, several industrial LCA studies emerged. Finally, the general structure and related terminology of "Life Cycle Assessment" was finally recognised in 1990s, thanks to the coordination activities promoted by SETAC, the Society of Environmental Toxicology and Chemistry (Guinée et al., 2011). Next to SETAC, the International Organization for Standardization (ISO) has been involved in LCA since 1994, with the formal task of standardizing methods and procedures through the development of the ISO 14040 series (ISO, 2006a; b).

In 2002, the United Nations Environment Programme (UNEP) and SETAC launched an International Life Cycle Partnership, known as the Life Cycle Initiative (Guinée et al., 2011), whose main aim was formulated as putting life cycle thinking into practice and improving the supporting tools through better data and indicators.

More recently, other Life Cycle Approaches (Life Cycle Costing and Social-LCA) were introduced to complete the picture of life-cycle thinking, dealing with the other key pillars of sustainability (Benoît and Mazijn, 2009; Klöpffer, 2003) and gaining a noteworthy role in achieving the Sustainable Development Goals defined in the 2030 Agenda (UN, 2016).

According to ISO 14040 (ISO, 2006a), LCA consists of four phases (Figure 1.2):

1. Goal and scope definition - this phase provides the framework of the study and defines the functional unit (FU, to which all subsequent inputs and outputs are related) and system boundary (within which the unit processes of the system are contained);
2. Life cycle inventory (LCI) - in which all relevant input and output processes are defined, quantified and summarized;
3. Life Cycle Impact Assessment (LCIA) – which links the LCI results to specific impact categories (e.g. Global Warming Potential, Acidification, etc.) through the application of characterization factors;
4. Analysis and interpretation of results.

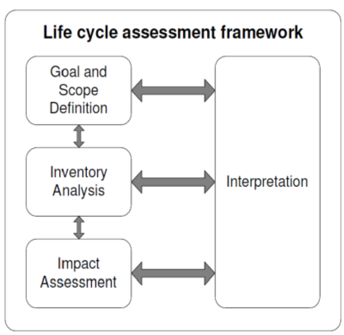


Figure 1.2. LCA framework.

Starting from 2000s, LCA has gained popularity in evaluation of milk production throughout the world. Several studies have been published as a consequence of the rising concern about livestock productions and the significant pressure they pose to the environment. Several international initiatives related to livestock carbon footprint (an LCA with global warming as the only impact category) were developed: FAO has modelled the carbon footprint of global dairy sector (FAO, 2010), and International Dairy Federation (IDF) has published guidelines on performing carbon footprint (IDF, 2010).

Important publications signed by FAO were the "Greenhouse emissions from ruminant supply chains" (Opio et al., 2013) and "Tackling climate change through livestock" (Gerber et al., 2013), which provide a synopsis of GHG emissions for all livestock sectors, including dairy, and exploiting mitigation potential and options (Notarnicola et al., 2015).

The launch of the Livestock Environmental Assessment and Performance (LEAP) Partnership in 2012, constituted an important step forward in achieving a higher level of standardization in food-LCA. Addressing the specific problems related to livestock production LCA, the LEAP aims at developing both globally accepted assessment methodologies and reference databases to support better environmental management of livestock production systems via environmental benchmarking. The first version of the "Environmental Performance of Large Ruminant Supply Chains: Guidelines for Assessment" was recently published (FAO, 2016), with the main purpose of developing clear guidelines for environmental performance assessment based on international best practices, providing sufficient definition of calculation methods and data requirements to enable consistent application of LCA across differing large ruminant supply chain.

References

Backes A. M., Aulinger A., Bieser J., Matthias V., 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., 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.

Benoît C., Mazijn B., 2009. Guidelines for social life cycle assessment of products, UNEP/SETAC Life Cycle Initiative. Sustainable Product and Consumption Branch Paris, France.

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

CLAL, 2016. Italian dairy cow population. October 2017. https://www.clal.it.

Dangal S. R., Tian H., Zhang B., Pan S., Lu C., Yang J., 2017. Methane emission from global livestock sector during 1890–2014: Magnitude, trends and spatiotemporal patterns. Global Change Biology.

EEA, 2017. European Union emission inventory report 1990–2015 under the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP). Publications Office of the European Union, 2017, Luxembourg.

Eisler M. C., Lee M., Tarlton J. F., Martin G. B., Beddington J., Dungait J., Greathead H., Liu J., Mathew S., Miller H., 2014. Agriculture: Steps to sustainable livestock. Nature; 507: 32.

EUROSTAT, 2012. Agricultural census in Italy. October 2017. http://ec.europa.eu/eurostat/statistics-explained/index.php?title=Agricultural_census_in_Italy&oldid=197716#Livestock.

FAO, 2010. Greenhouse Gas Emissions from the Dairy Sector: A Life Cycle Assessment. Rome.

FAO, 2016. Environmental performance of large ruminant supply chains: Guidelines for assessment. FAO, Rome, Italy.

Gerber P., Steinfeld H., Henderson B., Mottet A., Opio C., Dijkman J., Falcucci A., Tempio G., 2013. Tackling climate change through livestock.

Guinée J. B., Heijungs R., Huppes G., Zamagni A., Masoni P., Buonamici R., Ekvall T., Rydberg T., 2011. Life cycle assessment: past, present, and future. ACS Publications.

Herrero M., Henderson B., Havlík P., Thornton P. K., Conant R. T., Smith P., Wirsenius S., Hristov A. N., Gerber P., Gill M., 2016. Greenhouse gas mitigation potentials in the livestock sector. Nature Climate Change; 6: 452-461.

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., Oosting S., 2013. Mitigation of greenhouse gas emissions in livestock production – A review of technical options for non-CO2 emissions. Edited by Pierre J. Gerber B. H. a. H. P. S. M. FAO, Rome, Italy., FAO Animal Production and Health Paper No. 177.

Hunt R. G., Franklin W. E., 1996. LCA - How it Came about - Personal Reflections on the Origin and the Development of LCA in the USA. International Journal of Life Cycle Assessment; 1: 4-7.

IDF, 2010. A common carbon footprint approach for dairy - The IDF guide to standard life cycle assessment methodology for dairy sector. International Dairy Federation.

INEMAR, 2017. INventario Emissioni ARia. October 2017. http://www.inemar.eu.

IPCC, 2013. In: Stocker T., Qin D., Plattner G., Tignor M., Allen S., Boschung J., et al., editors. 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.

IPCC, 2014. Summary for Policymakers. In: Edenhofer O., Pichs-Madruga, R., Sokona, Y., Farahani, E., Kadner, S., Seyboth, K., Adler, A., Baum, I., Brunner, S., Eickemeier, P., Kriemann, B., Savolainen, J., Schlömer, S., von Stechow, C., Zwickel, T. and Minx, J.C., editors. Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge Univeristy Press, Cambridge, United Kingdom and New York, NY, USA.

ISO, 2006a. Environmental Management - Life Cycle Assessment-Principles and Framework. EN ISO 14040:2006. EN ISO 14040. International Organization for Standardization. Geneva, Switzerland.

ISO, 2006b. Environmental Management - Life Cycle Assessment-Requirements and Guidelines. EN ISO 14044:2006. EN ISO 14044:2006. International Organization for Standardization. Geneva, Switzerland.

Klöpffer W., 2003. Life-cycle based methods for sustainable product development. International Journal of Life Cycle Assessment; 8: 157-159.

Notarnicola B., Salomone R., Petti L., Renzulli P. A., Roma R., Cerutti A. K., 2015. Life Cycle Assessment in the Agri-food Sector: Case Studies, Methodological Issues and Best Practices. Springer,

O'Mara F. P., 2011. The significance of livestock as a contributor to global greenhouse gas emissions today and in the near future. Animal Feed Science and Technology; 166: 7-15.

Opio C., Gerber P., Mottet A., Falcucci A., Tempio G., MacLeod M., Vellinga T., Henderson B., Steinfeld H., 2013. Greenhouse gas emissions from ruminant supply chains–A global life cycle assessment. Food and agriculture organization of the United Nations (FAO), Rome.

Pelletier N., Tyedmers P., 2010. Forecasting potential global environmental costs of livestock production 2000–2050. Proceedings of the National Academy of Sciences; 107: 18371-18374.

Sejian V., Gaughan J., Baumgard L., Prasad C., 2015. Climate change impact on livestock: Adaptation and mitigation. Springer, New Delhi.

Tian H., Lu C., Ciais P., Michalak A. M., Canadell J. G., Saikawa E., Huntzinger D. N., Gurney K. R., Sitch S., Zhang B., 2016. The terrestrial biosphere as a net source of greenhouse gases to the atmosphere. Nature; 531: 225-228.

UN, 2016. Transforming our world: the 2030 Agenda for Sustainable Development October 2017. http://www.un.org/sustainabledevelopment/.

Velthof G., Lesschen J., Webb J., Pietrzak S., Miatkowski Z., Pinto M., Kros J., Oenema O., 2014. The impact of the Nitrates Directive on nitrogen emissions from agriculture in the EU-27 during 2000–2008. Science of the Total Environment; 468: 1225-1233.

Xu P., Koloutsou-Vakakis S., Rood M., Luan S., 2017. Projections of NH3 emissions from manure generated by livestock production in China to 2030 under six mitigation scenarios. Science of The Total Environment; 607-608: 78-86.

Yang W., Zhu A., Zhang J., Xin X., Zhang X., 2017. Evaluation of a backward Lagrangian stochastic model for determining surface ammonia emissions. Agricultural and Forest Meteorology; 234-235: 196-202.

Zhang B., Tian H., Lu C., Dangal S. R., Yang J., Pan S., 2017. Global manure nitrogen production and application in cropland during 1860-2014: a 5 arcmin gridded global dataset for Earth system modeling. Earth System Science Data; 9: 667-678.

Zucali M., Tamburini A., Sandrucci A., Bava L., 2017. Global warming and mitigation potential of milk and meat production in Lombardy (Italy). Journal of Cleaner Production; 153: 474-482.

CHAPTER 2

2 Framework of the thesis

2.1 Objectives

There is a raising interest in the identification of GHG and NH_3 mitigation possibilities achievable with currently available technologies (O'Mara, 2011). Indeed, techniques and management practices that could help reducing emissions exist, but are not widely used and their implementation is limited by cost implications (Gerber et al., 2013). This thesis aimed at understanding the environmental performance of different manure handling systems commonly applied in the Italian context, with a particular focus on the emissions arising from the housing facilities/structures hosting dairy cows. The appraisal was carried at two different levels: (i) measuring the actual level of emissions arising from different housing solutions for dairy farms, and (ii) applying the Life Cycle Assessment methodology for a broader evaluation of the environmental burdens associated to milk production.

Specific aims were to:

- evaluate the comparability and the level of harmonization among LCA studies;
- underline the strengths and weakness of milk LCA, identifying emerging issues and hot topics that should be included for further developments of the methodology;
- compare the emissions of NH_3, N_2O, CH_4, and CO_2 arising from different floor types in dairy barns, defining specific emission factors;

- analyze the contribution of different shed components (feeding alley and resting zone) to the gaseous emissions;
- calculate the environmental impact of different dairy farms using different data sources for the emissions arising from the manure management;
- evaluate the impact associated to different animal age classes and categories in order to identify their contribution to the overall impacts of milk production;
- estimate the methane production potential achievable from manure samples taken from different farms, understanding the influence of manure handling systems on the biogas yield of the Anaerobic Digestions (AD) process.

2.2 Overview of the chapters

A comprehensive evaluation of different manure handling solutions is detailed in the following chapters. Chapter 1 introduces the thesis, while Chapter 2 describes its scope. Scientific papers elaborated during the PhD course are presented in Chapter 3, 4, 5 and 6.

Chapter 3 describes the evolution of LCA studies applied to the dairy sector. A literature review was performed to provide an overview of the LCA methodology, and allowed a better understanding of the main findings emerging from the first decade of application of LCA in the milk sector. Statistical analyses were also carried out, to underline and quantify the influence of some practitioners' choice on the results of the environmental assessment.

In Chapter 4, four farms located in the Po Valley context were selected to measure the emission levels arising from different areas of the barns, equipped with different housing solutions.

A further step of the research is presented in Chapter 5, where the measurement data obtained in the aforementioned farms were used to conduct a full LCA study. A comparison between measured and estimated emissions from the manure management was carried out, aiming at identify the level of convergence of the different approaches.

Considering manure as a resource, and not as a waste material, is of paramount importance in the management of a farm, responding to the challenge of environmental sustainability. The strategy chosen for manure collection changes the physical and chemical characteristics of manure and influences the downstream treatments. In Chapter 6, an evaluation on the yield of biogas, and methane, achievable using manure samples originating from different handling systems (and thus with different characteristics) was performed.

Chapter 7 discusses and draws the general conclusions of the previous chapters.

References

Gerber P., Steinfeld H., Henderson B., Mottet A., Opio C., Dijkman J., Falcucci A., Tempio G., 2013. Tackling climate change through livestock.

O'Mara F. P., 2011. The significance of livestock as a contributor to global greenhouse gas emissions today and in the near future. Animal Feed Science and Technology; 166: 7-15.

CHAPTER 3

3 A critical review of the recent evolution of Life Cycle Assessment applied to milk production

Baldini, C., Gardoni, D., Guarino, M., 2017. Published in Journal of Cleaner Production; 140: 421-435.

Abstract

Life Cycle Assessment (LCA) is one of the key tools for the evaluation of the environmental sustainability of the agricultural sector. LCA related to milk production has gained attention in recent years, but the results are often discordant and conditioned by the practitioners' choices. This has made it difficult to clearly identify the most environmentally friendly way to produce milk.

In the present paper, 44 milk LCA studies published after 2009 are reviewed, in order to evaluate the level of harmonization and comparability of methods and results, and to discuss emerging issues and hot topics that would be worth further investigation. Furthermore, the effects that the choice of functional unit and allocation rule could have on the results were statistically analyzed. The understanding of the current research direction of milk-LCA studies is useful to promote a more responsible and sustainable livestock production in the perspective of increasing animal protein demand.

This review highlighted the difficulties encountered in comparing milk LCA studies, underlining the importance of practitioners' choices in determining the results.

Harmonization among LCA studies applied to the milk sector still represents a goal to be achieved. It appeared that future LCAs should investigate a broad range of impact categories, including biodiversity and water consumption; define one or more common functional units; improve transparency, giving a detailed description of system boundaries and reporting the method used for impact calculation; and systematically conduct a sensitivity analysis for a better understanding of the effects of the choices of method.

Keywords: Life Cycle Assessment; environmental impact; milk production; dairy farm; review.

3.1 Introduction

Nowadays agriculture and the food system which it underpins are at a crossroads (Soussana, 2014), facing the challenge of producing more food without intensifying environmental pressure (Sutton et al., 2013). The increasing demand caused by world population growth and dietary changes (the augmentation of meat and milk consumption) is driving the intensification of production, while environmental threats such as climate change, biodiversity loss and degradation of land and fresh water foster public concern about agriculture's environmental footprint (Foley et al., 2011). The shift towards a sustainable food system is becoming urgent, and this may be the main reason contributing to the marked spread of food-based Life Cycle Assessment (LCA) observed in recent years (O'Brien et al., 2012a; Van Der Werf et al., 2014).

Taking into account the whole life cycle of a product, the LCA method aims to quantify the environmental pressures and/or the benefits related to goods and services (products), as well as the trade-offs and the scope for improving areas of the production process (EPLCA, 2015).

The LCA is a fluid method, applicable to all production sectors. It was internationally standardized by ISO 14040 and 14044 (ISO, 2006a, 2006b). These documents outlined a procedure shared among all the sectors for which an LCA calculation could be of interest, and constitute one of the major attempts at harmonization among studies. The method defined by ISO is grounded upon four main pillars: goal and scope definition, inventory, impact assessment, interpretation of results. Every step entails several choices, and each one of them could affect the final results of the analysis. The LCA is now one of the leading methodologies for environmental metrics and it will potentially become a powerful strategic management and decision-making tool to make our society more sustainable and resource-efficient (Wolf et al., 2012; Teixeira, 2015). The main strength of this method is the systems perspective that aims to avoid the “shifting of burdens” from one environmental impact to another and from one stage of production to another (Hellweg and Canals, 2014).

Already in 2003, the European Commission officially recognized the role of LCA in providing “the best framework for assessing potential impacts of products currently available” -EU Communication, COM(2003)302- and undertook the effort of further debate about LCA standardization, launching the project for the International Reference Life Cycle Data System (ILCD) Handbook (IES, 2010a, 2010b, 2010c, 2010d, 2011). The ILCD Handbook consists of a set of documents that are in line with the ISO standards but which further specifies their broader provision, offering a basis for consistent, robust and quality-assured environmental LCA studies (Wolf et al., 2012). The intention of these documents was also to serve as a “parent” document for the development of sector-specific guidelines, useful to provide the most suitable solutions for day-to-day problems. For the dairy sector, this call was partially answered by the International Dairy Federation with the publication of “A common carbon footprint approach for dairy - The IDF guide to standard life cycle assessment methodology for dairy sector” (IDF, 2010). This document was the result of collaboration among the main organizations involved in improving the standardization of the LCA approach (ISO, British Standards Institution, FAO, IPCC, Carbon Trust, World Business Council for Sustainable Development and World Resources Institute). Even though it concerns only the global warming potential resulting from dairy activities, it was developed with the aim of unravelling ambiguities about some well-debated aspects within the method (functional unit, boundaries, land use change, co-products handling).

The first version of the “Environmental Performance of Large Ruminant Supply Chains: Guidelines for Assessment” by the LEAP Partnership (LEAP, 2016) constitutes the latest effort in defining a harmonized application of LCA in the livestock sector. The main purpose of the guidelines is "to provide sufficient definition of calculation methods and data requirements to enable consistent application of LCA across differing large ruminant supply chain". Compared to the IDF document, the LEAP guidelines are focused more generally on the livestock sector as a whole, not only on dairy production. The guidelines are explicitly addressed to climate change, fossil energy use and water use over the key stages of the cradle-to-primary-processing-gate. They are more exhaustive and offer many more details and practical examples, but are firmly addressed to experts with a good working knowledge of the LCA applied to animal production.

However, despite the four decades of methodological development (Teixeira, 2015) and the efforts for standardization previously described, the method still lacks of a fully harmonized approach. Indeed choices and hypotheses made by the practitioners, as well as the data used, can affect the comparability of the studies, and could lead to different results from the same subject matter (Lifset, 2012; Fantin et al., 2014; Pelletier et al., 2015). This reduces the power of LCA as decision tool, since the potential inconsistency between methodological choices acts as a deterrent in many public and policymaking contexts (Ridoutt et al., 2015). Hellweg and Canals (2014) specified that LCA is a tool permitting a comprehensive understanding of a problem rather than providing a single answer.

The aim of our review is to investigate the recent evolution of LCA applied to milk production, identifying trends among the main methodological approaches in order to evaluate the level of harmonization among studies and their comparability. Moreover, emerging issues and hot topics that would be worth further investigation in future LCA studies are underlined. The attempt at understanding the current research direction of milk-LCA studies and summarizing their results will be useful in order to promote a more responsible and sustainable livestock production in the perspective of an increasing animal protein demand.

3.2 Methods

A systematic search of scientific literature was carried out in order to find studies evaluating the environmental impacts of dairy farms. The checked databases were

Scopus and ISI Web of Knowledge (www.scopus.com; www.isiwebofknowledge.com), which were visited last time on 19th of May 2015. The inserted keywords were "Dairy LCA" and "Life Cycle Assessment dairy farms". The selection of papers was then refined by publication year, thus studies published before 2009 were excluded since they were already discussed in other LCA reviews (de Vries and de Boer, 2010; Yan et al., 2011; Arvanitoyannis et al., 2014). In order to make feasible and consistent the comparison among papers, the search results were examined by title and abstract and the following selection criteria were applied. i) The paper must be written in English and published in a peer-reviewed journal after 2009. ii) The study must be related to milk production from dairy cattle farming systems. Studies regarding the processing of milk after farm production (i.e. pasteurization of HQ milk, production of UHT milk, etc.) were retrieved, whereas those dealing with other products derived from milk (yoghurt, cheese, whey, etc.) were excluded. iii) Only studies performing an impact assessment with more than one LCA indicator were retained, considering impact categories and technical quantities (land use and non-renewable energy consumption).

All types of studies (i.e. original field investigations, modeling studies, Life Cycle Assessment studies and review articles) were included in the selection.

The selected studies were checked, tracing the LCA phases (goal definition, inventory analysis, impact assessment, interpretation) identified by the ISO standard (ISO, 2006a, 2006b). Hence, the key elements of the selected LCAs were extrapolated, applying the filtering criteria reported in Figure 3.1.

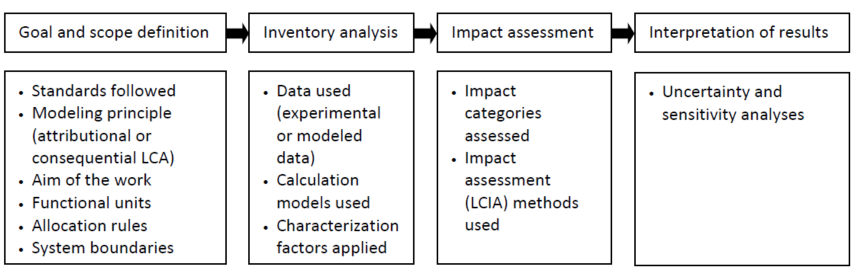


Figure 3.1. Evaluation scheme used for the original papers, modified from Laurent et al. (2014).

Data for the most commonly evaluated impact categories (Global Warming Potential - GWP, Acidification Potential - AP, Eutrophication Potential - EP, Energy Use - EU, Land Use - LU) were used to investigate the effect that methodological choices (functional unit and allocation method) had on the results. Since the probability density functions of the selected variables (the aforementioned impact categories) are unknown, a non-parametric test (Kruskal-Wallis test) was used to verify whether samples originated from the same distribution. SAS 9.3® software was used for the statistical analysis.

3.3 Results and discussion

After a first selection, a list of 130 publications was chosen for further evaluation. A substantial amount of studies (over 60) focused only on assessing climate change impacts with disregard for other environmental problems. This practice is not fully compliant with the principle, claimed in the ISO standard (ISO, 2006a), of avoiding the shifting of a potential environmental problem to another due to a lack of a comprehensive view of the environmental impacts (Čuček et al., 2012; Hellweg and Canals, 2014). These studies were therefore not considered. Among the review papers, only those explicitly considering milk production were retrieved.

The filtering resulted in 44 papers of which 29 were original papers, 8 reviews, 5 scenario analyses, 2 research directions. Table 3.1 reports all the selected studies, classified by publication type, geographical area, research focus, functional unit, system boundaries and impact coverage.

As outlined in Table 3.1, the majority of retained papers (25) investigated the problem in European countries (mainly Italy, Ireland and France), reflecting the existence of a lasting political focus and a growing public interest on environmental optimization of the dairy sector in these regions.

Table 3.1. List of selected studies.

Reference	Pub[1]	Country	Research focus[2]	FU[3]	System boundary[4]	Impact coverage[5]
Arsenault et al. (2009)	OA	Canada	C (high vs low use of pasture)	rm	Farm	GWP, AP, LU, EP, EU, POF, TET, FET, HT, OD, AD.
Arvanitoyannis et al. (2014)	R	-	-	-	-	-
Bartl et al. (2011)	OA	Peru	C (highland vs coast)	ECM, animal, ha	Farm	GWP, AP, EP.
Basset-Mens et al. (2009)	OA	New Zealand	C (level of intensification)	FPCM, ha	Farm	GWP, AP, LU, EP, EU.
Battini et al. (2014)	OA	Italy	C (presence of a biogas plant)	FPCM	Farm	GWP, AP, LU, FEP, MEP, EU, PMRI, POF.
Bava et al. (2014)	OA	Italy	ID, Man	FPCM, ha	Farm	GWP, AP, LU, EP, EU.
Castanheira et al. (2010)	OA	Portugal	ID	rm	Farm	GWP, AP, EP, POF, AD.
Cederberg et al. (2013)	RA	-	-	-	-	-
Chen and Corson (2014)	OA	France	ID, Un	FPCM, ha	Farm	GWP, AP, EP.
Crosson et al. (2011)	R	-	-	-	-	-
de Vries and de Boer (2010)	R	-	-	-	-	-
Djekic et al. (2014)	OA	Serbia	ID	final product	Factory	GWP, AP, EP, POF, HT, OD.
Fantin et al. (2012)	OA	Italy	ID	HQ milk TetraTop® packaged	Factory	GWP, AP,EP, EU, POF, OD, AD.
Garnett (2009)	SA	-	-	-	-	-
Garnett (2014)	SA	-	-	-	-	-
Gerber et al. (2011)	SA	-	-	-	-	-
González-García et al. (2013)	OA	Portugal	ID	ECM	Factory	GWP, AP, EP, EU, POF, TET, FET, MET, HT, OD, AD.
Guerci et al. (2013a)	OA	Italy	ID, Man	ECM	Farm	EU, LU, GWP, AP, EP.
Guerci et al. (2013b)	OA	Denmark, Germany, Italy	ID	FPCM	Farm	GWP, LUC*, AP, LU, EP, EU, Biodiversity.
Heller and Keoleian (2011)	OA	USA	ID, CM (allocation)	packaged fluid milk	Grave	GWP, EU.
Iribarren et al. (2011)	OA	Spain	ID, EE	rm	Farm	GWP, AP, LU, EP, EU.
Jan et al. (2012)	OA	Switzerland	C (hill vs mountain), EE	ns	Farm	GWP, AP, LU, EP, EU, TET; FET, MET, HT.
Kristensen et al. (2011)	OA	Denmark	ID, CM (allocation)	ECM	Farm	GWP, LU.
Meier et al. (2015)	R	-	-	-	-	-
Meul et al. (2014)	OA	Belgium	ID, Man	FPCM	Farm	GWP, LUC, AP, LU, EP, EU.
Nguyen et al. (2013a)	OA	France	Cons	FPCM, live weight	Farm	GWP, LUC, AP, LU, EP, EU.
Nguyen et al. (2013b)	OA	France	C (breed, feed, production system), CM (allocation)	FPCM	Farm	GWP, LUC*, LU.
Nijdam et al. (2012)	R	-	-	-	-	-
Notarnicola et al. (2012)	SA	-	-	-	-	-
O'Brien et al. (2012b)	OA	Ireland	C (confinement vs grazed)	FPCM, ha	Farm	GWP, LUC*, AP, LU, EP, EU.
Penati et al. (2013)	OA	Italy	ID, Man	FPCM, ha	Farm	GWP, AP, LU, EP, EU.
Roer et al. (2013)	OA	Norway	ID, CM (system boundaries)	ECM, carcass	Farm	GWP, AP, LU, FEP, MEP, POF, TET, FET, MET, HT, OD, AD.
Ross et al. (2014)	OA	United Kingdom	C (forage regimen and breeds), Un	ECM	Farm	GWP, LU.
Roy et al. (2009)	R	-	-	-	-	-
Sasu-Boakye et al. (2014)	OA	Sweden	C (localization of food production)	ECM	Farm	GWP, LUC, LU.
Thomassen et al. (2009)	OA	Netherlands	ID, EE	FPCM	Farm	GWP, AP, LU, EP, EU.
Tuomisto et al. (2012)	R	-	-	-	-	-
van der Werf et al. (2009)	OA	France	ID, New (tool)	FPCM, ha, €	Farm	GWP, AP, LU, EP, EU, TET.
Van Der Werf et al. (2014)	RA	-	-	-	-	-
Webb et al. (2014)	SA	-	-	-	-	-
Yan et al. (2011)	R	-	-	-	-	-
Yan et al. (2013a)	OA	Ireland	C (different LCA approaches)	ECM	Farm	GWP, AP.
Yan et al. (2013b)	OA	Ireland	ID, Man	ECM	Farm	GWP, LU.
Zehetmeier et al. (2014b)	OA	Germany	C (breeds and regions)	FPCM	Farm	GWP, LU.

[1] Publication type: OA=Original Article; R=review; RA=research address; SA= scenario analysis.
[2] Research focus: C=comparative study (object of comparison); CM=changing methodology (object of change); Cons=consequential LCA; EE=economic evaluation; ID=identification of environmental performance; Man=management evaluation; New=new approach or tool; UN=uncertainties evaluation.
[3] Functional Unit: ECM=Energy Corrected Milk; FPCM= Fat and Protein Corrected Milk; rm=raw milk; ha= hectare; ns=not specified.
[4] System boundaries: Farm=from cradle to farm gate; Factory=from cradle to factory gate; Grave=from cradle to grave.
[5] Impact coverage: Global Warming (GWP), Land Use Change (LUC), Acidification Potential (AP), Land Use (LU), Eutrophication Potential (EP), Freshwater Eutrophication Potential (FEP), Marine Eutrophication Potential (MEP), Energy Use (EU), Photochemical Oxidants Formation (POF),

3.3.1 Standardization

Among selected studies, only 12 clearly stated that they followed the ISO standards in their assessment; 6 studies mentioned them generically, while 11 studies did not cite them at all. The reference or lack of reference to the ISO standards cannot be considered as an indicator of the level of knowledge of the LCA method, nor as a guarantee of the reliability of the results, but it is interesting to note that almost 40% of the authors did not take these documents into account when preparing a scientific publication. This could be due to the general nature of the principles included in the ISO, which do not answer the specific problems related to the milk sector (functional unit, system boundaries, handling multifunctionality).

The IDF (2010) is the other important standard, specific for the dairy system. Since its publication in 2010, only a small number of studies have followed this guideline (10 of 24 selected papers, while 5 articles were published before its release date), making the efforts made to reach uniformity about LCA rather poorly undertaken. Due to its recent publication date, at the moment the reference to LEAP (2016) was not taken into account by any author.

3.3.2 Modeling principle

The LCA method (ISO, 2006a, 2006b) allows two different modeling principles to be used for the analysis of the system: the attributional and the consequential model. Due to its easier applicability, the attributional model is the most widely used in all the sectors where the LCA method has been applied (IES, 2010a). This was observed also for the papers considered in the present work. Only Nguyen et al. (2013a) used a consequential model in order to evaluate how climate change and land use could vary if the French population increased its consumption of milk obtained from a grass-based system.

3.3.3 Aims of the studies

According to their aims, LCA studies can be divided into two main groups: the descriptive and the comparative ones. In the first group of studies, the assessment aims to identify the environmental burdens of a selected system, while in the second group a direct comparison between two different systems is drawn. Among reviewed papers, 16 are descriptive (ID), while 12 of them are comparative (C) (see Table

3.1). It was not possible to attribute one of these two main categories to Nguyen et al. (2013a) since, as previously discussed, they used a consequential approach. As well as this raw division, the research focus of each work was identified in order to find out the principal research lines among the LCA studies.

The influence that management options (both the general farming strategy and the different farmer choices) could have on the environmental impact of a considered farm is an interesting subject matter. This topic was approached in various ways. Some authors compared a priori two management options such as different levels of intensification (Arsenault et al., 2009; Basset-Mens et al., 2009; Kristensen et al., 2011; O'Brien et al., 2012b), breeds or feeding regimes (Nguyen et al., 2013b; Ross et al., 2014), or localization of production (Bartl et al., 2011; Jan et al., 2012; Zehetmeier et al., 2014b). Other authors faced the issue a posteriori, considering a large number of farms and trying to evaluate the farm characteristics that most influence the results (Guerci et al., 2013a; Penati et al., 2013; Yan et al., 2013b; Bava et al., 2014; Meul et al., 2014). Finally, other authors (Thomassen et al., 2009; Iribarren et al., 2011; Jan et al., 2012) emphasized the economic aspect of the management choices, with an eco-efficiency evaluation.

Other recurrent topics were the evaluation of changes in method - different allocation rules, system boundaries, or LCA approach - (Heller and Keoleian, 2011; Kristensen et al., 2011; Nguyen et al., 2013b; Roer et al., 2013; Yan et al., 2013a) or of the uncertainty of results (Chen and Corson, 2014; Ross et al., 2014). It is interesting to note that other authors also mentioned these issues, but without reporting them in the scope of the study (Arsenault et al., 2009; Bartl et al., 2011; O'Brien et al., 2012b; Guerci et al., 2013b; Battini et al., 2014; Sasu-Boakye et al., 2014).

3.3.4 Impact coverage

Table 3.2 reports the impact categories addressed in the selected studies. The global warming potential (GWP) is the most widely studied impact category (all the 29 original articles). Other commonly considered environmental problems are the acidification potential (AP), eutrophication potential (EP), land use (LU) and energy use (EU). Finally, less investigated impact categories are (in decreasing order):

ecotoxicity, photochemical ozone formation, human toxicity, ozone depletion and abiotic depletion. Interesting and emerging topics not sufficiently addressed are: land use change (6 studies, of which only half provided quantitative results); biodiversity loss, considered only by Guerci et al. (2013a); and water consumption, investigated in none of the selected papers since this impact category is usually addressed in stand-alone assessments (Water Footprints).

Table 3.2. Impact coverage of the considered studies.

Environmental impact	Number of studies*
Global warming potential	
GWP	29
Land use change	6 (3)
Acidification potential	22
Land use	21
Eutrophication potential	
Eutrophication (not specified)	19
Freshwater eutrophication	2
Marine eutrophication	2
Energy use	17
Photochemical ozone formation	7
Ecotoxicity	
Freshwater ecotoxicity	4
Marine ecotoxicity	3
Terrestrial ecotoxicity	5
Human toxicity	5
Ozone depletion	5
Abiotic depletion	4
Biodiversity	1
Particulate matter/respiratory inorganics	1
Waste	1

* Number in brackets refers to studies that consider the impact category without giving quantitative results.

3.3.5 Functional unit

The functional unit (FU) gives a quantitative description of the primary function fulfilled by the system under study (Yan et al., 2011; Mc Geough et al., 2012) and, according to ISO 14044 (ISO, 2006b), only products with similar FUs can be compared.

Some authors (Arsenault et al., 2009; Castanheira et al., 2010; Iribarren et al., 2011) chose the mass (kg) or volume (L) of the raw milk as the FU, assuming milk production as the primary function of a dairy farm. A similar criterion was applied by Heller and Keoleian (2011), Fantin et al. (2012) and Djekic et al. (2014), who selected the packaged milk mass (kg) or volume (L) as FU, since they expanded the system boundaries beyond the farm gate.

On the other hand, other authors preferred to emphasize the nutritional function of milk, and to correct the raw production according to its energy content. This adjustment allows a fair comparison between milk with different fat and protein contents, accounting for animals of different breeds or feeding regimes (FAO, 2010; IDF, 2010). However, the standardization of this unit is not well established and two different correction formulae to predict the energy content of milk can be found in the literature: Energy Corrected Milk (ECM) (1), and Fat and Protein Corrected Milk (FPCM) (2).

(1) $kg\ ECM = kg\ milk \cdot (0.25 + 0.122\ fat\ \% + 0.077\ protein\ \%)$ (Sjaunja et al., 1990)

(2) $kg\ FPCM = kg\ milk \cdot (0.337 + 0.116\ fat\ \% + 0.06\ protein\ \%)$ (Fao, 2010)

Both those equations express the mass of milk required to provide the same energy of a milk with a standardized composition - 4% of fat and 3.3 % of protein, giving 3.15 MJ/kg (IDF, 2010) - but with different coefficients that lead to a slight difference in the final result. Yan et al. (2011) observed that the choice between these two formulae was based on geographical identity: Swedish and Irish scientists used ECM, while Dutch scientists used FPCM. However, the selection criteria of the equation were never specified and for this reason the results are not clear. This regional connotation cannot be totally confirmed, since in recent years was partially overlapped, i.e. see O'Brien et al. (2012b). Furthermore the IDF document (IDF, 2010) recommends using the first equation (1), but named it as FPCM instead of ECM, thus giving rise to possible misunderstanding.

In the reviewed articles, 13 authors used the FPCM formula, while 9 authors used the ECM formula to correct the milk production (see Table 3.1).

Reporting the entire formula used to correct the energy content of milk would be a good practice, since ambiguous information could arise if the formula for FU calculation is not clearly described or is reported without a reference source. Fourteen of the revised papers do not fulfill this requirement, making the comparison among the studies difficult.

Nine studies express the results using multiple functional units. Beside to those referring to product quantity, the mainly used FU is the hectare of land.

3.3.6 Allocation rules

Meat, energy from biogas production, and even manure, are some of the possible co-products of milk production. The overall impacts should be partitioned among the various outputs of the system, in order to calculate the actual environmental impact of a single product deriving from a multi-functional process, like dairy farming. The handling of co-products is one of the most debated and unresolved issue of the milk LCA method (Notarnicola et al., 2015), since the allocation factors strongly affect the results (FAO, 2010; Flysjö et al., 2011; Kristensen et al., 2011; Nguyen et al., 2013b; Notarnicola et al., 2015).

The ISO (2006b) provides a hierarchical level of criteria for dealing with multi-functional processes. Other standards, even if based on the same criteria, are inconsistent with the ISO hierarchy (BSI, 2008), and give rise to different interpretations that lead to divergent results (Dalgaard et al., 2014; Weidema, 2014; Pelletier et al., 2015). For this reason it is very important to establish a shared approach among the different guidelines (Flysjö et al., 2011; O'Brien et al., 2014).

Nevertheless, both the ISO (2006b) and the IDF (2010) define the same criteria and the same priority level for their application. First of all, allocation should be avoided: either dividing the process into sub-processes each one producing a single output, or applying the system expansion. This literally means an enlargement of the system boundaries in order to include the additional functions related to the co-products. For the dairy system, it implies that the meat produced by the dairy farm substitutes for another product that fulfills the same needs of the consumer (Flysjö et al., 2011). Hence, the substitution leads to an “avoided burden” that could be used as a credit for the dairy system.

Alternatively, if allocation could not be avoided, it should be based on physical relationship between products. Several allocation approaches were developed to divide impacts between milk and meat. The IDF (2010) recommended a biological method that is centered on the feed energy utilization and quantifies the energy needed by the cow to produce a kg of milk or meat. On the other hand, FAO (2010) underlined the primary function of the dairy sector to provide humans with protein, and proposed a protein allocation method that enables direct comparison with other food products.

Finally, if any other relationship cannot be identified, the ISO (2006b) and the IDF (2010) as well, will suggest basing the allocation on the economic value or the mass of the different outputs. The mass allocation accounts only for the raw production of milk and meat, even if there is not a causal relationship between milk and beef masses and impacts (Thoma et al., 2013). On the other hand, the main disadvantage of the economic allocation is that it depends on place and time and makes the comparison difficult across regions (FAO, 2010).

Among reviewed papers, the allocation rules applied to distribute environmental impacts between milk and meat were identified. For three studies (Jan et al., 2012; Djekic et al., 2014; Sasu-Boakye et al., 2014) it was not possible to identify the allocation rule applied, hence they were not included, while some authors (Bartl et al., 2011; Heller and Keoleian, 2011; Kristensen et al., 2011; Nguyen et al., 2013b) compared the results obtained using different allocation procedures. As shown in Figure 3.2, most of the authors (15 of the selected studies) chose the economic value as the criterion for repartitioning the environmental burdens among milk and meat, even if ISO (2006b) and IDF (2010) indicated that as the third choice. Also, de Vries and de Boer (2010), reviewing papers published up to 2009, observed that the economic allocation is the procedure most frequently applied. The biological criterion was not so widespread despite the recommendation of IDF (7 of the selected articles, 5 of them published after the publication of IDF guidelines). Other authors (6 of the selected studies) decided to attribute all the environmental impacts of the dairy farm to milk, applying no allocation factors. Finally, the remaining papers were based on different methods: mass allocation (3), protein allocation (2), system expansion (2) and other methods (4).

The results obtained by authors that directly compared different allocation procedures between milk and meat are reported in Table 3.3, in order to describe how this choice affects the results. Because of the great importance of this topic, in this case it was decided to enlarge the comparison, including also the results referring to Carbon Footprint (CF) studies, although they were originally excluded from our review. Other authors, among both reviewed studies and CF studies, have addressed this issue but their results cannot be included in Table 3.3 since they cannot be referred to a "no allocation" scenario (Thomassen et al., 2008; Bartl et al., 2011; O'Brien et al., 2012b).

The comparison among different allocation methods within a single study is useful to understand the consistency of the results obtained. In fact, as reported in Table 3.3, there is a significant difference in the estimated GHG emissions when different allocation procedures are applied.

Table 3.3. Percentage of global warming potential attributed to milk using different allocation rules. Only commonly adopted allocation procedures are reported. The 100% refers to "no allocation" scenario.

	Study	System expansion	Biological (IDF, 2010)	Protein (FAO, 2010)	Economic	Mass
LCA	Heller and Keoleian (2011)	-	-	-	94-95%	97-98%
LCA	Kristensen et al. (2011)	75-78%	71-76%	81-83%	86-88%	-
LCA	Nguyen et al. (2013)	49-73%	72-83%	"similar to Kristensen et al. (2011)"*	"similar to Kristensen et al. (2011)"*	-
LCA	Battini et al. (2014)	-	-	-	93.4%	97.4%
Carbon Footprint	Flysjö et al. (2011)	63-76%	85-86%	93-94%	88-92%	98%
Carbon Footprint	O'Brien et al. (2014)	59-71%	87-88%	94-95%	90-93%	98%
Carbon Footprint	Flysjö et al. (2012)	45-63%	-	-	-	90%
Carbon Footprint	Zehetmeier et al. (2014)	46-77%	-	-	-	-
Carbon Footprint	Mc Geough et al. (2012)	-	86%	-	91%	-
Carbon Footprint	Cederberg and Stadig (2003)	63%	85%	-	92%	-

*As stated in the text.

Otherwise, applying the same allocation method for comparing different dairy systems, O'Brien et al. (2014) observed a different percentage of GHG emissions allocated between milk and meat. As a result, the ranking of CF of milk from dairy systems was not consistent among allocation methods. Hence, when referring to a given dairy system, the selection of a particular allocation method could be influenced by the advantages and disadvantages that are entailed in this choice, leading to possible distortions.

The problem is emphasized when using system expansion, since the results obtained highly depend on the type of meat that is assumed to be replaced (Flysjö et al., 2011;

Kristensen et al., 2011; Nguyen et al., 2013b; O'Brien et al., 2014). Furthermore this approach was criticized since the "avoided burden" of beef meat production may not be true because of increasing meat consumption (Crosson et al., 2011). To this end, Flysjö et al. (2011) proposed to use a system analysis to study a more sustainable way to provide a growing population with animal protein, accounting for all animal sectors and considering milk and meat production in an integrated manner. As suggested also by Weidema et al. (2008), this is the only way to understand the implication of changes in milk production. The intensification of milk production, through increased milk yield per cow, will lead to a decrease of the cattle herd required to produce the current milk demand. Therefore, additional beef production could be necessary, in order to maintain meat consumption. Consequently, on a global scale, it there would not necessarily be any significant reduction in GHG emissions. Recently, the LEAP guidelines (LEAP, 2016) decided to exclude the application of system expansion from the allocation options.

Additionally, Bartl et al. (2011) underlined the need for further research focused on the quantification of the non-monetary values of the cattle herd, especially in subsistence systems, since the common allocation methods do not consider these aspects.

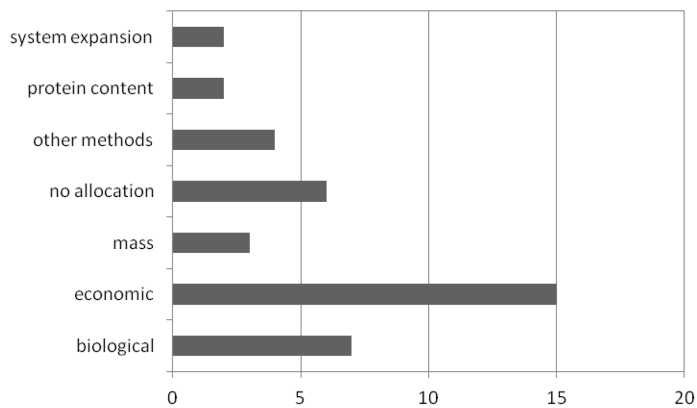


Figure 3.2. Allocation methods applied in the selected papers. Since some papers compare different allocation procedures, the total number of cases extracted from the selected papers was 39.

3.3.7 System boundaries

The definition of system boundaries is greatly influenced by the modeling framework. In the attributional context they trace the production-chain logic of the process, while in consequential modeling they are expanded to the processes conditioned by the consequences of the decisions under study (IES, 2010a).

An LCA analysis ideally entails all the aspects related to a product, "from cradle to grave" (i.e. from raw material acquisition through production, use, end-of-life treatment, recycling and final disposal). However, it is possible to circumscribe the assessment, focusing on specific phases of the production chain to limit the complexity of the study (Figure 3.3). This "from cradle to gate" analysis is generally preferred by milk LCA practitioners, who frequently conducted their analysis up to the farm gate (25 of the selected studies). Some authors (Djekic et al., 2014; Fantin et al., 2012; González-García et al., 2013) expanded the system boundaries up to the factory gate, considering also the impacts generated during milk processing but neglecting how the environmental impacts of milk are influenced by the final consumer behavior. Finally only Heller and Keoleian (2011) extended the analysis through the waste management after consumption, describing the first comprehensive LCA of a vertically integrated organic dairy of the USA. Limiting the LCA to the farm gate is justified by the fact that the impacts of the dairy farm phase dominate the total life cycle for all damage categories (Notarnicola et al., 2015). In addition, this choice allows to pay more attention to understanding the environmental hotspots associated with the dairy farm.

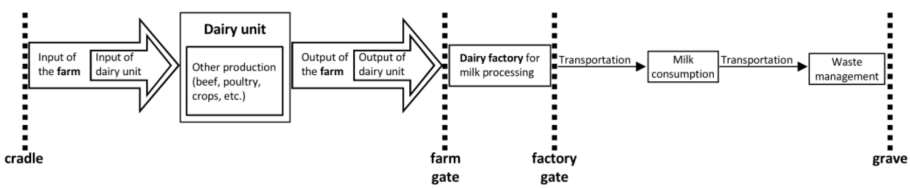


Figure 3.3. Schematic presentation of the life cycle milk production, defining the system boundaries that can be considered in a LCA of milk sector, modified from Yan et al. (2011).

An accurate definition of the system boundaries ensures the possibility of replicating the analysis and reduces the risk of burden shifting from one part of the life cycle to another. ISO 14044 (ISO, 2006) proposes to describe the system boundary using a flow diagram, in order to define in a clear manner what is accounted for, and to specify the inter-connection among the phases of the process. The scheme, if sufficiently detailed, is very useful for the reader and helps towards better understanding of the system considered. Although most studies comply with this requirement, it is not a systematic practice (the diagram was lacking in 7 reviewed papers). Jan et al. (2012) provided a good example of diagram defining the boundary of the system, where the authors divided inputs from production processes and reported them in a clear manner.

On the other hand, extrapolating unambiguous information about the system boundaries within the text of milk LCA studies was difficult. In fact, while some authors were extremely precise in the description and listed all the materials entering the system (Battini et al., 2014; Roer et al., 2013), others (Ross et al., 2014) specified in the text only what is not considered, omitting what is actually accounted for. Information about the system boundaries which was too generic was encountered in Kristensen et al. (2011), O'Brien et al. (2012), Sasu-Boakye et al. (2014), none of whom provided any flow diagram.

All the available data about system boundaries (reported in the text or in the scheme) presented in each LCA study were collected and are reported in Figure 3.4. It can be noted that almost all the selected studies were clearly declared to include purchased feed, mineral fertilizers, fossil fuels and other energy carriers. Other processes/input materials frequently included were pesticides, replacement animals, transportation of input to the farm and bedding materials. However, as already underlined by Yan et al. (2011), capital goods (infrastructure and machinery) and veterinary medicines are rarely included in milk LCA, both due to the lack of data and the assumption of similarities among farms.

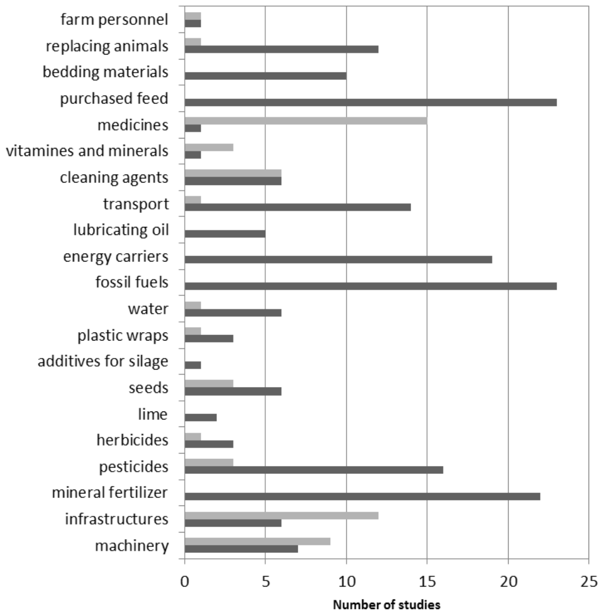


Figure 3.4. Main categories encountered in the system boundaries of the selected studies. Heavy and light gray represent respectively the number of studies that explicitly account or do not account for the considered category.

3.3.8 Life Cycle Inventory (LCI)

Usually, the data gathering for the quantification of all the inputs and outputs is the most demanding task in conducting an LCA study. It is very time consuming (Roy et al., 2009), and the quality of such data is considered as the major bottleneck of robust LCAs (Brandão et al., 2012; Notarnicola et al., 2015). In this context, it is useful to distinguish between two kinds of data: foreground and background data. Foreground data describe the actual system which is the object of the study, while background data refer to the systems delivering energy and material to the foreground system and include also the estimation of the pollutants' emission factors arising from the production chain (Yan et al., 2011; Pirlo and Carè, 2013).

Regarding the foreground data referring to the on-farm stage, the reviewed studies were divided into three categories:

1. real data, collected in real farms;
2. average data, taken from national inventories or referring to modeled farms;

3. literature data, collected in previous campaigns with different aims or referring to former LCA studies but re-elaborated in a different manner.

Fifteen of the selected studies collected inventory data from real farms by interviewing farmers, while other authors used literature or average data (respectively 7 and 7).

Concerning the calculation model used for the estimation of emissions, it was difficult to delineate a trend in the reviewed studies, since information about LCI was often incomplete. However considerations previously reported by Notarnicola et al. (2015) can be confirmed: there is more consensus in the estimation of GHG emissions, for which IPCC equations and reference values were generally used. On the other hand, NH_3 emissions were estimated with a broader range of emission factors, as already observed by de Vries and de Boer (2010). Quantification of NH_3 emission has to take into account manure composition and facilities (composition and pH, type of housing, storage and manure application technique) but also climatic conditions such as ambient temperature and wind speed (LEAP, 2016). Furthermore, Hristov et al. (2011) indicated that NH_3 emission factors from large ruminant production systems are highly variable with dairy farms. Data gaps for some NH_3 emission factors were claimed by Cederberg et al. (2013), who included the improving of models for N fluxes and emissions from soil in the top priorities of LCA research.

Regarding background information, Ecoinvent is the most widely used data set (cited in 18 of the selected papers), but other databases were also employed. BUWAL, IDEMAT, Franklin, and CCalc are different databases encountered in the selected studies.

Transparency in reporting LCI information is very important, in order to guarantee the reproducibility of the results. This basic principle was not always respected in the LCA studies reviewed. Just for an example, in some case, the Tier of IPCC used to calculate the emission factors of GHG was not indicated.

Furthermore, the understanding of the calculation models used is not simple for the reader due to the descriptive and not-exhaustive approach followed by many authors; hence, a much more useful approach would be to summarize the LCI information (models used, assumptions made and references) in a table, enabling the reader to focus on useful information. If space is limited, authors could take into account the possibility of using supplementary materials/annexes. Examples of good practice in reporting LCI data were found in Bartl et al. (2011), Bava et al. (2014) or Zehetmeier et al. (2014b).

3.3.9 Impact assessment (LCIA)

For each impact category, the calculation of the potential environmental impacts is based on characterization models describing the environmental mechanism that links the inventory data to an indicator. Models can be referred to global or regional scale. They define the characterization factors that weigh all the substances contributing to a certain impact category, and refer them to the selected indicator. According to their "positioning" in the environmental mechanism, two kinds of indicators could be identified: endpoint (also called damage oriented) and midpoint (also called problem oriented) indicators.

Usually, most of the LCA software includes different characterization methods, which consist of a list of impact categories, each one associated with a characterization model giving the characterization factors. The characterization methods could include or not the optional elements of LCIA (normalization, grouping, weighting and data quality analysis).

The ILCD has prepared several Handbooks on the LCIA topic, proving that the selection of the method is an extremely important issue (IES, 2010c, 2010d, 2011, 2012). Among these documents, in the volume titled "Recommendations for LCIA in the European context", the Institute for Environmental Sustainability of JRC (JRC-IES) analyzed and compared the existing characterization methods, selecting a shortlist of "current best practice" for each impact category (IES, 2012). The rich literature on LCIA methods proves that the debate on LCIA is still open. Transparency in reporting the LCIA method is essential, considering the potentially important influence that their selection may have on the results.

In milk LCA, the information about the LCIA method used is not uniformly reported: some authors report the method, overlooking the model and characterization factors, while others report the model chosen and the characterization factors applied, implying the method. The correspondence between

these two kinds of information (method or model) could only be reconstructed with difficulty, since in some cases it was not univocal (different methods could use the same model) (PRé, 2014). Furthermore, in some cases different methods were chosen to calculate the impacts associated with different impact categories.
This lack of homogeneity complicates the reconstruction of statistics regarding the application of LCIA methods.
In Table 3.4 an overview of the main methods encountered in revised LCA studies is given.

Table 3.4. LCIA methods used in the selected studies.

Methods	**# of studies**
CML	10
CML+CED	5
CML+CED+EDIP97	1
EPD	3
ReCiPe	2
ILCD	1
Other methods	2
IPCC*	4
Not specified	1

* Even if IPCC is not a method for LCIA, it is used in some papers as reference without specifying any other method.

Regarding the selected studies, CML is the most widely adopted method (16 cases considering all rows in Table 3.4 containing CML). Furthermore the method suggested to present an Environmental Product Declaration (EPD) could be considered as a restriction of the CML method, since all the impact categories of EPD are directly taken from the CML-baseline method (PRé, 2014), increasing the number of studies applying the CML method. The more recent ReCiPe method is less used. The ReCiPe method takes its origins from CML and Ecoindicator (respectively the version CML 2001 and Ecoindicator 99) and represents the attempt to address the need to join the problem oriented and damage oriented models in a consistent framework to combine the advantages of both concepts. It is interesting to note that just one author (Battini et al., 2014) takes into account the methods recommended by ILCD (IES, 2012), advising that efforts made in order to achieve a higher level of standardization were poorly acknowledged by practitioners.
PRé (2014) gives a brief description of the main methods developed for the LCIA.

Afterwards, the models and the characterization factors used in each paper were identified. The potential impact on global warming was without exception evaluated in line with the IPCC guidelines, even if different characterization factors were observed according to the version adopted.

For the acidification potential, the RAIN-LCA model developed for the European Countries (Huijbregts, 1999) was the most frequently applied, even if some authors referred to characterization factors proposed by Heijungs et al. (1992) (the first version of CML method).

Concerning the eutrophication potential, the majority of authors generically referred to Guinée et al. (2002) (the second version of the CML method). Also in this case, there was a minority of authors who still used the previous version of the CML method, referring to Heijungs et al. (1992).

Since there is a large level of coherence in the choice of LCIA method as observed also by Notarnicola et al. (2015), this variable is not further evaluated in the statistical analysis.

3.3.10 Quantitative results

For a quantitative evaluation of the LCA results the impact categories most widely evaluated (GWP, AP, EP, EU, LU, see Table 3.1) were selected. Only the values reported with the most common indicators were retained (kg CO_2eq for GWP, kg SO_2eq for AP, kg PO_4eq for EP, MJ for EU, $m^2 \cdot year^{-1}$ for LU). Results of studies with extended boundaries were reduced to farm gate when possible. If a range of values was reported, the worst scenario (the higher value) was considered for the analysis. The GWP values obtained by Bartl et al. (2011) were not used, since they were calculated using a 20 years' timeframe, instead of 100 years.

The influence that different FUs and allocation methods could have on the impact estimations founded in literature was analyzed. As previously mentioned, the model used for the LCA was not considered since a great coherence was found in the selected studies.

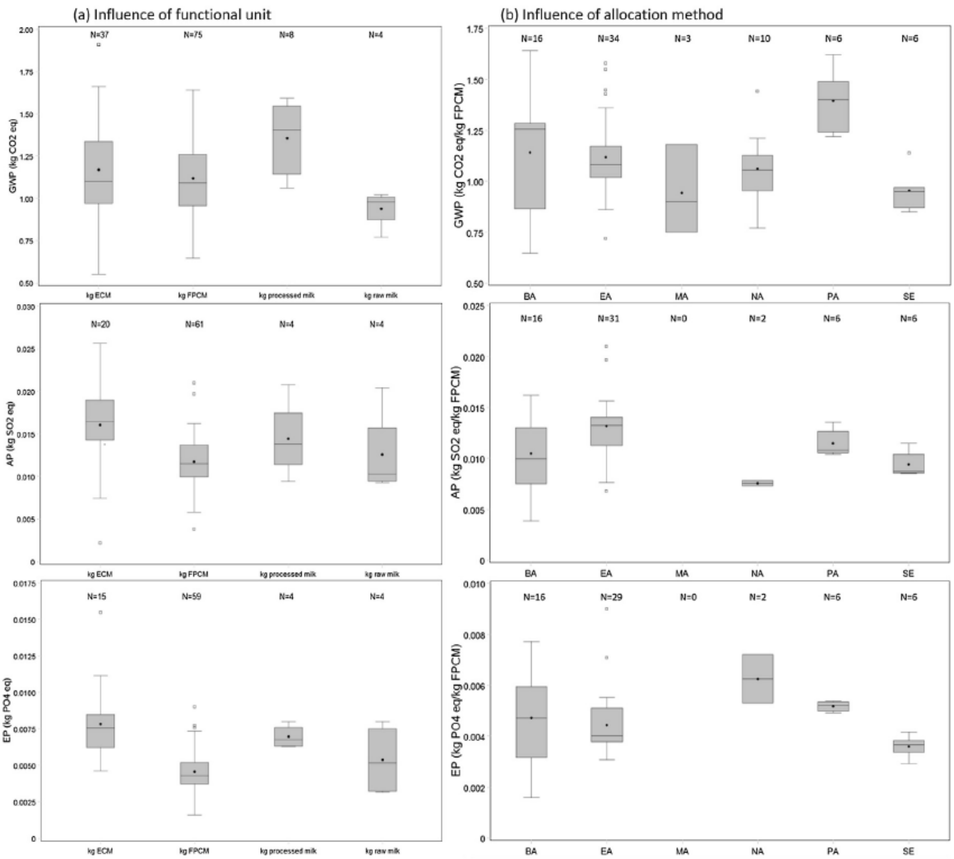
The units of measurement selected for the analysis of the functional units were kg ECM, kg FPCM, kg of raw milk and kg of processed milk. The processed milk included HQ milk, UHT milk and pasteurized milk. An average density of $1.03 \cdot 10^3$ kg m^{-3} was used to convert volume to mass of milk.

The analysis of the allocation methods was instead conducted only with results expressed per kg of FPCM, since that was the largest sample of data. The considered allocation methods are biological allocation (BA), economic allocation (EA), mass allocation (MA), no allocation (NA), protein allocation (PA) and system expansion (SE).

Figure 3.5 reports the distribution of values of considered impact categories, ordered according to the functional unit or to the allocation method used in each study, while results of the statistical analysis are reported in Table 3.5 and Table 3.6.

Table 3.5. Results of Kruskal-Wallis test referred to the influence of functional unit. "-" means no available data.

		Kg FPCM	Kg processed	Kg raw milk
GWP	kg ECM	0.473	0.050	0.079
	kg FPCM		0.009	0.094
	kg processed milk			0.007
AP	kg ECM	<0.001	0.394	0.278
	kg FPCM		0.24	0.743
	kg processed milk			0.387
EP	kg ECM	<0.001	0.617	0.162
	kg FPCM		0.006	0.822
	kg processed milk			0.468
EU	kg ECM	0.55	0.009	0.122
	kg FPCM		0.005	0.109
	kg processed milk			0.049
LU	kg ECM	<0.001	-	0.297
	kg FPCM		-	0.112
	kg processed milk			-


(a) Influence of functional unit
(b) Influence of allocation method
GWP (kg CO2 eq)
AP (kg SO2 eq)
EP (kg PO4 eq)
GWP (kg CO2 eq/kg FPCM)
AP (kg SO2 eq/kg FPCM)
EP (kg PO4 eq/kg FPCM)
N=37
N=75
N=8
N=4
N=20
N=61
N=4
N=4
N=15
N=59
N=4
N=4
kg ECM
kg FPCM
kg processed milk
kg raw milk
N=16
N=34
N=3
N=10
N=6
N=6
N=16
N=31
N=0
N=2
N=6
N=6
N=16
N=29
N=0
N=2
N=6
N=6
BA
EA
MA
NA
PA
SE

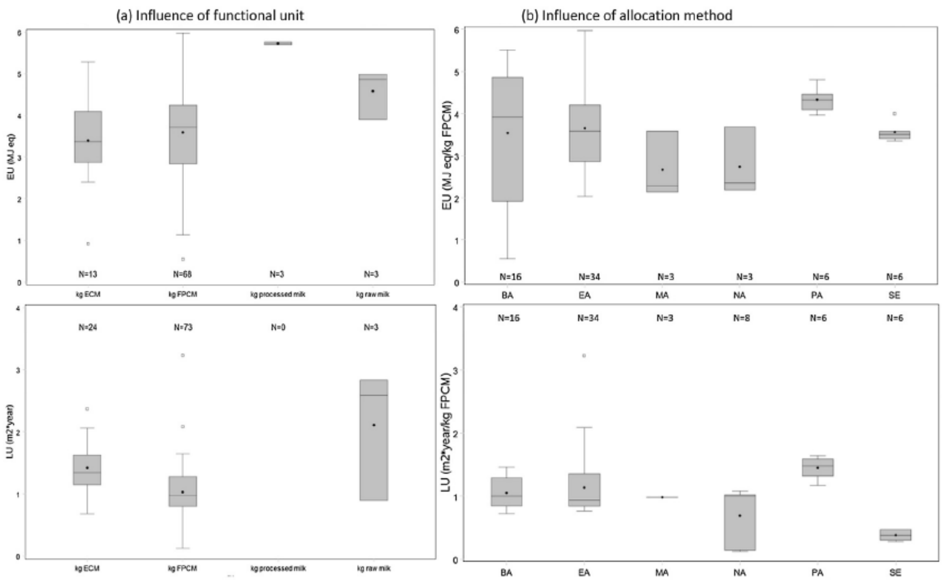


Figure 3.5. (a) Distribution of the results sorted by functional unit; kg ECM: kg of Energy Corrected milk ; kg FPCM: kg of Fat and Protein Corrected Milk; kg processed milk: includes kg of HQ milk, UHT milk and pasteurized milk. (b) Distribution of the results expressed per kg FPCM, sorted by allocation rule selected by the authors; BA: Biological Allocation; EA: Economic Allocation; MA: Mass Allocation; NA: No Allocation; PA: Protein Allocation; SE: System Expansion. In both cases upper and lower sides of boxes: upper and lower quartiles; tiles: extreme values; line through the box: median; •: mean; □: outliers over 1.5 box lengths from the upper or lower edge of the box; N: number of cases in the sample.

Table 3.6. Results of Kruskal-Wallis test referred to the influence of allocation method; BA: Biological Allocation; EA: Economic Allocation; MA: Mass Allocation; NA: No Allocation; PA: Protein Allocation; SE: System Expansion. "-" means no available data.

		EA	MA	NA	PA	SE
GWP	BA	0.371	0.179	0.246	0.077	0.14
	EA		0.266	0.425	0.004	0.022
	MA			0.311	0.02	0.796
	NA				0.005	0.193
	PA					0.004
AP	BA	0.02	-	0.261	0.376	0.507
	EA		-	0.035	0.027	0.002
	MA			-	-	-
	NA				0.045	0.045
	PA					0.025
EP	BA	0.569	-	0.206	0.113	0.122
	EA		-	0.064	0.027	0.063
	MA			-	-	-
	NA				0.182	0.045
	PA					0.004
EU	BA	0.884	0.219	0.263	0.14	0.376
	EA		0.084	0.113	0.049	0.909
	MA			0.513	0.020	0.302
	NA				0.020	0.302
	PA					0.007
LU	BA	0.893	1.000	0.177	0.004	< 0.001
	EA		0.867	0.378	0.017	< 0.001
	MA			0.533	0.018	0.018
	NA				0.002	0.436
	PA					0.004

As has already emerged in §3.3.7 and 3.3.8, the sample size variations among groups of observations are substantial, since kg of FPCM is the most used functional unit and the EA is the most common allocation method applied. This could limit the power of the data analysis, emphasizing particular cases, neverthless some interesting observations emerged.

The high level of coherence of the GHG estimations leads to a great convergence in the forecast GWP, also when comparing the main functional units (FPCM and ECM). On the other hand, for AP, EP and LU the choice to express the results as kg of FPCM or as kg of ECM resulted in a statistically different outcome (see Table 3.5). Regarding EU, the higher values obtained using the kg of processed milk as the functional unit, could be ascribed to an overestimation of the energy due to the enlargement of system boundaries.

Concerning allocation rules, the most commonly adopted allocation methods (BA and EA) were quite interchangeable (except for AP, no statistically significant differences were observed – see Table 3.6). Otherwise, PA and SE were more "sensitive" methods, confirming the conclusions reported by Kristensen et al. (2011) and Nguyen et al. (2013b) who suggested using more moderate options for allocation, namely biological (BA) or economic (EA) allocation.

3.3.11 Uncertainty and sensitivity analysis

Each phase of an LCA assessment has an associated uncertainty that should be quantified in order to increase the transparency of LCA data and results. In the past, a lack of standardization was observed even in the terminology. A clear definition of sensitivity analysis and uncertainty analysis can now be found in LEAP (2016).

The sensitivity analysis usually refers to a systematic variation of input parameters, to determine how sensitive the outputs are to each input. This is not a complete uncertainty propagation procedure, but it is useful to distinguish, among the input parameters, which is the most important in affecting the final results (Baker and Lepeh, 2009). On the other hand, Monte Carlo Simulation (MCS) is the preferred approach to propagate uncertainty throughout LCA models. It generates thousands of random samplings of the input data, resulting in a probabilistic distribution of the predicted impacts (Chen and Corson, 2014).

Among the reviewed studies, only a minority of authors carried out a sensitivity or uncertainty analysis. (the complete list is reported in Table 3.7). However, the standardization of the procedure for the uncertainty quantification and its systematic inclusion in the LCA studies should be promoted and recommended.

3.3.12 General discussion

Harmonization among LCA studies applied to the milk sector still represents a goal to be achieved. This clearly emerged from the in-depth analysis carried out in the present review.

Difficulties in reaching a shared approach emerged by analyzing the citation of reference standards, not always considered by the authors, or by observing that only one author followed the method for LCIA proposed by ILCD.

Table 3.7. Reviewed studies in which a sensitivity or uncertainty analysis of the results was conducted.

Study	Object of comparison
Battini et al. (2014)	Sensitivity analysis to evaluate the influence of: 1. storage emissions; 2. including LUC due to soybean meal.
Chen and Corson (2014)	Monte Carlo Simulation to test the variability of farm characteristics.
Ross et al. (2014)	Sensitivity analysis and Monte Carlo Simulation to assess the effect of IPCC coefficients and the system-specific emission factors.
Sasu-Boakye et al. (2014)	Sensitivity analysis in order to verify the influence of: 1. allocation factors of co-products (crop); 2. N_2O emissions; 3. enteric CH_4 emissions.
Guerci et al. (2013)	Simplified sensitivity analysis incorporating the direct land use change for soybean production.
Roer et al. (2013)	Sensitivity analysis in order to evaluate the consequence of: 1. changing the base case of different emission factors of 50%; 2. radical changes (50%) in diesel consumption; 3. changes in electricity mix.
O'Brien et al. (2012)	Sensitivity analysis comparing different scenarios: 1. considering carbon sequestration from grassland, 2. changing allocation rules; 3. substituting soybean meal with a similar protein meal; 4. using real data to estimate cows enteric fermentations; 5. using different emission factors for manure storage; 6. using default values for estimating N_2O.
Bartl et al. (2011)	Sensitivity analysis comparing alternative allocation methods and different LCIA methods.
Arsenault et al. (2009)	Sensitivity analysis testing the assumption made: 1. pasture system producing less during summer; 2. reducing grazing period.

A controversial point is the definition of a common FU. The choice of expressing the environmental output per kilogram of product or per hectare of land may alter conclusions, favoring one or another production system (Bartl et al., 2011; Garnett, 2014; Weiler et al., 2014). Moreover, a single metric does not fully describe the several outputs delivered from dairy farms and could be an inadequate measure of their environmental impacts (Garnett, 2014). This observation opened an interesting debate, still unresolved, on which is the most appropriate FU(s) to adopt among LCA practitioners.

Regarding co-product handling, there is still no convergence on “the best method” to be used, even if more moderate options (i.e. biological or economic allocation) were recognized as the most suitable choices (Rotz et al., 2010; O'Brien et al., 2014). It is interesting to note that the solution most commonly adopted (i.e. economic allocation) is not compliant with the ranking criteria defined by ISO, which prioritize avoidance of allocation or applying system expansion. On the contrary, the LEAP guidelines narrow the options available for allocation, excluding the use of system expansion by means of substitution when applying the attributional model. This seeming contradiction underlines the actual complexity of the co-product handling in the livestock sector. As suggested by Zehetmeier et al. (2014b), it is impossible to determine the "correct" allocation method but more attention should be paid in establishing the criteria to choose a particular allocation scheme. Nevertheless, a coherent and systematic application of multi-functionality solutions could be achieved only with a clear definition of the nature and purpose of LCA (IES, 2010a; Pelletier et al., 2015), since this choice should be closely related to them.

Contrarily to FU and allocation, a great level of convergence was observed regarding the LCIA method (CML was the most commonly used). To elaborate on this topic is quite complex. In the literature, the choice of the LCIA method is never justified and its citation is often difficult to trace. The LCA practitioners should consider whether this high level of coherence depends more on a scientific consensus or on a habit.

Another common issue among the papers reviewed is the problematic quantification of the non-monetary values of the herd. This is particularly clear for the smallholder systems in developing countries, where the productiveness and the consumer choices are not the only market rules, but also draught power and the capital assets provided by animals should be considered (Weiler et al., 2014). For this reason, the creation of alternative FUs and allocation methods are recommended.

Due to the high influence that the practitioners' choices have on the final results, there is evidently a need to systematically conduct a sensitivity analysis, in order to better understand the "choice-related" problem of milk LCA. This practice could help to understand the reliability of the estimated environmental burdens and to quantify the consequences of decisions and methodologies applied. These scenario analyses were already conducted studying the influence of the allocation method and, partially, of the FU. It would be interesting to apply this approach also to other potentially influencing parameters (e.g. databases used for LCI, LCIA applied methods).

3.4 Conclusions

The great potential of LCA as a decision tool could be strongly increased by reaching a broader level of standardization. This review highlights the difficulties encountered when comparing milk LCA studies, underlining the importance of practitioners' choices in determining the results. Furthermore, statistical analyses were conducted in order to quantify the relevance of choices (functional unit and allocation rules) on the final results.

LCAs should embrace a broad range of impact categories, limiting the shifting of the targeted environmental problems. Currently, global warming potential, acidification, eutrophication and energy use are the most frequently evaluated impact categories, while hotspots that need an in-depth analysis are land use change, biodiversity, ecotoxicity and water use.

More consistent results could be achieved through the definition of a common FU (such as FPCM, as recommended by IDF), that would allow a direct comparison of the results of different studies, although with different assumptions.

Future LCAs should also give a sufficiently detailed description of the system boundaries, accompanied by a flow diagram, in order to help the reader to promptly find out the main information about the considered input and output. The methods used for the calculation of the potential impacts should be made explicit in the text, improving transparency of the study. If possible, selected emission factors should be site-specific and a table resuming the equations used for their calculation would be appreciated.

Finally, the sensitivity analysis should be systematically conducted and the uncertainties associated with the selected input data should be quantified, since the choices made by the practitioners have an important effect on the final result.

References

Arsenault, N., Tyedmers, P., Fredeen, A., 2009. Comparing the environmental impacts of pasture-based and confinement-based dairy systems in Nova Scotia (Canada) using life cycle assessment. Int. J. Agric. Sustain. 7, 19-41.

Arvanitoyannis, I.S., Kotsanopoulos, K.V., Veikou, A., 2014. Life Cycle Assessment (ISO 14040) Implementation in Foods of Animal and Plant Origin: Review. Crit. Rev. Food Sci. Nutr. 54, 1253-1282.

Baker, J.W., Lepeh, M.D., 2009. Treatment of uncertainties in life cycle assessment. 10th International Conference on Structural Safety and Reliability, Osaka, Japan.

Bartl, K., Gómez, C.A., Nemecek, T., 2011. Life cycle assessment of milk produced in two smallholder dairy systems in the highlands and the coast of Peru. J. Clean. Prod. 19, 1494-1505.

Basset-Mens, C., Ledgard, S., Boyes, M., 2009. Eco-efficiency of intensification scenarios for milk production in New Zealand. Ecol. Econ. 68, 1615-1625.

Battini, F., Agostini, A., Boulamanti, A.K., Giuntoli, J., Amaducci, S., 2014. Mitigating the environmental impacts of milk production via anaerobic digestion of manure: Case study of a dairy farm in the Po Valley. Sci. Total Environ. 481, 196-208.

Bava, L., Sandrucci, A., Zucali, M., Guerci, M., Tamburini, A., 2014. How can farming intensification affect the environmental impact of milk production? J. Dairy Sci. 97, 4579-4593.

Brandão, M., Heath, G., Cooper, J., 2012. What Can Meta-Analyses Tell Us About the Reliability of Life Cycle Assessment for Decision Support? J. Ind. Ecol. 16, S3-S7.

BSI, 2008. Specification for the assessment of life cycle greenhouse gas emissions of goods and services. British Standard Institute, London, UK.

Castanheira, É., Dias, A.C., Arroja, L., Amaro, R., 2010. The environmental performance of milk production on a typical Portuguese dairy farm. Agr. Syst. 103, 498-507.

Cederberg, C., Henriksson, M., Berglund, M., 2013. An LCA researcher's wish list--data and emission models needed to improve LCA studies of animal production. Animal 7 Suppl 2, 212-219.

Cederberg, C., Stadig, M., 2003. System Expansion and Allocation in Life Cycle Assessment of Milk and Beef Production. Int. J. Life Cycle Ass. 8, 350-356.

Chen, X., Corson, M.S., 2014. Influence of emission-factor uncertainty and farm-characteristic variability in LCA estimates of environmental impacts of French dairy farms. J. Clean. Prod. 81, 150-157.

Crosson, P., Shalloo, L., O'Brien, D., Lanigan, G.J., Foley, P.A., Boland, T.M., Kenny, D.A., 2011. A review of whole farm systems models of greenhouse gas emissions from beef and dairy cattle production systems. Anim. Feed Sci. Technol. 166-167, 29-45.

Čuček, L., Klemeš, J.J., Kravanja, Z., 2012. A review of footprint analysis tools for monitoring impacts on sustainability. J. Clean. Prod. 34, 9-20.

Dalgaard, R., Schmidt, J., Flysjö, A., 2014. Generic model for calculating carbon footprint of milk using four different life cycle assessment modelling approaches. J. Clean. Prod. 73, 146-153.

de Vries, M., de Boer, I.J.M., 2010. Comparing environmental impacts for livestock products: A review of life cycle assessments. Livest. Sci. 128, 1-11.

Djekic, I., Miocinovic, J., Tomasevic, I., Smigic, N., Tomic, N., 2014. Environmental life-cycle assessment of various dairy products. J. Clean. Prod. 68, 64-72.

EPLCA, 2015. European Platform on Life Cycle Assessment. Available at http://ec.europa.eu/environment/ipp/lca.htm (accessed 27.05.2015).

Fantin, V., Buttol, P., Pergreffi, R., Masoni, P., 2012. Life cycle assessment of Italian high quality milk production. A comparison with an EPD study. J. Clean. Prod. 28, 150-159.

Fantin, V., Scalbi, S., Ottaviano, G., Masoni, P., 2014. A method for improving reliability and relevance of LCA reviews: The case of life-cycle greenhouse gas emissions of tap and bottled water. Sci. Total Environ. 476-477, 228-241.

FAO, 2010. Greenhouse Gas Emissions from the Dairy Sector: A Life Cycle Assessment. Food and Agriculture Organization of the United Nations, Rome; 2010.

Flysjö, A., Cederberg, C., Henriksson, M., Ledgard, S., 2011. How does co-product handling affect the carbon footprint of milk? Case study of milk production in New Zealand and Sweden. Int. J. Life Cycle Ass. 16, 420-430.

Flysjö, A., Cederberg, C., Henriksson, M., Ledgard, S., 2012. The interaction between milk and beef production and emissions from land use change - Critical considerations in life cycle assessment and carbon footprint studies of milk. J. Clean. Prod. 28, 134-142.

Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., Zaks, D.P.M., 2011. Solutions for a cultivated planet. Nature 478, 337-342.

Garnett, T., 2009. Livestock-related greenhouse gas emissions: impacts and options for policy makers. Environ. Sci. Policy 12, 491-503.

Garnett, T., 2014. Three perspectives on sustainable food security: Efficiency, demand restraint, food system transformation. What role for life cycle assessment? J. Clean. Prod. 73, 10-18.

Gerber, P., Vellinga, T., Opio, C., Steinfeld, H., 2011. Productivity gains and greenhouse gas emissions intensity in dairy systems. Livest. Sci. 139, 100-108.

González-García, S., Castanheira, T.G., Dias, A.C., Arroja, L., 2013. Using Life Cycle Assessment methodology to assess UHT milk production in Portugal. Sci. Total Environ. 442, 225-234.

Guerci, M., Bava, L., Zucali, M., Sandrucci, A., Penati, C., Tamburini, A., 2013a. Effect of farming strategies on environmental impact of intensive dairy farms in Italy. J. Dairy Res. 80, 300-308.

Guerci, M., Knudsen, M.T., Bava, L., Zucali, M., Schönbach, P., Kristensen, T., 2013b. Parameters affecting the environmental impact of a range of dairy farming systems in Denmark, Germany and Italy. J. Clean. Prod. 54, 133-141.

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H.A., de Bruijn, H., van Duin, R., Huijbregts, M.A.J., Lindeijer, E., Roorda, A.A.H., van der Ven, B.L., Weidema, B.P.,2002. Handbook on Life Cycle Assessment; Operational Guide to the ISO Standards. Institute for Environmental Sciences, Leiden University, The Netherlands.

Heijungs, R., Guinée, J.B., Huppes, G., Lankreijer, R.M., Udo de Haes, H.A., Wegener Sleeswijk, A.,1992. Environmental life cycle assessment of products: guide and backgrounds (Part 1). Center of Environmental Science, Leiden University, the Netherlands.

Heller, M.C., Keoleian, G.A., 2011. Life cycle energy and greenhouse gas analysis of a large-scale vertically integrated organic dairy in the United States. Environ. Sci. Technol. 45, 1903-1910.

Hellweg, S., Canals, L.M.I., 2014. Emerging approaches, challenges and opportunities in life cycle assessment. Science 344, 1109-1113.

Hristov, A., Hanigan, M., Cole, A., Todd, R., McAllister, T., Ndegwa, P., Rotz, A., 2011. Review: ammonia emissions from dairy farms and beef feedlots 1. Canadian journal of animal science 91, 1-35.

Huijbregts, M.,1999. Life-cycle impact assessment of acidifying and eutrophying air pollutants-Calculation of equivalency factors with RAINS-LCA. Interfaculty Department of Environmental Science, University of Amsterdam, Amsterdam, the Netherlands.

IDF, 2010. A common carbon footprint approach for dairy - The IDF guide to standard life cycle assessment methodology for dairy sector. International Dairy Federation.

IES (Institute for Environmental Sustainability), European Commission - Joint Research Center, 2010a. International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. First Edition March 2010. EUR 24708 EN. Publications Office of the European Union, Luxemburg.

IES (Institute for Environment and Sustainability), European Commission - Joint Research Centre, 2010b. International Reference Life Cycle Data System (ILCD) Handbook - Specific guide for Life Cycle Inventory data sets. First Edition March 2010. EUR 24709 EN. Publications Office of the European Union, Luxemburg.

IES (Institute for Environment and Sustainability), European Commission - Joint Research Centre, 2010c. International Reference Life Cycle Data System (ILCD) Handbook - Analysis of existing Environmental Impact Assessment methodologies for use in Life Cycle Assessment. First Edition. Publications Office of the European Union, Luxemburg.

IES (Sustainability, I.f.E.a.), European Commission - Joint Research Centre, 2010d. International Reference Life Cycle Data System (ILCD) Handbook - Framework and Requirements for Life Cycle Impact Assessment Models and Indicators. First Edition March 2010. EUR 24586 EN. Publications Office of the European Union, Luxemburg.

IES (Institute for Environment and Sustainability), European Commission - Joint Research Centre, 2011. International Reference Life Cycle Data System (ILCD) Handbook - Recommendations for Life Cycle Impact Assessment in the European context. First Edition November 2011. EUR 24571 EN. Publications Office of the European Union, Luxemburg.

IES (Institute for Environment and Sustainability), European Commission - Joint Research Centre, 2012. Characterisation factors of the ILCD Recommended Life Cycle Impact Assessment methods. Database and Supporting Information. First Edition February 2012. EUR 25167. Publications Office of the European Union, Luxemburg.

Iribarren, D., Hospido, A., Moreira, M.T., Feijoo, G., 2011. Benchmarking environmental and operational parameters through eco-efficiency criteria for dairy farms. Sci. Total Environ. 409, 1786-1798.

ISO, 2006a. Environmental Management - Life Cycle Assessment-Principles and Framework, EN ISO 14040:2006. International Organization for Standardization, Geneva, Switzerland.

ISO, 2006b. Environmental Management - Life Cycle Assessment-Requirements and Guidelines, EN ISO 14044:2006. International Organization for Standardization, Geneva, Switzerland.

Jan, P., Dux, D., Lips, M., Alig, M., Dumondel, M., 2012. On the link between economic and environmental performance of Swiss dairy farms of the alpine area. Int. J. Life Cycle Ass. 17, 706-719.

Kristensen, T., Mogensen, L., Knudsen, M.T., Hermansen, J.E., 2011. Effect of production system and farming strategy on greenhouse gas emissions from commercial dairy farms in a life cycle approach. Livest. Sci. 140, 136-148.

LEAP,2016. Environmental Performance of Large Ruminant Supply Chains: Guidelines for Assessment. Version 1. Livestock Environmental Assessment and Performance (LEAP) Partnership, FAO, Rome, Italy.

Lifset, R., 2012. Toward Meta-Analysis in Life Cycle Assessment. J. Ind. Ecol. 16, S1-S2.

Mc Geough, E.J., Little, S.M., Janzen, H.H., McAllister, T.A., McGinn, S.M., Beauchemin, K.A., 2012. Life-cycle assessment of greenhouse gas emissions from dairy production in Eastern Canada: A case study. J. Dairy Sci. 95, 5164-5175.

Meier, M.S., Stoessel, F., Jungbluth, N., Juraske, R., Schader, C., Stolze, M., 2015. Environmental impacts of organic and conventional agricultural products – Are the differences captured by life cycle assessment? J. Environ. Manage. 149, 193-208.

Meul, M., Van Middelaar, C.E., de Boer, I.J.M., Van Passel, S., Fremaut, D., Haesaert, G., 2014. Potential of life cycle assessment to support environmental decision making at commercial dairy farms. Agr. Syst. 131, 105-115.

Nguyen, T.T.H., Corson, M.S., Doreau, M., Eugène, M., Van Der Werf, H.M.G., 2013a. Consequential LCA of switching from maize silage-based to grass-based dairy systems. Int. J. Life Cycle Ass. 18, 1470-1484.

Nguyen, T.T.H., Doreau, M., Corson, M.S., Eugène, M., Delaby, L., Chesneau, G., Gallard, Y., Van Der Werf, H.M.G., 2013b. Effect of dairy production system, breed and co-product handling methods on environmental impacts at farm level. J. Environ. Manage. 120, 127-137.

Nijdam, D., Rood, T., Westhoek, H., 2012. The price of protein: Review of land use and carbon footprints from life cycle assessments of animal food products and their substitutes. Food Policy 37, 760-770.

Notarnicola, B., Hayashi, K., Curran, M.A., Huisingh, D., 2012. Progress in working towards a more sustainable agri-food industry. J. Clean. Prod. 28, 1-8.

Notarnicola, B., Salomone, R., Petti, L., Renzulli, P.A., Roma, R., Cerutti, A.K.,2015. Life Cycle Assessment in the Agri-food Sector: Case Studies, Methodological Issues and Best Practices. Springer.

O'Brien, D., Capper, J.L., Garnsworthy, P.C., Grainger, C., Shalloo, L., 2014. A case study of the carbon footprint of milk from high-performing confinement and grass-based dairy farms. J. Dairy Sci. 97, 1835-1851.

O'Brien, D., Shalloo, L., Patton, J., Buckley, F., Grainger, C., Wallace, M., 2012a. Evaluation of the effect of accounting method, IPCC v. LCA, on grass-based and confinement dairy systems' greenhouse gas emissions. Animal 6, 1512-1527.

O'Brien, D., Shalloo, L., Patton, J., Buckley, F., Grainger, C., Wallace, M., 2012b. A life cycle assessment of seasonal grass-based and confinement dairy farms. Agr. Syst. 107, 33-46.

Pelletier, N., Ardente, F., Brandão, M., De Camillis, C., Pennington, D., 2015. Rationales for and limitations of preferred solutions for multi-functionality problems in LCA: is increased consistency possible? Int. J. Life Cycle Ass. 20, 74-86.

Penati, C.A., Tamburini, A., Bava, L., Zucali, M., Sandrucci, A., 2013. Environmental impact of cow milk production in the central Italian Alps using Life Cycle Assessment. Ital. J. Anim. Sci. 12, 584-592.

Pirlo, G., Carè, S., 2013. A simplified tool for estimating carbon footprint of dairy cattle milk. Ital. J. Anim. Sci. 12, 497-506.

PRé, 2014. SimaPro Database Manual Methods library v2.7. Available at http://www.pre-sustainability.com/download/DatabaseManualMethods.pdf (accessed 18/03/2015).

Ridoutt, B., Fantke, P., Pfister, S., Bare, J., Boulay, A.M., Cherubini, F., Frischknecht, R., Hauschild, M., Hellweg, S., Henderson, A., Jolliet, O., Levasseur, A., Margni, M., McKone, T., Michelsen, O., Milà I Canals, L., Page, G., Pant, R., Raugei, M., Sala, S., Saouter, E., Verones, F., Wiedmann, T., 2015. Making sense of the minefield of footprint indicators. Environ. Sci. Technol. 49, 2601-2603.

Roer, A.G., Johansen, A., Bakken, A.K., Daugstad, K., Fystro, G., Strømman, A.H., 2013. Environmental impacts of combined milk and meat production in Norway according to a life cycle assessment with expanded system boundaries. Livest. Sci. 155, 384-396.

Ross, S.A., Chagunda, M.G.G., Topp, C.F.E., Ennos, R., 2014. Effect of cattle genotype and feeding regime on greenhouse gas emissions intensity in high producing dairy cows. Livest. Sci. 170, 158-171.

Roy, P., Nei, D., Orikasa, T., Xu, Q., Okadome, H., Nakamura, N., Shiina, T., 2009. A review of life cycle assessment (LCA) on some food products. J. Food Eng. 90, 1-10.

Sasu-Boakye, Y., Cederberg, C., Wirsenius, S., 2014. Localising livestock protein feed production and the impact on land use and greenhouse gas emissions. Animal 8, 1339-1348.

Sjaunja, L.O., Baevre, L., Junkkarinen, L., Pedersen, J., Setälä, J., 1990. A Nordic proposal for an energy corrected milk (ECM) formula. In the 27th session of the International Commission for Breeding and Productivity of Milk Animals, Paris.

Soussana, J.F., 2014. Research priorities for sustainable agri-food systems and life cycle assessment. J. Clean. Prod. 73, 19-23.

Sutton, M.A., Howard, C.M., Bleeker, A., Datta, A., 2013. The global nutrient challenge: From science to public engagement. Environmental Development 6, 80-85.

Teixeira, R.F.M., 2015. Critical Appraisal of Life Cycle Impact Assessment Databases for Agri-food Materials. J. Ind. Ecol. 19, 38-50.

Thoma, G., Jolliet, O., Wang, Y., 2013. A biophysical approach to allocation of life cycle environmental burdens for fluid milk supply chain analysis. Int. Dairy J. 31, S41-S49.

Thomassen, M., Dalgaard, R., Heijungs, R., de Boer, I., 2008. Attributional and consequential LCA of milk production. Int J Life Cycle Assess 13, 339-349.

Thomassen, M.A., Dolman, M.A., van Calker, K.J., de Boer, I.J.M., 2009. Relating life cycle assessment indicators to gross value added for Dutch dairy farms. Ecol. Econ. 68, 2278-2284.

Tuomisto, H.L., Hodge, I.D., Riordan, P., Macdonald, D.W., 2012. Does organic farming reduce environmental impacts? - A meta-analysis of European research. J. Environ. Manage. 112, 309-320.

Van Der Werf, H.M.G., Garnett, T., Corson, M.S., Hayashi, K., Huisingh, D., Cederberg, C., 2014. Towards eco-efficient agriculture and food systems: Theory, praxis and future challenges. J. Clean. Prod. 73, 1-9.

van der Werf, H.M.G., Kanyarushoki, C., Corson, M.S., 2009. An operational method for the evaluation of resource use and environmental impacts of dairy farms by life cycle assessment. J. Environ. Manage. 90, 3643-3652.

Webb, J., Audsley, E., Williams, A., Pearn, K., Chatterton, J., 2014. Can UK livestock production be configured to maintain production while meeting targets to reduce emissions of greenhouse gases and ammonia? J. Clean. Prod. 83, 204-211.

Weidema, B., 2014. Has ISO 14040/44 Failed Its Role as a Standard for Life Cycle Assessment? J. Ind. Ecol. 18, 324-326.

Weidema, B.P., Wesnæs, M., Hermansen, J., Kristensen, T., Halberg, N., Eder, P., Delgado, L., 2008. Environmental Improvement Potentials of Meat and Dairy Products. EUR 23491 EN. JRC Scientific and Technical Reports. Luxemburg; 2008.

Weiler, V., Udo, H.M.J., Viets, T., Crane, T.A., De Boer, I.J.M., 2014. Handling multi-functionality of livestock in a life cycle assessment: The case of smallholder dairying in Kenya. Current Opinion in Environmental Sustainability 8, 29-38.

Wolf, M.A., Pant, R., Chomkhamsri, K., Sala, S., Pennington, D., 2012. The International Reference Life Cycle Data System (ILCD) Handbook - Towards more sustainable production and consumption for a resource-efficient Europe. JRC66506. Institute for Environment and Sustainability-Joint Research Center. Publications Office of the European Union, Luxemburg; 2012.

Yan, M.J., Humphreys, J., Holden, N.M., 2011. An evaluation of life cycle assessment of European milk production. J. Environ. Manage. 92, 372-379.

Yan, M.J., Humphreys, J., Holden, N.M., 2013a. Evaluation of process and input-output-based life-cycle assessment of Irish milk production. J. Agric. Sci. 151, 701-713.

Yan, M.J., Humphreys, J., Holden, N.M., 2013b. Life cycle assessment of milk production from commercial dairy farms: The influence of management tactics. J. Dairy Sci. 96, 4112-4124.

Zehetmeier, M., Gandorfer, M., Hoffmann, H., Müller, U.K., De Boer, I.J.M., Heißenhuber, A., 2014a. The impact of uncertainties on predicted greenhouse gas emissions of dairy cow production systems. J. Clean. Prod. 73, 116-124.

Zehetmeier, M., Hoffmann, H., Sauer, J., Hofmann, G., Dorfner, G., O'Brien, D., 2014b. A dominance analysis of greenhouse gas emissions, beef output and land use of German dairy farms. Agr. Syst. 129, 55-67.

CHAPTER 4

4 Comparison among NH_3 and GHGs emissive patterns from different housing solutions of dairy farms

Baldini C., Borgonovo F., Gardoni D., Guarino M., 2016. Published in Atmospheric Environment; 141: 60-66.

Abstract

Agriculture and livestock farming are known to be activities emitting relevant quantities of atmospheric pollutants. In particular, in intensive animal farming, buildings can be identified as a relevant source of ammonia and greenhouse gases. This study aimed at: i) determining the emission factors of NH_3, N_2O, CH_4, and CO_2 from different dairy farms in Italy, and ii) assessing the effects of the different floor types and manure-handling systems used, in order to minimize the impact of this important productive sector.

A measurement campaign was carried out for 27 months in four naturally ventilated dairy cattle buildings with different floor types, layouts and manure management systems, representative of the most common technologies in the north of Italy. Gas emissions were measured with the "static chamber method": a chamber was placed above the floor farm and an infrared photoacoustic detector (IPD) was used to monitor gas accumulation over time.

In the feeding alleys, emissions of NH_3 were higher from solid floors than from flushing systems and perforated floors. N_2O emissions were significantly different

among farms but the absolute values were relatively low. CH_4 and CO_2 emissions were higher from perforated floors than from other types of housing solution. Regarding the cubicles, the emissions of NH_3 were approximately equal from the two housing solution studied. Contrariwise, N_2O, CH_4 and CO_2 emissions were different between the cubicles with rubber mat and those with straw where the highest values were found.

Keywords: Dairy; Emissions; Ammonia; GHGs; Housing solutions; Manure handling.

4.1 Introduction

Emissions from livestock and agriculture have a great environmental impact and a significant political relevance. Livestock production is an important source of atmospheric pollutants such as ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4). About the 68% of the global anthropogenic ammonia emission is associated with livestock management, while the 65% of nitrous oxide emissions comes from livestock activities, and 35-40% of methane emission comes from enteric fermentation and manure management (Samer, 2013).

Ammonia is an atmospheric pollutant that can cause acidification of soil, nutrient-N enrichment of ecosystems, and eutrophication of terrestrial and aquatic ecosystems (Erisman et al., 2007). In the atmosphere it reacts with other compounds to form ammonium sulfate and ammonium nitrate aerosols, leading to the formation of secondary particulates ($PM_{2.5}$) that are a potential health hazard (Adviento-Borbe et al., 2010; Ansari and Pandis, 1998; Leytem et al., 2013). Moreover, ammonia nitrification and subsequent denitrification in the soil produce N_2O with a consequent effect on global warming (Pereira et al., 2012; Samer, 2013; Wu et al., 2012).

Greenhouse gases (GHGs) emitted from livestock comes mainly from enteric fermentation and manure management. Nitrous oxide (N_2O) and methane (CH_4) contribute to global warming (298 and 34 CO_2-equivalent, respectively), and N_2O is also implicated in the reduction of stratospheric ozone (IPCC, 2013).

Ammonia and greenhouse gases are emitted by excreta deposited over indoor and outdoor surfaces of dairy cattle housing systems (Braam and Swierstra, 1999; Samer, 2013). Several factors influence the emissions of these gases: the diet of the animals, their breed and age, the climatic conditions, the time of day, the building design, the ventilation system, the flooring system and the manure removal system (Philippe et al., 2011; Philippe and Nicks, 2015).

The emission of ammonia is mainly determined by the decomposition of urea, excreted in the urine of the animals. Urea, present in slurries, is first converted to ammonium ion (NH_4^+) that is in equilibrium with the concentration of molecular ammonia (NH_3). The rate of this reaction is defined by the urease enzyme activity. The second step is the volatilization of ammonia which depend on the concentrations of ammonia in the slurry and in the air, and on the transfer processes from the liquid to the gaseous phase (Braam and Swierstra, 1999).

The emission of nitrous oxide (N_2O) is influenced by nitrogen and carbon content of manure, duration of storage, and type of treatment. Nitrous oxide is produced by nitrifiers and denitrifiers microorganisms. The nitrification process takes place under aerobic conditions and leads to the formation of nitrites and nitrates. N_2O is an undesired compound produced during the oxidation of hydroxylamine, an intermediate of this process. In the denitrification phase, nitrites and nitrates are then converted to molecular nitrogen. This reaction occurs under anoxic conditions and could be mediated by denitrifier organisms (Adviento-Borbe et al., 2010; IPCC, 2006). In denitrification, N_2O is an intermediate, which may escape when the rates of N_2O production and consumption differ. The amount of N_2O released from denitrification depends on the absence of molecular O_2 and the presence of NO_3^- and metabolizable organic carbon. Animal production systems create lots of opportunities for partial anaerobiosis, which is suggested to favour denitrification processes (Oenema et al., 2005).

The major source of methane emissions is enteric fermentation of organic material by ruminants as part of their digestive process. Methane (CH_4) is also produced by the decomposition of manure under anaerobic conditions (methanogenesis) during the storage and management of slurries inside and outside of the buildings (Sommer et al., 2007). These emissions are produced more easily in those situation (confined area) where large numbers of animals are bred, such as dairy farms, beef feedlots, swine and poultry farms (IPCC, 2006). Finally, carbon dioxide (CO_2) is generally produced as a result of aerobic decomposition of organic matter of the manure and urea hydrolysis through the reaction of ammonium carbonate [$(NH_4)_2CO_3$] with H^+ ion. Nevertheless, since CO_2 emitted from manure management is considered as natural recycling, the CO_2 emissions do not have practical impact on GHGs emissions from commercial dairy cattle houses.

Relationship among housing solution and related floor type, manure collection and storage systems affect NH_3 and GHGs emission levels (Adviento-Borbe et al., 2010; Wu et al., 2012). Furthermore, among factors influencing the emission levels, the flooring system and the manure removal system are easily overlooked (Cai et al., 2015).

The objectives of the present study were to: i) determine the emission factors of NH_3, N_2O, CH_4, and CO_2 in four different dairy housing solutions, and ii) assess the effect of the floor types and manure-handling system adopted with a direct comparison. Furthermore, different shed components (feeding alley and resting zone) were evaluated, in order to identify the area where each gas is primarily emitted.

4.2 Materials and methods

Measurements were carried out in four naturally ventilated dairy cattle (Holstein Friesian) farms, differing in floors types and manure removal systems. Farms were all located in the Po Valley (north of Italy) and were equipped with the most common building solutions used in this region. Emission data were acquired from different shed areas, namely feeding alleys and resting areas (i.e. cubicles, equipped with straw or rubber mat). The difference between the emissions before and after the cleaning of the surfaces was evaluated in the farms equipped with scrapers. The monitoring campaign was conducted during 9 seasons for an overall period of 27 months. Measurements were carried out in triplicate for a total of 324 sampling points.

The main building characteristics are resumed in Table 4.1.

Table 4.1. Characteristics of the considered farms.

Farm	Number of animals in the barn	Floor type	Manure removal system	Feeding alley (m^2 $animal^{-1}$)	Cubicles	Cubicles (m^2 $animal^{-1}$) Width, m
1	114	perforated floor	periodically removed	5.52	rubber mat	2.7
2	66	concrete floor	delta scraper	5.75	concrete floor and straw	3.3
3	76	concrete floor	flushing system	5.99	ubber mat and solid fraction of slurry	2.5
4	65	rubber mat on concrete floor	delta scraper	5.83	concrete floor and straw	3.3

4.2.1 Farm

Farm 1 was equipped with perforated concrete floor with holes diameter of 3.5 cm. Each section of the housing module consisted of a feeding alley, two rows of head-to-head free stalls, a resting alley, and a row of single free stalls. The manure accumulated in the pit below the slatted surface and periodically removed (approximately every 14 days). Freestalls were equipped/covered with rubber mats and cleaned manually.

Farm 2 had solid concrete floor, provided with several longitudinal grooves to prevent cattle from slipping. The housing building consisted of two specular housing sections with a feeding alley and a row of freestalls, separated by a central feed aisle. The manure was removed five time per day from the feeding alley using delta scrapers. The freestalls were equipped with straw and renewed weekly.

Farm 3 was equipped with a flushing system. The feeding and the resting alley had a convex (1.5% slope) and inclined (3% slope) concrete surface, in order to increase the cleaning efficiency. The flushing was carried out twice a day with a flow rate of 0.15 m^3/s for about ten minutes. The flush system utilized mainly recycled effluent from a screw press solid-liquid separator or occasionally water from the municipal water supply network. The freestalls were equipped with rubber mat and covered with the solid fraction derived from the manure separation system.

Farm 4 had solid floor covered with a rubber mat pavement. Cows were housed in a free-stall barn divided into feeding and resting alley, with two rows of head to head freestalls located between the two areas. Manure was removed with delta scrapers running twice a day. The freestalls were equipped with straw and cleaned weekly.

The diet of the animals was the same in all the four farms for the entire duration of the research. This allows a direct comparison among results. Dry matter supply was 20-30 kg/d, and different feedstuffs were used in order to satisfy the productive needs of cows during seasons. During cold seasons, 26–28 kg of corn silage, 5 kg of alfalfa (Medicago sativa), and 10 kg of concentrate (maize and/or soy flours) with a vitamin supplement per cow per day. During warm seasons, 26–28 kg of corn silage, 3–4 kg of alfalfa (Medicago sativa), and 6 kg of concentrate (cotton and/or sugar beet seeds) per cow per day.

4.2.2 Emission measurements

The emissive fluxes from the different housing solutions were studied according to the "non-steady-state chamber-method" (Brewer and Costello, 1999; Dinuccio et al., 2008; Hornig et al., 1999; Ogink et al., 2013). A closed chamber was placed over the emitting surface, creating a headspace where gas concentration increases over time. The plastic chamber has a truncated pyramidal shape, with the top base surface of 0.096 m^2, lower base surface of 0.147 m^2 and 0.12 m height, defining a volume of 0.017 m^3 above the emitting floor. A small fan was installed inside the chamber, in order to mix the air in the headspace, thus limiting measurement errors due to gas stratification.

The concentration of NH_3, N_2O, CH_4 and CO_2 were measured by means of an Infrared Photoacoustic Detector (IPD; Bruel&Kjaer, multi gas monitor type 1302). The instrument automatically fetches and analyses air samples at regular time intervals (every 2 mins), then it re-enters the sample into the chamber (closed circuit).

The emission factors (mg m^{-2} h^{-1}) were calculated according to the equation (1):

$$Emission\ factor_{gas}\left[\frac{mg}{m^2 \cdot h}\right] = \frac{\delta C\left[\frac{mg}{m^3}\right]}{\delta t[h]} \cdot \frac{V_{ch}[m^3]}{A_{ch}[m^2]} \qquad (1)$$

where $\delta C/\delta t$ is the angular coefficient of the regression line of the linear branch of the gas saturation function, while V_{ch} and A_{ch} are the volume and the lower base of the chamber.

4.2.3 Statistical analysis

Statistical analysis were carried out using SAS statistical software (SAS version 9.3; SAS Institute, Cary, NC, USA, 2012).

The emission factors of each gas were calculated with a linear regression, using as input data the concentrations measured in the chamber at min 0, 2 and 4. The calculated emission factors were then used for the next steps of the statistical analysis, using only values obtained from regression lines with a R^2 higher than 0.9.

The Kolmogorov-Smirnov test was used in order to evaluate the probability density function of emission factors. We used the non-parametric Kruscal Wallis test to compare the emissive pattern of the different farms. The null hypothesis of Kruscal Wallis test is that the medians of all groups are equal, while the alternative hypothesis is that at least one population median of one group is different from the population median of at least one other group. The Steel-Dwass-Critchlow-Fligner test was selected as a post-hoc test for multiple comparisons, in order to verify which median differs from others. Finally a Wilcoxon test was conducted for comparing the difference among emission factors originated from clean or dirty surfaces in Farm 2 and 4.

4.3 Results and discussion

In the following paragraphs, specific emission factors of the considered gases will be presented and discussed. All the relevant results of the statistical analyses are reported in the Supplementary Material (Annex I). The emissions will be referred to the different technologies adopted in the selected farms, in order to allow the evaluation of their environmental performances. For each gas, the Kolmogorov-Smirnov test showed that the probability density functions of the emission factors are non-normal, therefore all the comparisons will be based on the median values of the observations.

4.3.1 NH_3 emissions

The emissions of NH_3 from feeding alleys in all the studied farms ranged from 6.20 to 54.53 mg m^{-2} h^{-1} (Table 4.2). Emission factors were higher in farms equipped with scrapers (Farm 2 and 4), while lower values were observed from perforated floor or flushing system for manure removal. The Kruscal Wallis test showed a significant difference among emission levels of the selected farms ($p<0.0001$) and the post-hoc test (Steel-Dwass-Critchlow-Fligner) identified three groups.

The pavement surface could modulate ammonia emissions. On the one hand, texture and porosity of the floor influence the amount of urine present on the surface after urination, while the contact area between urine and urease present in the feces influences the percentage of urea actually converted into ammonia (Braam and Swierstra, 1999). Thus, NH_3 emissions are expected to be higher in building allowing instantaneous contact of urine and feces (Hristov et al., 2011). These mechanisms could explain the higher NH_3 emissions registered on farms with solid floors, because of the roughness of the surface. Furthermore, scrapers leave a thin layer of slurry increasing the surface area onto which urine is spread and decreasing the thickness of urine pools, thus enhancing N volatilization (Moreira and Satter, 2006). This behavior was observed also by Wu et al. (2012) and Zhang et al. (2005), who reported higher values of NH_3 emission in buildings with solid floor compared to slatted floors. In the pit under perforated floor air circulation is limited, lowering the ammonia emissions. Regarding the flushing, Ogink and Kroodsma (1996) identified this technology as a promising approach to reduce ammonia emissions from barns, throughout the dilution of ammonia concentration of urine pools remaining on the floor. Our results confirmed that emissions of ammonia from a barn equipped with a flushing system are up to one order of magnitude lower if compared to scrapers.

The emissions before and after the cleaning of surfaces were measured in those farms equipped with scraper (Farm 2 and Farm 4). In the case of ammonia this comparison did not show significant differences among clean and dirty surfaces (Figure 4.1A), due to the specific mechanism discussed.

If compared to those observed in the feeding alleys, the NH_3 emissions from cubicles are lower, ranging from 1.72 to 5.10 mg m-2 h-1. Furthermore, the emission levels among farms are not significantly different ($p=0.139$). This implies that the emission variability is not dependent from the chosen technology. Usually a higher amount of urine and feces accumulates in the feeding alleys, justifying the observed trend of emissions. On the other hand, in the resting area the excreted nitrogen can be absorbed by the straw covering the cubicles, lowering its exposure

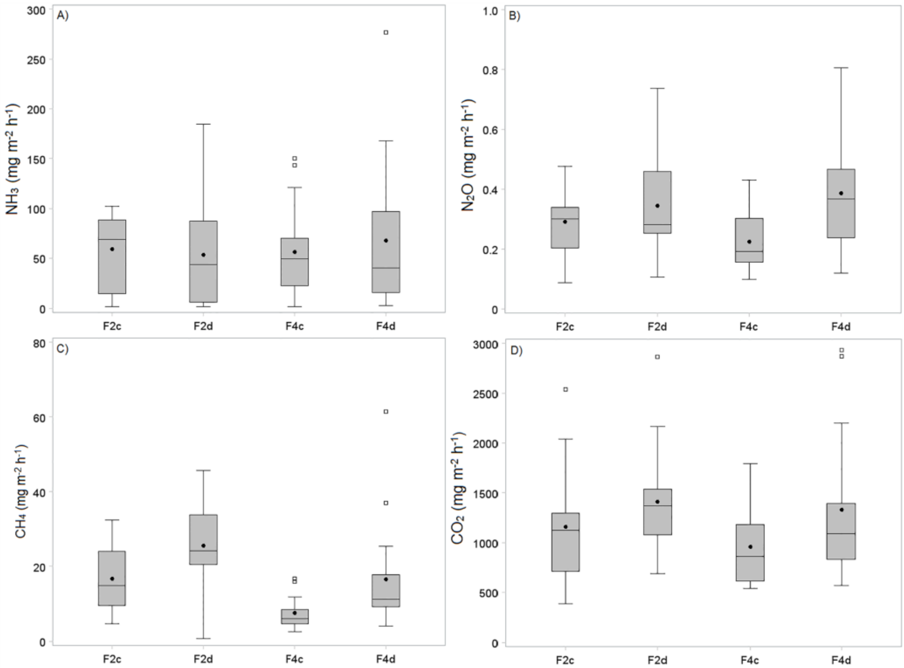


Figure 4.1. Comparison between emission factors of NH3 (A), N2O (B), CH4 (C) and CO2 (D) from clean (c) and dirty (d) surfaces of the feeding alley in farms equipped with scraper (Farm 2 and Farm 4).

Table 4.2. NH_3 emission factors (mg m^{-2} h^{-1}) from selected farms.

NH_3	FARM	Manure management	Obs	Median	Min	Max	Mean	SD	Group
Feeding alley	**1**	Perforated floor	34	24.81	1.61	59.94	26.05	17.06	B
	2	Scraper on concrete floor	38	54.53	1.23	184.61	56.11	44.26	C
	3	Flushing system	30	6.20	0.34	93.56	12.55	18.97	A
	4	Scraper on rubber mat	34	45.76	1.24	276.48	62.15	61.50	C
Resting area	**1**	Rubber mat cubicles	15	1.72	0.37	12.22	2.86	3.18	A
	2	Straw cubicles	14	3.40	0.79	5.06	3.34	1.25	A
	3	Rubber mat cubicles	12	3.45	0.75	6.86	3.69	1.98	A
	4	Straw cubicles	7	5.10	0.19	7.62	4.61	2.91	A

observations from farm 4 depends on saturation curves with low R2 values (that were not considered). This was probably due to the partial retention of NH_3 in the straw that induced low and irregular emissions.

4.3.2 N_2O emissions

N_2O emissions from the feeding alleys of the considered farms range from 0.22 to 0.91 mg m^{-2} h^{-1} (Table 4.3).

Higher emission factors were observed in the farm equipped with perforated floor (Farm 1), while lower emissions were measured in the farm with the flushing system (Farm 3). The Steel-Dwass-Critchlow-Fligner test clearly distinguished in particular Farm 1 as the highest emitting farm ($p<0.0001$ in all couple comparisons). Nevertheless, it was not possible to identify a unique group for Farm 4 (equipped with scraper on rubber mat), since its comparison with Farm 2 and Farm 3 did not result in a statistically significant difference ($p=0.866$ and $p=0.260$ respectively).

In the pit under the perforated floor aerobic and anaerobic conditions coexist, enhancing the nitrification-denitrification processes and resulting in an increased level of N_2O emissions. Otherwise, the periodical removal of manure from concrete floors, through scraping or flushing, limits the creation of the anaerobic conditions required for the production of nitrous oxide (denitrification process). Finally, the washing out operated by the flushing system with stabilized slurry (Farm 3) entails the dilution and aeration of manure residuals leaved on the surface, reducing the denitrification processes leading to N_2O emissions.

Low N_2O concentrations are reported also in previous studies (Leytem et al., 2013; Wu et al., 2012). Moreover, Ngwabie et al. (2009) discussed how liquid manure systems and frequent manure removal do not constitute a major source for this gas.

As shown in Figure 4.1B, comparing dirty or clean surfaces in the farms with scraper (i.e Farms 2 and 4), we found different emission levels only in Farm 4 (Wilcoxon test, $p=0.0105$). Hence, regarding N_2O emissions, the rubber pavement seems to enhance the efficiency of the scraper, increasing the adherence of the blade on the surface and limiting the accumulation of a significant unwanted layer of excreta. On the contrary, the scraper passage on concrete floor of Farm 2 did not significantly reduce N_2O emissions (p=0.5738), since the manure removal is limited by the pavement roughness.

Higher emission factors were observed in the resting area, ranging from 0.29 to 5.92 mg m-2 h-1 (Table 4.3). The Kruscal Wallis and the subsequent post hoc tests highlighted statically significant differences among farms ($p<0.0001$). In particular, significant N_2O emissions were measured from straw containing cubicles (Farms 2 and 4), while rubber mat cubicles emit lower levels of N_2O (Farms 1 and 3). Also Chadwick et al. (2011) reported significant N_2O emissions occurring from straw bedded buildings, confirming our results and suggesting the adoption of slurry based system to mitigating its emissions. As mentioned by Monteny et al. (2006), the mixture of manure and straw/litter, combined with (partial) compaction of the bedding, creates conditions that favor passive aeration, resulting in uncontrolled nitrification and denitrification.

Table 4.3. N_2O emission factors (mg m^{-2} h^{-1}) from selected farms.

N_2O	FARM	Manure management	Obs	Median	Min	Max	Mean	SD	Group
Feeding alley	1	Perforated floor	26	0.91	0.25	1.91	0.77	0.57	C
	2	Scraper on concrete floor	35	0.32	0.09	0.74	0.28	0.15	B
	3	Flushing system	18	0.22	0.05	0.66	0.19	0.15	A
	4	Scraper on rubber mat	29	0.31	0.10	0.81	0.25	0.18	A/B
Resting area	1	Rubber mat cubicles	16	0.29	0.14	0.63	0.22	0.15	A
	2	Straw cubicles	22	4.25	0.85	10.72	3.45	2.99	B
	3	Rubber mat cubicles	17	0.55	0.07	2.56	0.43	0.58	A
	4	Straw cubicles	21	5.92	0.71	9.95	5.89	2.68	B

4.3.3 CH_4 emissions

In the feeding alleys of the selected farms CH_4 emission levels range from 12.52 to 38.71 mg m^{-2} h^{-1} (Table 4.4). The manure removal systems influence the emissions ($p<0.0001$) and two groups were identified using the multiple comparisons test of the medians (Steel-Dwass-Critchlow-Fligner).

Methane emissions were the highest in the farm equipped with perforated floor (Farm 1). Intermediate emission levels were registered in Farms 2 and 3 (using scraper on concrete floor or flushing system respectively). The lowest emissions were measured in Farm 4, where the scraper cleans a rubber pavement.

In the housing solution with perforated floor, the reduced removal frequency of the slurry from the pit below the pavement allows the anaerobic fermentation processes, and favors CH_4 emissions. Methane has a low solubility in water (22.7 mg L^{-1} at 1 atm and 20°C) and passes directly to the gaseous phase, while the CO_2 is more soluble (1.45 g L^{-1} at 1 atm and 20°C) and it is distributed between the gaseous phase and the liquid phase. On the contrary, removing manure from the barns (scraping or flushing) eliminates the most relevant quantity of the sources of this gas.

Leytem et al. (2013) reported that accumulated manure in the barns contribute to greater CH_4 emission rates, even if the contribution of enteric fermentation represent the main source of methane (Sun et al., 2008).

Also Ngwabie et al. (2011) found higher CH_4 emissions from slatted floor buildings compared to farm with scraper. In the latter solution, the manure kept inside the building is small and did not significantly increase the overall CH_4 emissions, mainly produced from enteric fermentation. Furthermore, in a laboratory conducted study, Sommer et al. (2007) demonstrated that CH_4 could be generated in liquid manure held for about 2-3 weeks in under-floor storage pits where complete cleaning after each emptying is usually not carried out. On the other hand, comparing flushing and scraping, Cortus et al. (2015) did not find any evidence of the influence of manure handling system on the emission levels of CH_4.

The emission factors measured before and after the cleaning of the surface (Figure 4.1C), showed statistically significant differences in farms equipped with scrapers ($p=0.0419$ in Farm 2 and $p=0.0065$ in Farm 4). The frequent removal of slurry from the barns could be considered as a mitigation strategy for CH_4 emissions. Furthermore, the rubber mat posed on the floor improves the contact of the scraper blade with a smoother surface, enhancing the cleaning efficiency of the system.

In the cubicles the emission levels range from 4.81 to 179.91 mg m^{-2} h^{-1} (Table 4.4). The Kruscal Wallis test showed statistically significant difference among farms

($p<0.0001$), and the subsequent post-hoc test identified three groups. Higher emissions come from straw bedded cubicles, while lower emissions are associated with cubicles covered with rubber mat. This could be due to combination of anaerobic conditions and increasing temperature due to the heat production of the animal. In Farm 4 the measured emission factors were very high. We were not able to identify the reason of this peculiar behavior, probably due to some management practice not declared by the farmer.

Table 4.4. CH_4 emission factors (mg m^{-2} h^{-1}) from selected farms.

CH_4	FARM	Manure management	Obs	Median	Min	Max	Mean	SD	Group
Feeding alley	1	Perforated floor	15	38.71	10.14	92.14	22.86	29.72	B
	2	Scraper on concrete floor	36	21.59	0.76	45.66	21.36	11.25	B
	3	Flushing system	25	19.12	2.52	62.16	11.81	16.36	A/B
	4	Scraper on rubber mat	29	12.52	2.52	61.47	9.46	11.95	A
Resting area	1	Rubber mat cubicles	12	12.53	4.42	30.40	10.85	8.22	B
	2	Straw cubicles	13	52.99	3.54	110.94	64.07	41.84	B
	3	Rubber mat cubicles	16	4.81	0.57	32.94	2.85	7.59	A
	4	Straw cubicles	16	179.91	6.10	344.76	195.54	113.54	C

4.3.4 CO_2 emissions

CO_2 emitted from livestock farming is biogenic and it has not to be considered as a greenhouse gas. In fact, biogenic CO_2 is related to the natural carbon cycle and to the processing of biologically based materials (not including, therefore, fossil fuels). Nevertheless, the study of the emission of CO_2 can be useful to obtain a more complete overview of the livestock farming interaction with the environment.

CO_2 emissions from the feeding alleys range from 915 to 1720 mg m^{-2} h^{-1} (Table 4.5) and differences among the selected manure removal strategies were found using the Kruskal Wallis test ($p=0.0007$). The comparisons of medians distinguished farms into two groups, but it was not possible to clearly define to which group belongs the

Farm 4 ($p=0.404$ in the comparison with Farm 2, and $p=0.059$ in the comparison with Farm 3).

The ranking of emissions levels among farms is similar to those observed for N_2O. The highest values were found from the farm equipped with perforated floor (Farm 1), followed by Farm 2 and Farm 4, where the manure removal is obtained with scrapers (on concrete floor or solid floor covered with a rubber mat). Finally lower values characterized the farm that uses the flushing system (Farm 3).

Contrary to our measurements, Pereira et al. (2011) reported higher emissions from solid floors compared to slatted floors. Otherwise, Adviento-Borbe et al. (2010) highlighted a correlation between CO_2 and CH_4 emissions, produced simultaneously during organic matter decomposition and microbial aerobic and anaerobic respiration, that could explain the higher emissions we measured from the perforated floor.

The comparison among emission factors obtained on dirty or clean surfaces did no show statistically significant difference (Figure 4.1D).

In the resting area, CO_2 emissions range from 832 to 13066 mg m^{-2} h^{-1} (Table 4.5). Straw cubicles showed higher emissions than rubber mat cubicles ($p<0.0001$), as observed for other GHGs emissions. Furthermore, as already outlined for CH_4, the highest emission factors were observed in the Farm 4, confirming the link between CO_2 and CH_4 emissions (Adviento-Borbe et al., 2010).

Table 4.5. CO_2 emission factors (mg m^{-2} h^{-1}) from selected farms.

CO_2	FARM	Manure management	Obs	Median	Min	Max	Mean	SD	Group
Feeding alley	1	Perforated floor	24	1720	331	3947	1566	1096	B
	2	Scraper on concrete floor	38	1292	385	2565	1278	544	B
	3	Flushing system	32	915	65	3433	604	29	A
	4	Scraper on rubber mat	33	1149	542	2930	971	598	A/B
Resting area	1	Rubber mat cubicles	19	1030	105	2797	952	712	A
	2	Straw cubicles	24	7051	1209	14671	7278	3757	B
	3	Rubber mat cubicles	19	832	60	3405	534	862	A
	4	Straw cubicles	20	13066	1812	20756	14932	5517	C

4.4 Conclusions

The aim of this study was to investigate the influence of housing solutions and manure removal strategies on NH_3 and GHGs emissions.

Regarding the feeding alleys, where the majority of excreta accumulate, the use of scrapers increases the emission of ammonia as a consequence of the spreading of urine and of the increased air-exchanging surface, which enhance NH_3 volatilization. Otherwise, in the farm equipped with perforated floor GHGs emissions (N_2O and CH_4) from feeding alleys were higher. In this kind of housing solution, the manure remains under the floor surface for longer periods. Within the pit, the limited air circulation establishes anaerobic conditions in the slurry, which in turn promote the proliferation of denitrifying and methanogenic microorganisms.

The flushing system is associated to lower emissions for all the considered pollutants. The slope of the alley contributes to a prompt recovery of the liquid fraction of the slurry, while the washing process allows an almost complete removal of the excreta. Under these conditions, the NH_3 volatilisation is limited and the biological processes mediating N_2O and CH_4 emissions from the alley are less relevant with respect to other technologies. However, in the present work it was not possible to quantify the emissions during the flushing phase, which constitutes the most critical phase of this technology.

In any case, flushing should be conducted with a well-stabilized liquid fraction of the slurry in order to avoid secondary emissions during the washing out. Furthermore, this system generates a large amount of wastewater, which potentially becomes a source of gaseous emissions during its storage. Hence, projecting and managing correctly this technology is of paramount importance.

In the resting zone, measured emission factors were always higher in cubicles covered with straw. Although straw is optimal from an animal welfare point of view, its use as bedding material lead to a statistically significant increase of GHGs emissions. Once soiled, the straw augments the specific emitting surface of the resting area. Furthermore, the frequency of renewing and other management factors greatly influence the emissions levels of cubicles.

In order to control the emissions of greenhouse gases and ammonia from farms, it is appropriate to:

1. Ensure frequent and complete removal of manure from floors, also in presence of perforated floor;
2. Correctly incline floors in order to achieve faster separation of urine from solid fractions;
3. Use rubber mats, instead of concrete floors, coupled to scrapers in order to increase the cleaning efficiency;
4. Renew cubicles regularly in order to avoid the establishment of anaerobic conditions in deep layers of straw.

Acknowledgments

The research was conducted with support from the EU Research Project LIFE 09 ENV/IT/0214 Gas-Off. The authors thank Professor Pierluigi Navarotto for his precious teachings.

References

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

Ansari, A.S., Pandis, S.N., 1998. Response of Inorganic PM to Precursor Concentrations. Environmental Science & Technology, 32 (18), 2706-2714.

Braam, C.R., Swierstra, D., 1999. Volatilization of ammonia from dairy housing floors with different surface characteristics. Journal of Agricultural Engineering Research, 72 (1), 59-69.

Brewer, S.K., Costello, T.A., 1999. In situ measurement of ammonia volatilization from broiler litter using an enclosed air chamber. Transactions of the ASAE, 42 (5), 1415-1422.

Cai, L., Yu, J., Zhang, J., Qi, D., 2015. The effects of slatted floors and manure scraper systems on the concentrations and emission rates of ammonia, methane and carbon dioxide in goat buildings. Small Ruminant Research, 132 103-110.

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

Cortus, E.L., Jacobson, L.D., Hetchler, B.P., Heber, A.J., Bogan, B.W., 2015. Methane and nitrous oxide analyzer comparison and emissions from dairy freestall barns with manure flushing and scraping. Atmospheric Environment, 100 57-65.

Dinuccio, E., Balsari, P., Berg, W., 2008. GHG emissions during the storage of rough pig slurry and the fractions obtained by mechanical separation. Australian Journal of Experimental Agriculture, 48 (2), 93-95.

Erisman, J.W., Bleeker, A., Galloway, J., Sutton, M.S., 2007. Reduced nitrogen in ecology and the environment. Environmental Pollution, 150 (1), 140-149.

Hornig, G., Turk, M., Wanka, U., 1999. Slurry covers to reduce ammonia emission and odour nuisance. Journal of Agricultural Engineering Resource, 73 151-157.

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

IPCC, 2006. Emissions from livestock and manure management (Ch.10). *In:* IPCC Guidelines for national greenhouse gas inventories, Prepared by the National Greenhouse Gas Inventories Programme, vol.4. Eggleston H.S., B.L., Miwa K., Ngara T. and Tanabe K. (Eds.), IGES, Japan,

IPCC, 2013. *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, Stocker, T., Qin, D., Plattner, G., Tignor, M., Allen, S., Boschung, J., Nauels, A., Xia, Y., Bex, B., Midgley, B. (Eds.), Cambridge University Press, Cambridge, United Kingdom.

Leytem, A.B., Dungan, R.S., Bjorneberg, D.L., Koehn, A.C., 2013. Greenhouse Gas and Ammonia Emissions from an Open-Freestall Dairy in Southern Idaho. Journal of Environmental Quality, 42 (1), 10-20.

Monteny, G.-J., Bannink, A., Chadwick, D., 2006. Greenhouse gas abatement strategies for animal husbandry. Agriculture, Ecosystems & Environment, 112 (2–3), 163-170.

Moreira, V.R., Satter, L.D., 2006. Effect of Scraping Frequency in a Freestall Barn on Volatile Nitrogen Loss from Dairy Manure1. Journal of Dairy Science, 89 (7), 2579-2587.

Ngwabie, N.M., Jeppsson, K.H., Gustafsson, G., Nimmermark, S., 2011. Effects of animal activity and air temperature on methane and ammonia emissions from a naturally ventilated building for dairy cows. Atmospheric Environment, 45 (37), 6760-6768.

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

Oenema, O., Wrage, N., Velthof, G.L., Groenigen, W.J., Dolfing, J., Kuikman, P.J., 2005. Trends in Global Nitrous Oxide Emissions from Animal Production Systems. Nutrient Cycling in Agroecosystems, 72 (1), 51-65.

Ogink, N.W.M., 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 (3), 197-204.

Ogink, N.W.M., Mosquera, J., Calvet, S., Zhang, G., 2013. Methods for measuring gas emissions from naturally ventilated livestock buildings: Developments over the last decade and perspectives for improvement. Biosystems Engineering, 116 (3), 297-308.

Pereira, J., Fangueiro, D., Misselbrook, T.H., Chadwick, D.R., Coutinho, J., Trindade, H., 2011. Ammonia and greenhouse gas emissions from slatted and solid floors in dairy cattle houses: A scale model study. Biosystems Engineering, 109 (2), 148-157.

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

Philippe, F.-X., Cabaraux, J.-F., Nicks, B., 2011. Ammonia emissions from pig houses: Influencing factors and mitigation techniques. Agriculture, Ecosystems & Environment, 141 (3–4), 245-260.

Philippe, F.X., 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.

Samer, M., 2013. Emissions inventory of greenhouse gases and ammonia from livestock housing and manure management. Agricultural Engineering International: CIGR Journal, 15 (3), 29－54.

Sommer, S., Petersen, S., Sørensen, P., Poulsen, H., Møller, H., 2007. Methane and carbon dioxide emissions and nitrogen turnover during liquid manure storage. Nutrient Cycling in Agroecosystems, 78 (1), 27-36.

Sun, H., Trabue, S.L., Scoggin, K., Jackson, W.A., Pan, Y., Zhao, Y., Malkina, I.L., Koziel, J.A., Mitloehner, F.M., 2008. Alcohol, Volatile Fatty Acid, Phenol, and Methane Emissions from Dairy Cows and Fresh Manure. Journal of Environmental Quality, 37 (2).

Wu, W., Zhang, G., Kai, P., 2012. Ammonia and methane emissions from two naturally ventilated dairy cattle buildings and the influence of climatic factors on ammonia emissions. Atmospheric Environment, 61 232-243.

Zhang, G., Strøm, J.S., Li, B., Rom, H.B., Morsing, S., Dahl, P., Wang, C., 2005. Emission of Ammonia and Other Contaminant Gases from Naturally Ventilated Dairy Cattle Buildings. Biosystems Engineering, 92 (3), 355-364.

CHAPTER 5

5 Milk production Life Cycle Assessment: a comparison between estimated and measured emission inventory for manure handling

Baldini, C., Bava, L., Zucali, M., Guarino, M., 2017. Published in Science of the Total Environment; 625: 209-219.

Abstract

Measuring emissions from manure management operations (from the barns to the land) is a challenging task, subject to different uncertainties related to the spatial-temporal variability in the process leading to gaseous release. At the same time, emissions inventory is a prerequisite of Life Cycle Assessment (LCA) studies. Manure management emissions are usually estimated using equations developed by Intergovernmental Panel on Climate Change (IPCC, in the case of greenhouse gases emissions) and European Environmental Agency (EEA) for Nitrogen-related emissions. In the present study, the environmental impacts associated to three Italian dairy farms were calculated through a comparative LCA using two different approaches for complying the emission inventory. In the "estimated" approach (E) the commonly adopted IPCC and EEA equations were used, while in the "measured" approach (M) emissions actually measured were taken as input data to quantify the emissions associated to manure management. The results showed that the IPCC equation underestimates the manure management emissions, leading to a

10–42% lower global warming potential comparing E to M approach. On the other hand, ammonia related impact categories showed higher values if they were calculated using the estimated approach, underling that a safer level of estimation is maintained.

Keywords: ammonia, greenhouse gases, emission measurements, dairy farms, LCA, inventory filling.

5.1 Introduction

The concept of *sustainability* has become a key driver in the last few years, steering the more recent political and socio economical choices. With the publication of "The livestock long shadow" in 2006, livestock's production in general, and in particular cattle, has been included among major responsible of environmental pollution and climate change. Since then, the awareness about emission reduction from livestock activities (GHG and other pollutants) has increased, resulting in a large number of researches focused on quantifying the environmental burden of milk production (O'Brien et al., 2012; van der Werf et al., 2014).

The environmental impact of livestock farming is strictly related to the emissions of methane (CH_4), nitrous oxide (N_2O) and ammonia (NH_3), arising from the manure management continuum (i.e. the animal housing, yards, manure storage and treatment, and land spreading (Chadwick et al., 2011)), and responsible for climate change, acidification and eutrophication effects, among other impacts. Gaseous losses from ruminant livestock in the form of manure management are responsible for 15.2% of agricultural emissions (Holly et al., 2017). Emissions of CH_4, N_2O and NH_3 may occur simultaneously from different sources: enteric fermentations and manure storages are the most important source of CH_4; while animal excreta in housing, manure storage systems and land application constitute the main source of N_2O and NH_3 (Hou et al., 2015).

Life cycle assessment (LCA) is a structured, comprehensive, international standardized and widely adopted method to assess the environmental impacts of a product or a process (Battini et al., 2014; O'Brien et al., 2014). LCA studies have four main pillars: the goal and scope definition; the inventory analysis; the impact assessment and the interpretation of results (ISO, 2006a). During the inventory phase, LCA practitioners refer to internationally recognized models to account for GHG and nitrogen emissions. The method proposed by Intergovernmental Panel on Climate Change (IPCC 2006 a, b) is the most used (and recommended) for GHG estimation, while for NH_3 emissions, the most commonly selected reference are the equations developed by the European Environmental Agency (EEA, 2013) for the European area (Notarnicola et al., 2015). These models are based on emission factors (EFs) that were developed for the use in national GHG inventories, designed for the accounting at national scale (Nemecek and Ledgard, 2016). Their use for specific farming systems might be inappropriate, since the suggested EFs often do not take into account specific conditions of the investigated systems (Owen and Silver, 2015; Peter et al., 2016). Furthermore, recent researches indicate that the IPCC methodology may significantly underestimate CH_4 contributions from liquid dairy manure storage production, with discrepancies between inventory estimates and actual on-farm emissions (Baldé et al., 2016; Leytem et al., 2017; Lory et al., 2010).

Agricultural emissions are from nonpoint sources, characterized by high degree of variability due to climatic conditions, soil type, and agricultural practices (Goglio et al., 2017). For this reason, measuring emissions from manure management operations (from the barns to the land) is a challenging task, subject to different uncertainties related to the spatial-temporal variability in the process leading to gaseous release, which is strongly and complexly influenced by environmental conditions (Calvet et al., 2013; Owen and Silver, 2015). Despite the considerable efforts extended to measure gaseous emissions from natural ventilated buildings, measurement accuracy and standardization of methodology still are goals to be achieved (Takai et al., 2013).

Dairy system plays an outstanding role in the Italian context, but the high animal density characterizing the Northern regions pose a risk to the environment. The accurate estimation of the potential burdens associated to dairy farms is the first step for the identification of the best mitigation options that should be recommended to producers. In this context, manure handling systems play an important role, because different treatments and management strategies can alter manure composition, affecting GHG and NH_3 emissions from all the manure continuum (Holly et al., 2017).

The IPCC and EMEP/EEA equation are widely used for the estimation of emissions from the manure management. The aim of this work was to use two different data sources, field measurements or estimated emissions, to calculate the environmental impact associated to milk production in Italian dairy farms. In particular, results of LCA analysis conducted using the IPCC and EMEP/EEA equations for manure management were compared to the environmental impacts calculated using measured gaseous emissions. The use of these two different approaches for LCA calculation would allow to verify the degree of convergence of the methodologies applied for LCA and to underline their strengths and weakness. A Monte Carlo Simulation was also performed, in order to evaluate whether the two different approaches used for the LCA calculation could lead to different results even considering the high variability associated to measurements. Moreover, the impact caused by different animal categories (lactating or dry cows, heifers and calves) was investigated, to understand the contribution of the different physiological phases of animal growth to environmental burdens associated to milk production.

Results of the considered impact categories were separately discussed, highlighting differences achieved using the two calculation approaches (measured-M or estimated-E). The differences among impact associated to animal categories were underlined in a dedicated paragraph.

5.2 Materials and methods

5.2.1 Farms

For the present study three farms located in the North of Italy were monitored over one year (2015). The farms bred Holstein Friesians cows in permanent confinement. The main characteristics of the selected farms were resumed in Table 5.1. Farm 1 and Farm 2 can be considered of medium size for Italian conditions, as number of lactating cows and as arable land. Land was destined largely to cereal and annual forages. Farm 3, although smaller than the others, achieved a high production for cow.

Table 5.1. Farm characteristics.

	Unit	Farm 1	Farm 2	Farm 3
Herd				
Lactating cows	n	450	300	110
Dry cows	n	110	45	20
Heifers (12-24 mo)	n	300	150	64
Heifers (6-12 mo)	n	150	90	30
Calves (<6 mo)	n	150	90	34
Yield per cow	kg milk yr^{-1}	11,111	9,667	10,136
Livestock units	LU*	868	513	196
Replacement rate	%	25	30	24
Stocking rate	LU* ha^{-1}	8.35	3.29	3.93
Milk production intensity	t FPCM§ ha^{-1}	48.3	19.3	22.5
Annual milk production	t FPCM§	5025	3009	1127
Annual meat production	t live weight	90	87	40
Land				
Farm land	ha	104	156	50
Alfalfa	ha			4.5
Barley	ha	10		
Maize	ha	50	100	16.5
Maize after ryegrass	ha	40		29
Meadow	ha		16	
Ryegrass	ha	40		29
Sorghum	ha		20	
Soybean	ha	4	20	
Triticale	ha	10		
Wheat	ha		40	
Land productivity				
Alfalfa, hay	t DM ha^{-1}			12.0
Barley silage	t DM ha^{-1}	10.6		
Maize, high moisture ear maize	t DM ha^{-1}	14.7	14.7	
Maize, silage	t DM ha^{-1}	16.5	20.2	14.9
Meadow, hay/silage	t DM ha^{-1}		11.7	
Ryegrass, silage	t DM ha^{-1}	8.4		9.0
Sorghum, silage	t DM ha^{-1}		12.3	
Soybean, grain	t DM ha^{-1}		3.8	
Soybean, silage	t DM ha^{-1}	4.2		
Triticale, silage	t DM ha^{-1}	12.5		
Wheat, silage	t DM ha^{-1}	13.4		

	Unit	Farm 1	Farm 2	Farm 3
Purchased feeds				
Maize meal	t yr^{-1}	500		191
Soybean meal	t yr^{-1}	465	323	116
Sunflower meal	t yr^{-1}		206	6
Cotton seed	t yr^{-1}	168		
Molasses from sugar beet	t yr^{-1}	543		
Min&Vit supplements	t yr^{-1}	57	152	167
Straw	t yr^{-1}	482	179	11
Hay	t yr^{-1}	417	277	280
Other	t yr^{-1}	4591	270	
Manure handling system		perforated floor	concrete floor and flushing system	concrete floor and scraper

*LU: livestock unit (factors used for the calculation were: 1 for lactating cows; 0.8 for dry cows and heifers older than 2 years; 0.7 for heifers with age between 1 and 2 years old; 0.4 for calves and heifers younger than 1 year).
§FPCM: fat and protein corrected milk.

In the three farms, the barns hosting cows had more consistent construction features, reflecting some farmer's management choices for manure handling, while higher variability was observed in barns where replacement herd lives. In particular, barns destined to cows were equipped with different flooring type and different manure removal systems, representative of the most common option spread in the Po Valley, as better described below.

Farm 1 was equipped with perforated concrete floor (holes diameter of 3.5 cm). The manure accumulated in the pit below the slatted surface and was periodically removed (approximately every 14 days). The cubicles were covered with rubber mats and were cleaned manually.

Farm 2 was equipped with flushing system. The feeding and the resting alley had a convex (1.5% slope) and inclined (3% slope) concrete surface, in order to increase the cleaning efficiency. The flushing was carried out twice a day with a flowrate of 0.15 m^3 s^{-1} for about ten minutes. The flush system utilized mainly recycled effluent from a screw press solid-liquid separator or occasionally water from the municipal water supply network. The cubicles were equipped with rubber mats and covered with the solid fraction derived from the manure separation system.

Farm 3 had solid floor covered with a rubber mat pavement. Manure was removed with delta scrapers running twice a day. The cubicles were equipped with straw and cleaned weekly.

5.2.2 Life Cycle Assessment

An attributional LCA was performed according to the ISO 14040 and 14044 standards (ISO, 2006a; b), using the software Simapro PhD 8.4.0.0 (Pré Consultants, 2016).

5.2.2.1 Goal and scope definition

The aim of this study was to compare the environmental impact of three dairy farms with different manure handling options, using two different data set of emissions (measured or estimated emissions factors form manure management). The final scope was to verify the soundness of the LCIA results, in order to verify the goodness of the estimation approach in catching differences between the management alternatives commonly adopted in dairy farms. Furthermore, to quantify the importance of replacement animals to the environmental impacts associated to each farm, the contribution of different age classes in which the herd is usually subdivided and managed was analyzed and discussed.

5.2.2.2 Functional unit, allocation, system boundaries

The environmental impacts of farms were evaluated using 1 kg of Fat and Protein Corrected Milk (FPCM) as functional unit (FU). The FAO (2010) correction formula was used to adjust the raw milk production to a quantity of milk with standardized quality (4.0 % of fat and 3.3% of protein). Milk was considered as the main product of the farms and the biological allocation factor proposed by IDF (2010) was used to account for meat as co-product.

As described in the flow diagram drawn in Figure 5.1, the analysis was conducted “from cradle to farm gate”, considering as system boundaries all the on-farm processes (i.e. forages and crop production, fuel and electricity use, manure and livestock management) and the off farm processes linked to the production of external inputs (production of fertilizer and pesticides, fodders and bedding materials, feed concentrate, electricity and fuel, and associated transport). The study did not take account of farm personnel and of capital goods, such as buildings and machinery. Inputs such as medicines, detergents and disinfectants were excluded because their impact was estimated to be negligible (Ross et al., 2014). No account was made of carbon sequestration or loss resulting from land-use change in this study, since this is the current choice for standard footprinting methodology, because of a lack of scientific data at the world level (Daneshi et al., 2014; IDF, 2010).

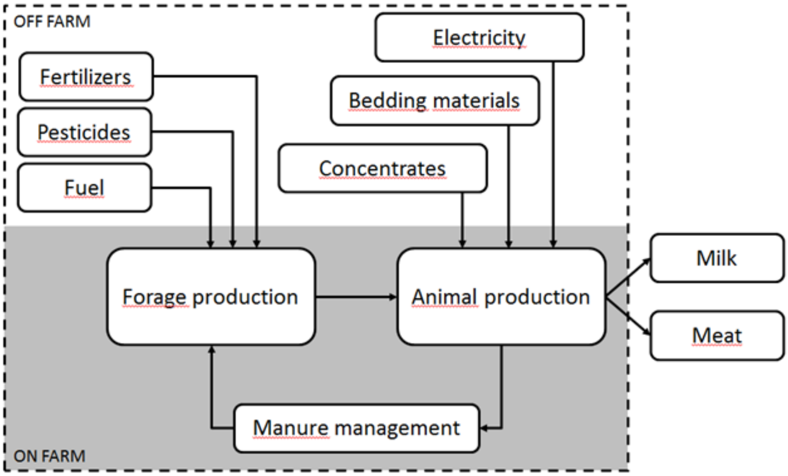


Figure 5.1. System boundaries.

The pathway was divided by farm activities (purchased materials, feed produced on farm, feed produced off farm, energetic consumptions and manure management) and then further divided by animal age classes (calves <6 mo, heifers 6-12 mo, heifers 12-24 mo, dry cows, cows) in order to understand the contribution of each animal category.

5.2.2.3 Life cycle inventory

Personal interviews with farmers were used in the data collection step. They provide details about cropping systems and field operations, fuel consumption, number of animals and housing systems, manure storage and animal diets. Questions about the inputs entering the farms were also posed, including amount of purchased feeds (both roughages and concentrates), fertilizers and pesticides, bedding materials, and number and origin of purchased replacement animals.

The amount of milk produced by each farm was provided by the farmers, whereas the amount of meat (as animal liveweight) was estimated on the basis of the number of animals sold for slaughter and their liveweight declared by the farmers.

5.2.2.3.1 Animal and farm operations

Animal diets: Information about specific diet of each animal category was collected. The CPM-Dairy Ration Analyzer Beta V3 software (Cornell-Penn-Miner, 2004) was used to estimate the key parameters of the diet composition (e.g. DM, CP, ether extract (EE), crud fiber, NDF, ADF and ADL, etc.). These parameters were used to calculate the gross energy intake and the digestibility of the diets according respectively to IPCC (2006a) and NRC (2001), and are summarised in Table 5.2. The composition of concentrate feed was estimated in the same way, using the raw materials reported in the commercial labels.

GHG emissions: CH_4 enteric emissions of each animal category were estimated starting from the gross energy of the feed diets, while CH_4 emissions from manure management were estimated using the volatile solids excretion calculated from the gross energy of the diets, following the Tier 2 IPCC (2006a) method.

N_2O emissions from manure, both direct and indirect, were calculated from the nitrogen excretion of the animals, as a result of the difference between the nitrogen intake and the nitrogen retained and excreted with milk (IPCC, 2006b).

CO_2 emissions from livestock respiration and manure were not accounted. It was assumed that they were balanced by the carbon previously absorbed and metabolized by crops composing the dairy diet, thus, being part of the carbon cycle, they not constitute an additional source of CO_2 (Holly et al., 2017).

Other emissions: NH_3 and NO_x emissions from farm operations were estimated according to the EEA method (EEA, 2013). The selected Tier 2 method starts from the nitrogen excreted by animals and applies a mass flow approach to calculate the NH_3 emissions, giving specific emission factors for each manure type (solid or slurry) and each step in the handling, expressed as a percentage of the NH_3-N content of manure.

CO_2 emissions from fuel combustion were estimated on the basis of fuel consumption declared by farmers.

The estimated emissions associated to animal and farm operations were reported in Table 5.3, disaggregated by animal category.

Table 5.2. Animal diets parameters.

Diet	Farm 1					Farm 2					Farm 3				
	LC	DC	H12-24	H6-12	C<6	LC	DC	H12-24	H6-12	C<6	LC	DC	H12-24	H6-12	C<6
DMI	25.90	10.20	11.2	4.60	6.08	22.56	8.76	6.13	3.39	4.53	20.9	10.52	10.65	6.18	6.08
CP	15.83	9.77	9.70	9.75	18.01	17.44	13.11	13.11	13.02	20.51	16.39	12.49	14.78	18.05	18.01
NDF	33.69	49.99	50.67	50.31	41.76	29.30	51.42	51.42	51.52	35.03	29.80	52.75	48.23	43.81	41.76
ADF	21.54	32.27	32.00	32.41	32.58	18.84	31.87	31.87	31.94	21.27	19.28	37.90	35.78	31.77	32.58
EE	3.45	2.64	2.51	2.62	2.94	3.85	3.17	3.17	3.14	2.81	3.30	3.07	2.94	2.83	2.94
NFC	42.66	30.06	31.48	29.78	27.14	46.16	27.52	27.52	27.53	36.64	40.82	22.01	25.77	25.01	27.14
Ash	6.17	8.92	7.7	8.91	14.11	4.91	6.44	6.44	6.44	6.95	12.78	13.32	11.58	13.89	14.11

LC: lactating cows; DC: dry cows, H12-24: heifers 12-24 months; H6-12: heifers 6-12 months; C<6: calves <6 months. DMI: Dry matter Intake (kg day^{-1}); CP: Crude Protein (% DM); NDF: Neutral Detergent Fiber (% DM); ADF: Acid Detergent Fiber (% DM); EE: Ether Extract (% DM); NFC: Non Fiber Carbohydrates (% DM); Ash (% DM).

Table 5.3. Estimated emission from animals and farm operations disaggregated by animal categories and expressed as kg of gas head^{-1} year^{-1}.

Emissions	Farm 1					Farm 2					Farm 3				
	LC	DC	H12-24	H6-12	C<6	LC	DC	H12-24	H6-12	C<6	LC	DC	H12-24	H6-12	C<6
CH_4 enteric	143.38	52.25	56.82	37.67	24.29	128.96	50.86	55.46	35.78	20.96	132.31	56.51	57.65	34.95	23.67
CH_4 manure management	63.13	3.06	5.81	2.63	5.76	153.35	37.32	70.70	1.85	1.11	33.61	12.73	16.84	4.98	3.36
N_2O dir	0.62	0.19	0.15	0.16	0.44	0.00	0.00	0.00	0.00	0.00	1.13	0.60	0.69	0.46	0.22
N_2O ind	0.87	0.15	014	0.11	0.30	0.00	0.25	0.19	0.13	0.32	0.99	0.45	0.61	0.35	0.17
NH_3 housing	20.89	19.82	15.59	19.93	20.45	20.98	21.95	20.98	20.98	19.93	20.98	20.35	20.98	20.35	22.94
NH_3 storage	18.54	12.71	17.19	22.78	20.48	18.18	12.53	18.18	18.18	22.78	18.18	20.91	18.18	20.91	7.93
NO_x	0.13	0.31	0.99	1.49	0.75	0.02	0.01	0.02	0.02	1.49	0.02	0.89	0.02	0.89	0.26

LC: lactating cows; DC: dry cows, H12-24: heifers 12-24 months; H6-12: heifers 6-12 months; C<6: calves <6 months.

5.2.2.3.2 Measured emissions

The NH_3, CH_4, N_2O and emission factors arising from the cow barns of the selected farms were taken from Baldini et al. (2016). In that study, the emissions data were seasonally monitored over a global period of 27 months and acquired from different shed areas (i.e. feeding alley and cubicles). The concentration of the different gases was measured simultaneously by means of an Infrared Photoacoustic Detector (IPD; Bruel&Kjaer, multi gas monitor type 1302) and subsequently elaborated to obtain the emission factors expressed as mg m^{-2} h^{-1}.

Measured emission factors were used as reference for the calculation of the GHG and NH_3 emissions from the barns of each farm (Table 5.4).

Storage emissions of CH_4 and N_2O were calculated using the data reported by Owen and Silver (2015). Reviewing published researches on field-scale measurements of GHG emissions from dairies, they provided average emission rates (kg head^{-1} year^{-1}) for CH_4, N_2O and CO_2 from different kind of slurry storages. These figures were used in order to fill the gap between estimated emissions and the field measured emissions: in fact the IPCC approach uses "combined" emission factors, that do not allow to distinguish between emissions from the barns and the storage. For NH_3 this step was not necessary, since EEA equations allow to separate emissions arising from different steps of manure handling.

Table 5.4. Comparison between estimated (E) and measured (M) emissions from manure management for lactating cows. Data were disaggregated by source (dairy barns or slurry storage) and expressed as kg of gas head^{-1} year^{-1}.

		CH_4		N_2O dir		NH_3	
		dairy barns	storage	dairy barns	storage	dairy barns	storage
Farm 1	**E**	63.13		0.62		20.90	18.54
	M	2.17	101*	0.05	0.3*	1.24	18.54
Farm 2	**E**	153.35		0.00		20.98	18.18
	M	1.11	368*	0.02	0.9*	0.40	18.18
Farm 3	**E**	33.61		1.13		20.98	18.18
	M	5.84	101*	0.19	0.3*	2.48	18.18

* Data taken from Owen et al. (2015).

5.2.2.3.3 *External inputs*

Off-farm activities related emissions were modeled using Ecoinvent® 3.3 database (Ecoinvent, 2016). The considered processes included the production chain of commercial feed (from crop growing to feed factory processing), production of purchased forages and bedding material, production of chemical fertilizers and pesticides, and diesel and electricity used in the farms. Transportation was also accounted, considering an average distance between farms and feed producers of 150 km, using a 16-32t lorry. The origin of the feed was taken into account (Italy, Europe, and extra Europe).

5.2.2.3.4 Land operations

The NH_3 and NO_x arising from manure and synthetic fertilizers application were estimated using the equations of EEA (2013). Direct and indirect N_2O losses from fertilizer application were estimated following respectively the Tier 2 and Tier 1 methods suggested by IPCC (2006b), accounting in the estimation the amount of nitrogen applied to soils both from synthetic fertilizers and from manure (slurry and solid) plus the nitrogen from crop residues.

Emissions occurring during field operations (i.e., plowing, harrowing, sowing, harvesting, etc.) were estimated using the processes of the Ecoinvent® 3.3 database (Ecoinvent, 2016).

Concerning emissions to water, the amount of nitrogen leached was estimated following the IPCC (2006b) model, while the emissions of PO_4^{3-} were calculated considering the amount of phosphorus drained away with water (run-off) and leached, as proposed by Nemecek and Kägi (2007).

For accounting purposes, the emissions that occurred after the land application of manure were assigned to the production of crops given that manure was used as a nutrient source.

5.2.2.4 Life cycle impact assessment

In order to understand the effect of different data sources used in this study on the potential impacts associated to a dairy farm, the following impact categories and technical quantities were evaluated per 1 kg of FPCM:

a. Global Warming, kg CO_2 eq

b. Acidification (A), mmol H^+ eq

c. Particulate matter formation (PMF), g $PM_{2.5}$ eq

d. Photochemical ozone formation (POF), g NMVOC eq

e. Terrestrial eutrophication(TE), mol N eq

f. Marine eutrophication (ME), g N eq

g. Mineral, fossil and renewable resource depletion (RD), mg Sb eq

The assessment was performed at midpoint using methods recommended by ILCD Handbook (IES, 2012). This shortlist of current best characterization methods represents the big effort to reach a higher level of standardization among LCA studies, undertaken by the European Joint Research Center. To make easier comparison with literature, in some case (for acidification and eutrophication impact categories) the potential impacts were recalculated using the CML method (Guinée et al., 2002). Indeed, LCA studies related to milk production are frequently performed using this method to conduce the LCIA (Baldini et al., 2017).

A Monte Carlo Simulation was performed in order to assess to what extent the uncertainties related to the measured data (CH_4, N_2O, NH_3 housing emissions) used in the study can influence the observed environmental impacts. The analysis was conducted with a confidence interval of 95% and 1000 iterations.

5.3 Results and discussion

Table 5.5 shows environmental impacts evaluation of milk production in three dairy farms using estimated (E) or measured (M) emissions arising from manure handling.

Table 5.5. Potential environmental impacts associated to the three selected farms, calculated with estimated (E) or measured emissions (M) and expressed per kg of FPCM.

Impact category	Unit	Farm 1		Farm 2		Farm 3	
		E	M	E	M	E	M
Global warming	kg CO_2 eq	1.62	1.69	1.11	1.58	1.26	1.38
Acidification	mmol H^+ eq	45.73	40.96	33.84	28.60	31.52	27.17
Particulate matter formation	g $PM_{2.5}$ eq	1.44	1.33	0.84	0.73	0.89	0.79
Photochemical ozone formation	g NMVOC eq	5.65	5.68	2.14	2.33	2.53	2.59
Terrestrial eutrophication	mol N eq	0.18	0.16	0.15	0.13	0.14	0.12
Marine eutrophication	g N eq	14.91	14.76	6.23	6.07	6.27	6.14
Mineral, fossil and ren. resource depletion	mg Sb eq	17.99	17.99	3.83	3.83	6.31	6.31

5.3.1 Global warming

For global warming impact category, the results ranged from 1.11 to 1.69 kg CO_2 eq kg^{-1} FPCM and were aligned with values reported by Italian researchers (Bacenetti et al., 2016; Battini et al., 2014; Bava et al., 2014; Guerci et al., 2013). The measured emissions led to increment the global warming potential (M/E: 4% for Farm 1, 42% for Farm 2, 10% for Farm 3).

This result was due to the higher quantity of CH_4 emissions from manure directly measured compared to CH_4 estimated through IPCC equations. Measured CH_4 emissions were always higher than estimated ones (see Table 5.4). This difference constituted the main cause leading to the increased global warming impact in the calculation approach using measured emission factors. The influence of N_2O emissions was limited. Indeed, they increased only in Farm 2, while in other farms measured emissions were lower than estimated ones. This was confirmed also by the contribution analysis for this impact category, which attributed to CH_4 the largest share of the impact contribution (50%), followed by CO_2 (37%) and N_2O (18%), using 1, 25, 298 CO_2 equivalent as characterization factors for 100-year time horizon for CO_2, CH_4 and N_2O respectively (IPCC, 2007).

Differently from field studies, frequently focused on the quantification of emissions from a particular stage of the manure management continuum, the IPCC estimations are based on "combined" emission factors that join together emissions arising from barns and storage. As regarding CH_4, IPCC estimations are function of Volatile Solids (VS) excreted by animals and thus loaded in the management system, the maximum CH_4-producing capacity of the manure (B_0), and CH_4 conversion factors (MCFs, defining the percentage of the B_0 achievable with the selected manure management system). The choice of the proper MCF is crucial for the representativeness of the final result, due to their broad variation also within the same climatic zone. Furthermore, MCFs cannot reflect the variety of possible solutions for manure treatment and are grouped in generic categories poorly defined.

In the case of direct N_2O emissions, the IPCC equation reflects the amount of N excreted by the animal categories corrected for an emission factor (named EF_3). The EF_3 is equal to zero for uncovered anaerobic lagoons, but our data do not support this result.

As outlined by Battini et al. (2014) the second main contributor to total GHG emissions, after enteric emissions, are storage emissions. However, they have a high degree of variability and are rarely experimentally measured. Battini et al. (2014) conducted a sensitivity analysis and demonstrated that the range of results found in many studies could be simply explained by the variation of this parameter. Therefore, they pointed out that additional experimental results quantifying storage emissions from manure and digestate management are essential in order to have a precise picture of GHG emissions from dairy farms.

A recent study conducted by Aguirre-Villegas and Larson (2017) highlights the key role of manure management in GHG emission reductions. The authors investigated how management practices and manure treatments affect emissions, identifying storage systems as weak points of the manure-handling continuum, and underling the importance of storage covers to reduce emissions.

5.3.2 Acidification

The acidification potential ranges from 27.17 to 45.73 mmol H^+ eq kg $FPCM^{-1}$.

As outlined by results reported in Table 5.5, calculations ran using estimated emission factors resulted in higher acidification potential in all the selected farms. The rank among farms observed in Table 5.4 was consistent with LCIA results. However, EEA equations resulted in greater estimation of NH_3 emissions of about 50%, while the difference observed among measured and estimated LCIA results was narrow (M/E: -10% for Farm 1, -16% for Farm 2, -14% for Farm 3; Table 5.5). Indeed, despite its high share of contribution, manure management was not the only factor affecting this impact category. Contributions to acidification were spilt among feed production (both on and off farm), with a share of 31-57%, and manure management operations, for the remaining 42-66%, while a small proportion of acidification can be attributed to energy consumption and purchased materials (Figure 5.2).

Farm 1 showed the higher impact compared to other farms. This value was mainly due to the higher proportion of feed produced off farm. For this farm, the main substances contributing to acidification were ammonia and sulfur dioxide with the 77% and 15% respectively (average of M and E). On the other hand, Farm 2 and 3 showed similar acidification potentials, and the ammonia and sulfur dioxide contributes to acidification were 92% and 4% on average. The high contribute of sulfur to acidification observed in Farm 1 may be due to a particular pesticide used by the farmer, containing high concentration of this element.

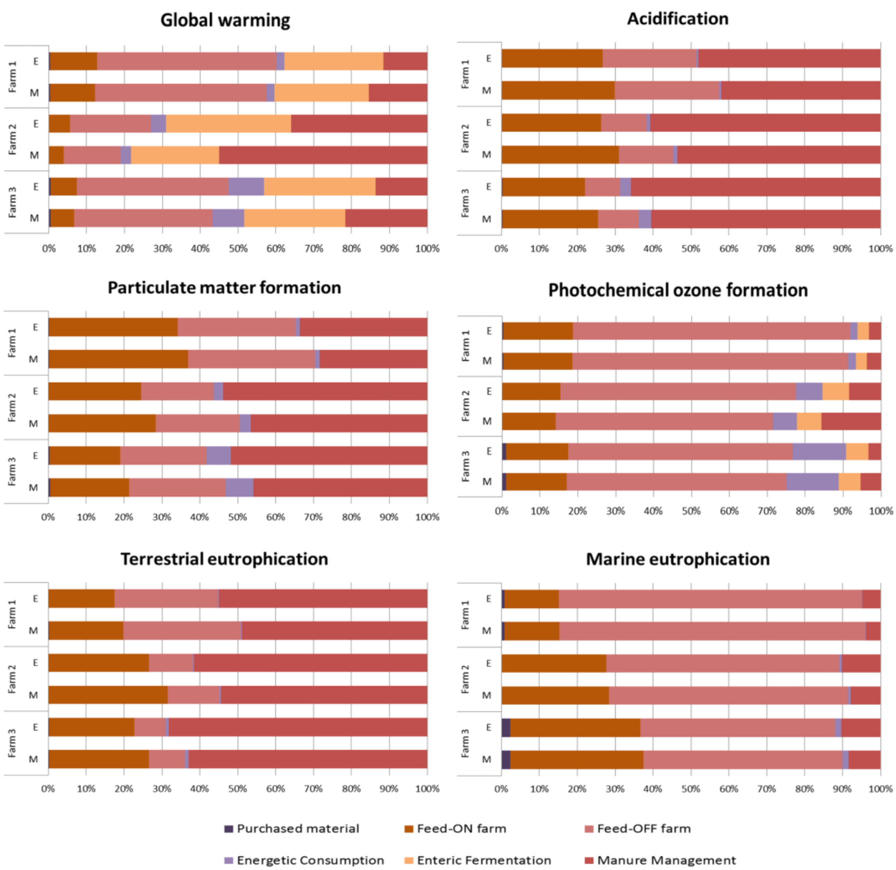


Figure 5.2. Potential environmental impacts associated to the three selected farms, contribution of farm activities.

The values reported for this impact category, recalculated according to the CML method for comparative purposes, were in line with those observed in literature (Castanheira et al., 2010; Guerci et al., 2013; Meul et al., 2014), apart from Farm 1, to which were associated high level of acidification as previously outlined (Table 5.6). Furthermore, also Battini et al. (2016) found that the acidification potential was mainly caused by NH_3 emissions from animal housing and from fertilizer application to the soil, but also by sulfur dioxide and nitrogen oxides from diesel combustion.

5.3.3 Particulate matter formation

The particulate matter formation ranged between 0.73 and 1.44 g $PM_{2.5}$ eq kg $FPCM^{-1}$ and differences were observed between calculations made using estimated or measured emissions (M/E: -8% for Farm 1, -14% for Farm 2, -11% for Farm 3). The higher impact was associated to Farm 1, while the potential impacts of Farm 2 and 3 showed lower values.

Farm activities that mainly contribute to this impact category were feed produced both on and off farm and the manure management operations, but with different percentage among farms. In Farm 1, where the amount of required feed is bigger (see Table 5.1), the contribution of auto-produced and purchased feed was respectively 35% and 32%, while the manure management accounts for 31% (average values among E and M). Otherwise, in Farm 2 and 3 the biggest contribution was associated to manure management operations (from 46 to 54%) while feed production accounted for a maximum of 28% (both on farm and off farm).

Particulate matter is strictly dependent from ammonia emission (Backes et al., 2016). Indeed, NH_3 is involved in reactions with sulfuric and nitric acid that lead to the formation of secondary inorganic particulate matter. This was confirmed by the high shares of impact contribution attributable to NH_3 (from 52% of Farm 1 M to 84% of Farm 2 E), in accordance to results previously reported by Battini et al. (2014). Direct emission of particulate matter ranged from 10% (Farm 2 E) to 21% (Farm 1 M and Farm 3 M).

5.3.4 Photochemical ozone formation

The photochemical ozone formation ranged from 2.33 to 5.68 g NMVOC eq kg $FPCM^{-1}$. E and M calculation approaches resulted in slightly different impact estimation (M/E: 0.5% for Farm 1, 9% for Farm 2, 2% for Farm 3).

The feed produced off-farm was the major contributor of this impact category, with a share ranging from 57% to 73%. Among the major species responsible of POF there were NO_x (63% on average), followed by NMVOC compounds (10% on average). However, the major differences among farms were observed in the contribution given by CH_4, ranging from 4% (Farm 1 E) to 21% (Farm 2 M).

Important contribution due to CH_4 emissions were reported also by González-García et al. (2013). Our results were in line with values reported by Battini et al. (2014), but they were higher compared to those referred to the farm subsystem in studies evaluating the UHT milk production (Castanheira et al., 2010; Djekic et al., 2014; Fantin et al., 2012) (Table 5.6).

5.3.5 Terrestrial and Marine Eutrophication

The impact on eutrophication was divided into two categories: terrestrial and marine.

For terrestrial eutrophication, significant percentage of variation between M and E calculation approaches were observed (M/E: -12% for Farm 1, -16% for Farm 2, -14% for Farm 3). Farm 1 showed the highest impact compared to other farms, due to the highest contribution of feed produced off-farm (29% for Farm 1, compared to 13% and 9% of Farm 2 and 3 respectively, average values).

Ammonia was the major contributors for this impact (93% on average) followed by NO_x (7% on average).

As regarding marine eutrophication, the results did not highlight important differences between M and E calculation approaches (M/E: -1% for Farm 1, -3% for Farm 2, -2% for Farm 3).

Feeds produced off farm gave the highest contribution to this impact category, ranging from 51% of Farm 3 E to 81% of Farm 1 M. This is partially in contrast to what previously observed by Battini et al. (2014), who reported a high share of field emissions contributing to this impact category, and may be due to the lower amount of feed purchased in the farm studied by those authors, compared to the farms investigated in this study. Nitrate was the species that mainly contribute to marine eutrophication, ranging from 75% (Farm 3 E and Farm 2 E) to 82% (Farm 1 M). Major differences were observed in ammonia contribution: 7% for Farm 1, 15% for Farm 2, 13% for Farm 3 (average values). This may be due to the characterization factor given to NH_3 in the ILCD method (Goedkoop et al., 2009) that is higher than the factor given to NO_3^{2-} (0.824 and 0.226 respectively).

Most of the studies found in literature use the CML method (Heijungs et al., 1992) to account for eutrophication potential. Impact assessment recalculated using this alternative method leads to results in accordance with the values obtained by other researchers (Bava et al., 2014; Nguyen et al., 2013; van der Werf et al., 2009).

5.3.6 Mineral, fossil and renewable resource depletion

Results for mineral, fossil and renewable resource depletion ranged from 3.83 mg Sb eq kg $FPCM^{-1}$ to 17.99 mg Sb eq kg $FPCM^{-1}$. These values were comparable to those obtained in the farm subsystem by Hospido et al. (2003), but they seemed quite low if compared to other literature data (Arsenault et al., 2009; Castanheira et al., 2010; González-García et al., 2013).

In all the considered farms, resource depletion is mainly due to feed production on farm land and outside of dairy farm with values from 91 to 99%. The highest estimations for this impact category expressed per kg of FPCM was associate to Farm 1, as a consequence of the high quantity of feed purchased.

Table 5.6. Comparison among acidification, eutrophication and photochemical oxidation impacts reported in literature and those obtained in the present study using CML as life cycle impact assessment method.

Reference	**FU**	**LCIA method**	**Acidification potential** (kg SO_2 eq)	**Eutrophication** (kg PO_4^{3-} eq)	**Photochemical oxidation** (kg C_2H_4 eq)
Present study	FPCM	CML	15.14-27.26	5.8-11.3	0.31-0.52
Arsenault et al., 2009	raw milk	CML	9.6	3.17	0.23
Bacenetti et al., 2016	FPCM	EPD	6.5	2.95	0.75
Bava et al., 2014	FPCM	EPD			
Castanheira et al., 2010	raw milk	CML	20.41	7.04	0.19
Djeick et al., 2014	UHT milk	CCalC			0.26
Fantin et al., 2012	HQ milk	CML			0.32-0.35
Gonzalez-Garcia	ECM	CML			0.27
Guerci et al., 2013	ECM	EPD	7.44-25.64	4.61-11.12	
Meul et al., 2014	FPCM	CML	11.26-15.62	3.7-4.3	
Nguyen et al., 2013	FPCM	CML	9.85-12.09	4.37-5.05	
van der Werf et al., 2009	FPCM	CML	7.6	7.1	

5.3.7 Monte Carlo Simulation (MCS)

Uncertainty analysis of the measured data was carried out using Monte Carlo statistical methodology. Generating thousands of random samplings of the input data, MCS propagates the uncertainties throughout the LCA model and gives a probabilistic distribution of the predicted impacts (Chen and Corson, 2014). In this case, a comparative analysis was carried out to understand whether the LCA conducted using the different approaches described in this study (measured vs estimated data) lead to different results, even considering the high variability associated to measurements.

Figure 5.4 shows the graphical results of the uncertainty analysis for the comparison between environmental impact assessment of 1 kg of FPCM, using measured data (M) or estimated (E) for manure management emissions. For each indicator, the blue bar represents the probability that environmental impact calculated using estimated data could result higher than the impact calculated using measured data (E≥M), while the orange bar represents the opposite (E<M).

The uncertainty analysis confirmed that environmental impacts calculated using measured data (M) resulted lower than estimated (E) for acidification, particulate matter, terrestrial and marine eutrophication (level of statistical significance > 99.9%). These results underlined that gas emissions measurements, despite its variability, lead to significantly different environmental impact estimations.

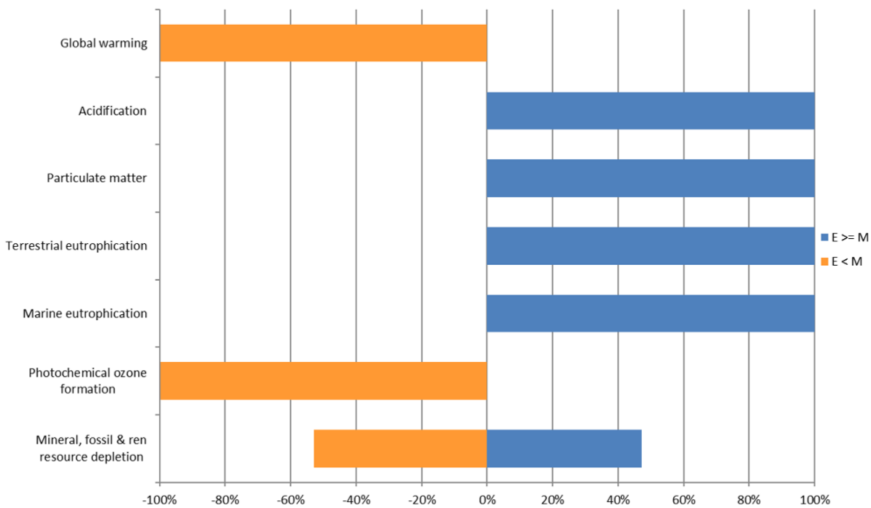


Figure 5.3. Results of the uncertainty analysis for the comparison between LCA results using estimated (E) and measured emissions (M) for the evaluated impact categories.

5.3.8 Contribution of different animal categories

5.3.8.1 Lactating and dry cows

Lactating cows were responsible of the largest contribute to all impact categories considered in the study (Figure 5.4). Compared to other animal categories, lactating cows were the larger emitters of GHGs, giving the highest contribute to climate change, ranging from 58 to 83% of the kg CO_2 eq kg FPCM. The number of animals and the higher feeding requirements, resulting in a larger feed consumption (both purchased and produced on farm), and the significant contribution of enteric fermentation were major responsible of the high share of global warming attributable to cows.

Farm 2 had the largest number of dairy cows, as percentage of the total herd (44.4% Farm 2; 42.6% Farm 3 and 41.5% Farm 1), this caused higher impacts of these animals on total farm global warming (79%). Instead, Farms 1 and 3 had similar percentage of dairy cows but their contribution was different, 59% for Farm 1 and 68% for Farm 3.

Dry cows contributed to global warming for 4-9%. The highest contribution was found in Farm 1, where the percentage of dry cows on the total herd were 10% (7% Farm 2 and 8% Farm 3). During dry period, usually lasting 60 days, cows still contribute to overall emissions of CH_4, NH_3 and other pollutants but they are not milked. From an environmental point of view, this period is quite crucial: if it is too long farm annual milk production decreases and the environmental impacts per unit of product increase.

Lactating cows contributed for 47-57% on acidification, as a consequence of NH_3 emission from manure and feed production (ON and OFF farm). For the other impact categories, lactating cows' contribution changed from 46 to 86%; the highest value was found in resource depletion, due to the high values of energy needed to produce feed for lactating period.

5.3.8.2 Heifers and Calves

On average 49% of the herd was represented by heifer (from 6 to 24 months) and calves (<6 months). This data was almost the same in the three farms, but the contribution of these animal categories to the environmental impacts were quite different among farms.

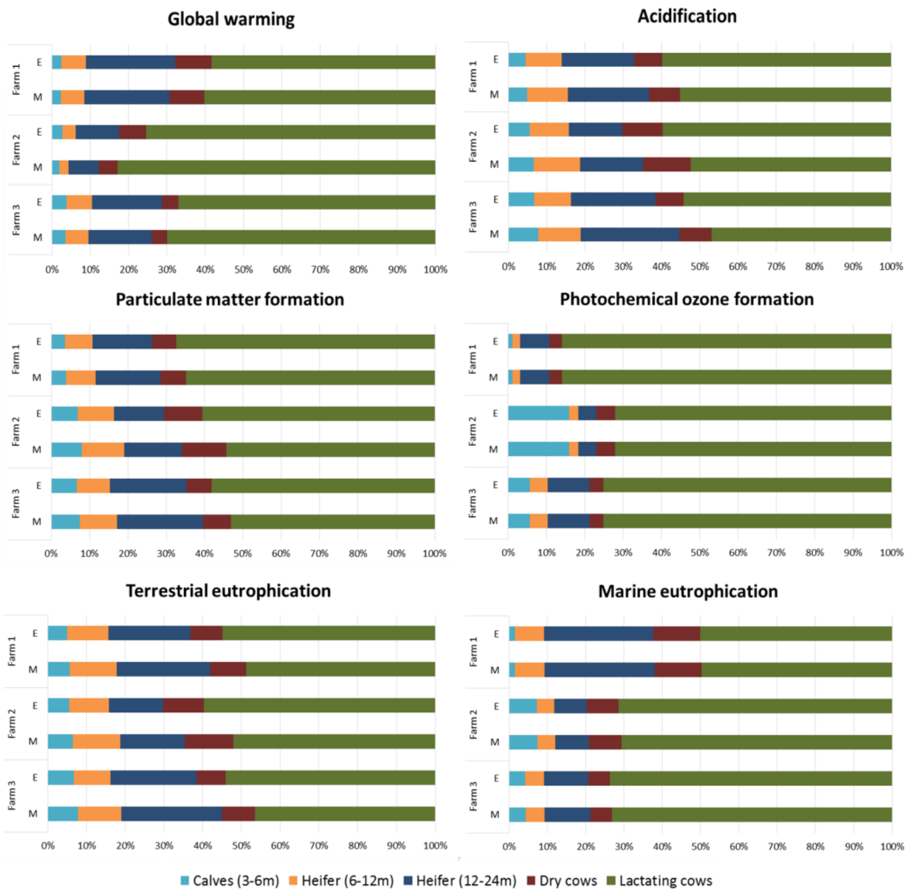


Figure 5.4. Potential environmental impacts associated to the three selected farms, contribution of different animal categories.

For global warming, heifer and calves contribution ranged from 12 to ≥ 30%. Heifers from 12 to 24 months are major responsible of this figures. This growing phase is crucial for the success of the whole production system, since in this period is usually preformed the first insemination of the heifers. If management problems occur, the age of first calving is postponed and the unproductive period becomes longer. Management choices could increase reproduction efficiency in this phase in order to optimize parameters, such as heat detection rate and pregnancy rate. From an environmental point of view, these operational parameters would decrease the unproductive period and increase milk production. Indeed, as suggest by de Boer et al. (2011) improving fertility of the herd would reduce net GHG emissions, because fewer animals, and hence less feed, are needed to produce the same amount of product.

For environmental impact categories highly dependent from NH_3 production, as acidification, particulate matter formation and terrestrial eutrophication, the contribution of heifer and calves ranged from 30 to 45%. The type of manure produced may influence this result. Indeed, replacement animals were often reared on litter based systems, which potentially increase GHG emissions (Hou et al. 2015). The shift towards slurry based system also for these animal categories, followed by proper managing of storages and manure spreading, could help improve the environmental impact of the replacement herd.

5.4 Conclusions

Measured and estimated calculation approaches led to different LCA results. In particular, the global warming impact seemed to be underestimated by the IPCC equations. This method, developed to compile national inventories, is unlikely to accurately approximate the emissions from manure management if applied to a specific dairy farm. More experimental data are needed to make emission factors (MCF and EF_3) more precise and flexible in order to place estimations closer to the actual level of emissions. Detailed data from representative manure systems are needed to guide climate change mitigation strategies. On the other hand, NH_3 related impact categories showed a higher values if they were calculated using the estimated approach, underling that a safer level of estimation was maintained. The innovative approach of this study allowed to underline the share of environmental impact from different animal categories. A large part of environmental impact comes from unproductive and young animals, for this reason management choices for housing and feeding of these animals can be crucial for the sustainability of the whole milk production process.

References

Aguirre-Villegas H. A., Larson R. A., 2017. Evaluating greenhouse gas emissions from dairy manure management practices using survey data and lifecycle tools. Journal of Cleaner Production; 143: 169-179.

Arsenault N., Tyedmers P., Fredeen A., 2009. Comparing the environmental impacts of pasture-based and confinement-based dairy systems in Nova Scotia (Canada) using life cycle assessment. International Journal of Agricultural Sustainability; 7: 19-41.

Bacenetti J., Bava L., Zucali M., Lovarelli D., Sandrucci A., Tamburini A., et al., 2016. Anaerobic digestion and milking frequency as mitigation strategies of the environmental burden in the milk production system. Science of the Total Environment; 539: 450-459.

Backes A. M., Aulinger A., Bieser J., Matthias V., Quante M., 2016. Ammonia emissions in Europe, part II: How ammonia emission abatement strategies affect secondary aerosols. Atmospheric Environment; 126: 153-161.

Baldé H., VanderZaag A. C., Burtt S., Evans L., Wagner-Riddle C., Desjardins R. L., et al., 2016. Measured versus modeled methane emissions from separated liquid dairy manure show large model underestimates. Agriculture, Ecosystems and Environment; 230: 261-270.

Baldini C., Gardoni D., Guarino M., 2017. A critical review of the recent evolution of Life Cycle Assessment applied to milk production. Journal of Cleaner Production; 140: 421-435.

Baldini C., Borgonovo F., Gardoni D., Guarino M., 2016. Comparison among NH3 and GHGs emissive patterns from different housing solutions of dairy farms. Atmospheric Environment; 141: 60-66.

Battini F., Agostini A., Boulamanti A. K., Giuntoli J., Amaducci S., 2014. Mitigating the environmental impacts of milk production via anaerobic digestion of manure: Case study of a dairy farm in the Po Valley. Science of the Total Environment; 481: 196-208.

Battini F., Agostini A., Tabaglio V., Amaducci S., 2016. Environmental impacts of different dairy farming systems in the Po Valley. Journal of Cleaner Production; 112: 91-102.

Bava L., Sandrucci A., Zucali M., Guerci M., Tamburini A., 2014. How can farming intensification affect the environmental impact of milk production? Journal of Dairy Science; 97: 4579-4593.

Calvet S., Gates R. S., Zhang G., Estellés F., Ogink N. W., Pedersen S., et al., 2013. Measuring gas emissions from livestock buildings: a review on uncertainty analysis and error sources. Biosystems engineering; 116: 221-231.

Castanheira É., Dias A. C., Arroja L., Amaro R., 2010. The environmental performance of milk production on a typical Portuguese dairy farm. Agricultural Systems; 103: 498-507.

Chadwick D., Sommer S., Thorman R., Fangueiro D., Cardenas L., Amon B., et al., 2011. Manure management: Implications for greenhouse gas emissions. Animal Feed Science and Technology; 166–167: 514-531.

Chen X., Corson M. S., 2014. Influence of emission-factor uncertainty and farm-characteristic variability in LCA estimates of environmental impacts of French dairy farms. Journal of Cleaner Production; 81: 150-157.

Cornell-Penn-Miner, 2004. CPM Dairy. Dairy cattle ration analyzer, version 3.0.6. Cornell Univeristy, Ithaca, NY.

Daneshi A., Esmaili-sari A., Daneshi M., Baumann H., 2014. Greenhouse gas emissions of packaged fluid milk production in Tehran. Journal of Cleaner Production; 80: 150-158.

de Boer I. J. M., Cederberg C., Eady S., Gollnow S., Kristensen T., Macleod M., et al., 2011. Greenhouse gas mitigation in animal production: towards an integrated life cycle sustainability assessment. Current Opinion in Environmental Sustainability; 3: 423-431.

Djekic I., Miocinovic J., Tomasevic I., Smigic N., Tomic N., 2014. Environmental life-cycle assessment of various dairy products. Journal of Cleaner Production; 68: 64-72.

Ecoinvent, 2016. Ecoinvent database v3.3. Swiss Centre for Life Cycle Inventories. Dübendorf, Switzerland.

EEA, 2013. 3.B. Manure management. EMEP/EEA emission inventory guidebook 2013. European Environment Agency, Copenhagen, 2013.

Fantin V., Buttol P., Pergreffi R., Masoni P., 2012. Life cycle assessment of Italian high quality milk production. A comparison with an EPD study. Journal of Cleaner Production; 28: 150-159.

FAO, 2010. Greenhouse Gas Emissions from the Dairy Sector: A Life Cycle Assessment. Rome.

Goedkoop M., Heijungs R., Huijbregts M., De Schryver A., Struijs J., Van Zelm R., 2009. ReCiPe 2008. A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level; 1.

Goglio P., Smith W., Grant B., Desjardins R., Gao X., Hanis K., et al., 2017. A comparison of methods to quantify greenhouse gas emissions of cropping systems in LCA. Journal of Cleaner Production; 172: 4010-4017.

González-García S., Castanheira T. G., Dias A. C., Arroja L., 2013. Using Life Cycle Assessment methodology to assess UHT milk production in Portugal. Science of the Total Environment; 442: 225-234.

Guerci M., Knudsen M. T., Bava L., Zucali M., Schönbach P., Kristensen T., 2013. Parameters affecting the environmental impact of a range of dairy farming systems in Denmark, Germany and Italy. Journal of Cleaner Production; 54: 133-141.

Guinée J. B., Gorrée M., Heijungs R., Huppes G., Kleijn R., de Koning A., van Oers L., Wegener Sleeswijk A., Suh S., Udo de Haes H. A., de Bruijn H., van Duin R., Huijbregts M. A. J., Lindeijer E., Roorda A. A. H., van der Ven B. L., Weidema B. P., 2002. Handbook on Life Cycle Assessment; Operational Guide to the ISO Standards. Institute for Environmental Sciences, Leiden University, The Netherlands.

Heijungs R., Guinée J. B., Huppes G., Lankreijer R. M., Udo de Haes H. A., Wegener Sleeswijk A., et al., 1992. Environmental life cycle assessment of products: guide and backgrounds (part 1).

Holly M. A., Larson R. A., Powell J. M., Ruark M. D., Aguirre-Villegas H., 2017. Greenhouse gas and ammonia emissions from digested and separated dairy manure during storage and after land application. Agriculture, Ecosystems & Environment; 239: 410-419.

Hospido A., Moreira M., Feijoo G., 2003. Simplified life cycle assessment of Galician milk production. International Dairy Journal; 13: 783-796.

Hou Y., Velthof G. L., Oenema O., 2015. Mitigation of ammonia, nitrous oxide and methane emissions from manure management chains: a meta- analysis and integrated assessment. Global Change Biology; 21: 1293-1312.

IDF, 2010. A common carbon footprint approach for dairy - The IDF guide to standard life cycle assessment methodology for dairy sector. International Dairy Federation.

IES, 2012. Characterisation factors of the ILCD Recommended Life Cycle Impact Assessment methods. Database and Supporting Information. In: Institute for Environment and Sustainability, editor. EUR 25167. Publications Office of the European Union. Luxemburg.

IPCC, 2007. In: Solomon S., Quin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L., editors. Climate change 2007-the physical science basis: Working group I contribution to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press.

IPCC, 2006a. Emissions from Livestock and Manure Management. In: Eggleston H., Buendia L., Miwa K., Ngara T., Tanabe K., editors. IPCC guidelines for national greenhouse gas inventories. Volume 4: Agricolture, Forestry and Other Land Use. Chapter 10. Institute for Global Environmental Strategies, Hayama, Japan, 2006a.

IPCC, 2006b. N2O emissions from managed soils, and CO2 emissions from lime and urea application. In: Eggleston H., Buendia L., Miwa K., Ngara T., Tanabe K., editors. IPCC guidelines for national greenhouse gas inventories. Volume 4: Agricolture, Forestry and Other Land Use. Chapter 11. Institute for Global Environmental Strategies, Hayama, Japan, 2006b.

ISO, 2006a. Environmental Management - Life Cycle Assessment-Principles and Framework. EN ISO 14040:2006. EN ISO 14040. International Organization for Standardization. Geneva, Switzerland.

ISO, 2006b. Environmental Management - Life Cycle Assessment-Requirements and Guidelines. EN ISO 14044:2006. EN ISO 14044:2006. International Organization for Standardization. Geneva, Switzerland.

Leytem A., Bjorneberg D., Koehn A., Moraes L., Kebreab E., Dungan R., 2017. Methane emissions from dairy lagoons in the western United States. Journal of Dairy Science; 100: 6785-6803.

Lory J. A., Massey R., Zulovich J., 2010. An evaluation of the USEPA calculations of greenhouse gas emissions from anaerobic lagoons. Journal of environmental quality; 39: 776-783.

Meul M., Van Middelaar C. E., de Boer I. J. M., Van Passel S., Fremaut D., Haesaert G., 2014. Potential of life cycle assessment to support environmental decision making at commercial dairy farms. Agricultural Systems; 131: 105-115.

Nemecek T., Kägi T., 2007. Life cycle inventories of Swiss and European agricultural production systems. Final report ecoinvent v2.0 No.15. Agroscope Reckenholz-Taenikon Research Station ART, Swiss Centre for Life Cycle Inventories Zurich and Dübendorf, Switzerland.

Nemecek T., Ledgard S., 2016. Modelling farm and field emissions in LCA of farming systems: the case of dairy farming. In: Holden N. M., editors. The 10th International Conference on Life Cycle Assessmnet of Food (LCA Food 2016). Univeristy College Dublin (UCD), Dublin, Ireland, 2016.

Nguyen T. T. H.,Yan Doreau M., Corson M. S., Eugène M., Delaby L., Chesneau G., et al., 2013. Effect of dairy production system, breed and co-product handling methods on environmental impacts at farm level. Journal of Environmental Management; 120: 127-137.

Notarnicola B., Salomone R., Petti L., Renzulli P. A., Roma R., Cerutti A. K., 2015. Life Cycle Assessment in the Agri-food Sector: Case Studies, Methodological Issues and Best Practices: Springer.

NRC, 2001. Nutrient Requirements of Dairy Cattle. 7th revised edition. Washington, DC: The National Academies Press.

O'Brien D., Capper J. L., Garnsworthy P. C., Grainger C., Shalloo L., 2014. A case study of the carbon footprint of milk from high-performing confinement and grass-based dairy farms. Journal of Dairy Science; 97: 1835-1851.

O'Brien D., Shalloo L., Patton J., Buckley F., Grainger C., Wallace M., 2012. Evaluation of the effect of accounting method, IPCC v. LCA, on grass-based and confinement dairy systems' greenhouse gas emissions. Animal; 6: 1512-1527.

Owen J. J., Silver W. L., 2015. Greenhouse gas emissions from dairy manure management: A review of field-based studies. Global Change Biology; 21: 550-565.

Peter C., Fiore A., Hagemann U., Nendel C., Xiloyannis C., 2016. Improving the accounting of field emissions in the carbon footprint of agricultural products: a comparison of default IPCC methods with readily available medium-effort modeling approaches. The International Journal of Life Cycle Assessment; 21: 791-805.

PRé Consultants (2016). SimaPro (8.4.0.0). LCA Software. Amersfoort, the Netherlands.

Ross S. A., Chagunda M. G. G., Topp C. F. E., Ennos R., 2014. Effect of cattle genotype and feeding regime on greenhouse gas emissions intensity in high producing dairy cows. Livestock Science; 170: 158-171.

Takai H., Nimmermark S., Banhazi T., Norton T., Jacobson L. D., Calvet S., et al., 2013. Airborne pollutant emissions from naturally ventilated buildings: proposed research directions. Biosystems engineering; 116: 214-220.

van der Werf H. M. G., Garnett T., Corson M. S., Hayashi K., Huisingh D., Cederberg C., 2014. Towards eco-efficient agriculture and food systems: Theory, praxis and future challenges. Journal of Cleaner Production; 73: 1-9.

van der Werf H. M. G., Kanyarushoki C., Corson M. S., 2009. An operational method for the evaluation of resource use and environmental impacts of dairy farms by life cycle assessment. Journal of Environmental Management; 90: 3643-3652.

CHAPTER 6

6 The influence on biogas production of three slurry handling systems in dairy farms

Coppolecchia, D., Gardoni, D., Baldini, C., Borgonovo, F., Guarino, M., 2015. Published in Journal of Agricultural Engineering; 46(1): 30-35.

Abstract

Handling systems can influence the production of biogas and methane from dairy farm manures. A comparative work performed in three different Italian dairy farms showed how the most common techniques (scraper, slatted floor, flushing) can change the characteristics of collected manure. Scraper appears to be the most "neutral" choice, as it does not significantly affect the original characteristics of manure. Slatted floor produces a manure that has a lower methane potential in comparison with scraper, due to: a lower content of volatile solids caused by the biodegradation occurring in the deep pit, and a lower specific biogas production caused by the change in the characteristics of organic matter. Flushing can produce three different fluxes: diluted flushed manure, solid separated manure and liquid separated manure. The diluted fraction appears to be unsuitable for conventional anaerobic digestion in completely stirred reactors (CSTR), since its content of organic matter is too low to be worthwhile. The liquid separated fraction could represent an interesting material, as it appears to accumulate the most biodegradable organic fraction, but not as primary substrate in CSTR as the organic matter

concentration is too low. Finally, the solid-liquid separation process tends to accumulate inert matter in the solid separated fraction and, therefore, its specific methane production is low.

Keywords: manure handling, biogas production, dairy farms, anaerobic digestion.

6.1 Introduction

Anaerobic digestion is a robust and widely applied biochemical conversion process for the production of energy from biodegradable organic matter (Appels et al., 2011). Livestock and agricultural waste, and energy crops are commonly used as substrates for their abundance and availability: in particular, dedicated energy crops (e.g. maize, triticale, sugar beet, etc.) emerged in specific situations as a cost-effective option in order to increase the return of the invested capital (Gissén et al., 2014). However, the ethical issues regarding energy crops have been the subject of continuous debate in the last years. In fact, the demand for energy crops appears to increase the direct and indirect competition among energy, land and food (Fritsche et al., 2010). It becomes therefore important to enhance the energetic conversion of other low value substrates, in particular of abundant organic waste. In livestock farming, this approach corresponds to both digestion processes that efficiently converts organic matter into methane, and manure/slurry management systems that allows a complete and prompt recovery of fresh excreta (Holm-Nielsen et al., 2009).

As already outlined in the literature (Larney et al., 2006; Martinez et al., 2009), different handling systems can determine the "freshness" of the available organic matter (i.e. the time elapsed between faeces deposition and collection/utilisation), influencing the quality of manure. Freshness is a key element, as biodegradation can occur also before the introduction of manure in the anaerobic reactor. Consequently, the longer the interval between the excretion and the beginning of the anaerobic process, the higher the amount of non-collected biogas (Møller et al., 2004a; Gopalan et al., 2013).

From a practical point of view, manure collection is strictly related to housing systems and bedding options. Conveyance cleaning systems like scraping, flushing-scraping and flushing are common in free stall sheds with solid floors, while in sheds with slatted floors manure is removed by gravity, and litter is manually renewed if present in resting areas (Meyer et al., 2011). Scrapers mechanically collect excreta preserving their characteristics, while flushing systems collects excreta hydraulically, diluting them. In housing facilities equipped with scrapers or flushing systems, the collection of faeces and urine is frequent (1-2 times per day), and manure freshness is always guaranteed. On the contrary, when slatted floors with underlying deep pits are adopted, a longer time interval occurs between faeces production and utilization, and the biogas potential is reduced as a function of the retention time (Moset et al., 2012). A decrease of the potential methane production of 4.3–6.6% after 15 days storage and of 7.7–11.9% after 30 days storage was observed (Møller et al., 2004b; Møller et al., 2004a). When cow manure was stored for a period of 2 months, biogas losses were of 30–40% (Fabbri and Piccinini, 2012) especially during summer. In farms with deep litter, faeces are not removed for long periods (months) and undergo complex degradation processes that can be either aerobic, where oxygen is available, or anaerobic (Tait et al., 2009). In both cases, a fraction of organic substance is converted into non-collectable carbon dioxide (aerobic process) or biogas (anaerobic process). It was observed that, in a litter of 6 months, the biogas potential of excreta was reduced by 40–50% if compared to the same fresh dairy manure (Fabbri and Piccinini, 2012). This loss can be partly balanced by the increased presence of straw in the litter (Garlipp et al., 2011), even if the high content of lignocellulosic compounds represents a strong limit for the biologic degradation of materials such as straw or maize stalks (Song et al., 2014).

Only few fragmentary and disaggregated data are available regarding the correlation between manure management system and methane production. Applied technologies are rarely specified in scientific publications. In addition, the few available data in literature are difficult to compare, since obtained under different operative conditions. Comprehensive works that compare the influence of housing system are therefore rare (Rigolot et al., 2010).

The aim of this research was to discuss the methane potential production of manure samples from different handling systems.

6.2 Materials and methods

6.2.1 Farms and sampling

Three commercial Holstein-Friesian farms located in Lombardy (in the north of Italy) were considered during an experimental campaign lasting 18 months. During this period, the average number of cows in the considered sections of each farm was around 90. In order to reduce the influence on the biogas production among farms, all the dairy cows were fed with the same diet. Dry matter supply was 21–23 kg/d, and different feedstuffs were used in order to satisfy the productive needs of cows during seasons. During cold seasons, cows were daily fed with 26–28 kg of corn silage, 5 kg of alfalfa (Medicago sativa), and 10 kg of concentrate (maize and/or soy flours) with a vitamin supplement. During warm seasons, cows were daily fed with 26–28 kg of corn silage, 3–4 kg of alfalfa (Medicago sativa), and 6 kg of concentrate (cotton and/or sugar beet seeds).

Samples of each effluent were taken twice a season, from summer 2012 to autumn 2013, for a total of 180 samples (6 per effluent per season). Samplings were carried out periodically on the same effluent every 4-8 weeks, according to the availability of free batch reactors for the determination of the Biochemical Methane Potential (BMP). Manure samples were taken directly during the clean-up operation, from the manure collection basin, depending on the technology installed in each farm. Each representative sample was obtained mixing two sub-samples of 3 litres. Samples were collected in 10-L plastic tanks and temporarily maintained (max. 24 hours) at a temperature of 4°C before use.

Farm 1 - Free stall dairy system equipped with scrapers. Cows were housed in a free-stall barn divided into feeding and resting zone. Two rows of head to head free stalls were located between the two areas. The feed alley (3.5×80 m) and the resting alley (3×80 m) were covered with a rubber mat pavement and equipped with scrapers for manure removal. Scrapers were used twice a day. At the end of the alleys, manures were collected in a catch basin. Crossover passages between alleys were placed every 15 stalls. Manure in those areas was removed during daily maintenance and was not considered in this work. Samples were collected at the end of the scraper run, before the discharge.

Farm 2 - Free stall dairy barn equipped with flushing system. The feeding (5x60 m) and the resting alley (3x60 m) of cows were in convex (1.5% slope) and inclined (3% slope) concrete, in order to facilitate cleaning. The flushing flow rate in the considered section of the farm was 0.15 $m^3 s^{-1}$. The flushing was carried out twice a day, usually at the time of milking, for ten minutes. The flush system utilized mainly recycled effluent from the manure separation system or occasionally water from the municipal water supply network. The flushed wastewater was collected directly to a primary storage basin. Samples were taken just before the discharge. Wastewater was then pumped to a screw press solid-liquid separator (5 kW, treating 25 $m^3 h^{-1}$ of slurry), and both the liquid and solid separated fractions were sampled.

Farm 3 - Free stall dairy system equipped with slatted floors. Each section of the housing module consisted of a feed alley (3.5x50 m), two rows of head-to-head free stalls, a resting alley (3x50 m), and a row of single free stalls. Free stalls (128 cubicles of 185x120 cm) were equipped with rubber mats and cleaned manually. The floor of the feeding and resting alleys was made of perforated concrete with holes of 3.5 cm. Slurries were collected by gravity in deep pits located under the floor. Each housing module had its own separate pit, and every pit was emptied cyclically according to a time interval that varied from 12 to 20 days (14 days was the most common time interval). Samples were collected while emptying. During summer, foam was often present on the surface of pits, but was not sampled.

6.2.2 Samples characterization

Total Solids (TS) and Volatile Solids (VS) were determined for each sample, according to Standard Methods (APHA et al., 2012). Analyses were carried out in triplicate. Biochemical Methane Potential (BMP) tests were performed using a custom experimental platform made up of 18 identical parallel lines at controlled temperature. Each line was equipped with a 5-L batch Plexiglas reaction tank coupled to a variable volume (max. 1 L) aluminium-polyethylene gas storage. Batch reactors were housed into thermally insulated containers (6 batches per container). Gas storages were connected in through automatic valves to a main unit equipped with a condenser to remove humidity, a drum counter for the volumetric measurement of the biogas (TGO5, Ritter Apparatebau Gmbh, Bochum, Germany) and a non-dispersion infrared/fuel cell (NDIR/ECD) gas analyser for oxygen, carbon dioxide and methane determination (Gasboard 3200 provided by WuHan Cubic Optoelectronics, Wuhan, China; the entire platform was assembled by Ambra Sistemi, Grugliasco, Italy). Biogas was automatically pumped from the storages to the analyser when the 80% of the maximum volume was reached. Results were automatically recorded on a PC. Reaction tanks were filled with 3 L of a mixture of inoculum and substrate. The mixture respected a 2:1 ratio between inoculum and substrate VS mass, in order to avoid any accumulation of fatty acids during the early days of digestion. The inoculum was obtained from the supernatant of the effluent of a mesophilic anaerobic digestion plant, operating with 50-days hydraulic retention time and treating dairy cow manure. The inoculum was filtered at 1 mm and kept at 40°C for 72 hours before use in order to remove the residual, easily biodegradable organic compounds. At least two batch reactors for each set of measurements were used as control, measuring the BMP of the inoculum. At the beginning of each test, the headspace of the reactors and the storages were washed with N_2 for 2 minutes at 2 bars, and then depressurized to -0.4 bars, in order to remove residual oxygen and to identify any leakage of the system. Then the internal pressure was equilibrated to atmospheric pressure at the incubation temperature of 40±0.5 °C. Temperature was continuously monitored and maintained constant through electric air heaters coupled with proportional-integral-derivative (PID) logic controllers. The reactors were incubated in the dark and mechanically stirred for a minute once a day. The incubation period lasts until the cumulated production of biogas had a daily marginal increase of less than 1% and, in any case, at least for 30 days.

6.2.3 Statistical analysis

Statistical analysis of the characteristics of different manures was carried out using SAS statistical software (SAS version 9.3; SAS Institute, Cary, NC, USA, 2012). Correlation analyses were carried out using the CORR procedure to study the relationship between type of manure and season as a function of TS, VS, and methane production. The same data were submitted to variance analysis (PROC GLM) to evaluate the seasonal effects. Methane production, TS and VS data were analysed using the ANOVA procedure (Waller-Duncan K-ratio t-test) to study the effect of the different manure handling systems.

6.3 Results and discussion

6.3.1 Characteristics of manures

Lactating cows produced about 50 kg of manure per day, corresponding to about 6 kg of dry matter per day (Total Solids, TS=12.0±1.3%, Volatile Solids, VS=78.1±3.3%, referred to the TS content of the manure sample; average values of samples taken during various seasons in the three farms). Table 6.1 shows TS and VS, representing the dry and organic matter content of manures; the results suggest a certain effect of handling techniques on manure characteristics. The most relevant comparisons are discussed, assuming scraping as reference point. In fact, scraping does not affect in substantial ways the characteristics of manure, since the collection is mechanical and very frequent.

Table 6.1. Total and volatile solids content of manure samples collected (%, mean value±standard deviation of the six samples taken during each season).

Effluents		Summer 2012	Autumn 2012	Winter 2013	Spring 2013	Summer 2013	Autumn 2013	Mean
Scraper	TS	14.0±1.2	13.5±0.9	11.6±1.5	12.5±1.1	13.6±1.5	13.6±1.8	13.2±0.9
	VS	79.3±2.1	80.7±3.0	83.1±2.7	76.3±2.1	75.1±1.1	83.8±2.5	77.8±4.5
Slatted floor	TS	8.5±0.9	13.5±1.2	11.1±1.1	11.2±1.0	10.8±0.9	12.0±1.0	11.1±1.6
	VS	71.9±1.5	73.5±2.1	73.4±2.0	72.1±2.3	71.1±1.8	72.1±0.9	73.0±1.4
Flushing (raw)	TS	2.3±0.5	2.3±0.6	2.8±0.5	3.0±0.6	2.9±0.3	3.0±0.4	2.3±0.5
	VS	73.2±1.8	75.1±2.5	73.9±1.5	70.1±1.9	73.0±1.7	73.1±1.8	71.9±2.3
Flushing (liquid fraction)	TS	1.9±0.3	2.0±0.7	1.8±0.4	2.3±0.5	2.5±0.4	2.6±0.5	2.2±0.3
	VS	65.2±1.8	63.0±1.9	62.5±1.5	67.0±1.2	69.7±1.7	61.6±2.1	64.8±3.0
Flushing (solid fraction)	TS	34.4±2.5	29.8±1.9	30.3±1.8	27.9±1.1	28.3±1.1	29.3±1.0	30.6±2.7
	VS	94.2±2.1	94.5±1.7	92.2±1.8	87.2±2.1	90.2±2.1	89.2±1.7	92.3±3.4

Slatted floor slurry had a lower content of TS ($p<0.001$) and VS ($p<0.001$) than scraped manure. These results were probably due to (at least) two causes: (i) slatted floor slurries remained for several days in the deep pit, where the rapidly biodegradable organic matter could be partly decomposed by heterotrophic and/or anaerobic bacteria and converted to gaseous products (foams were observed on the liquid surface); (ii) deep pits were not mixed, and this could have favoured sedimentation or floatation of solids (that are never removed during the usual operations of the farm, as the emptying of the pit was never complete).

Raw flushed manure was very diluted due to its origin, and acted like a liquid. Therefore, a TS comparison with other manures makes no sense. On the contrary, VS can be compared since expressed as referred to TS. In raw flushed manure, VS were lower than in scraped manure ($p<0.001$) probably because flushing process was operated by means of stabilised liquid fraction, which had a higher concentration of inert solids (VS=61.7±2.4%), as also observed by Wilkie et al. (2004). TS variability within seasons was relatively high with significant differences only between samples collected in summer 2012 and spring 2013. A probable cause was that the liquid fraction of manure was stored in an open tank utilized also for the storage of rainwater runoff, as commonly in many farms, producing anomalies in the characteristics of the fluid during washing operations. Solid-liquid separation operated differently on VS. In particular, it produced a solid fraction with a significantly increased VS concentration (up to an average value of 91.3±2.9%), and a liquid fraction with a reduced VS concentration (64.8±3.1%). This behaviour was already observed by the authors on other plants (unpublished data) and by others (Jørgensen and Jensen, 2009), and was probably due to the fact that organic solids are larger than inorganic (Levine et al., 1985). A clear trend of the characteristics of manures during seasons was not observed. Statistical analyses showed no significant differences ($p<0.05$) among seasons and total and volatile solids concentrations.

6.3.2 Biogas yield and methane content

Specific biogas productions from different manure handling systems are reported in Table 6.2, and are expressed as normal litres of methane produced per kg of VS subjected to anaerobic digestion. Little or no surface accumulation of solids was observed in samples before and during biochemical methane potential tests. Since biogas losses can be considered negligible after a few hours from excretion (Møller et al., 2004a; Kirk and Faivor, 2012), manure handling systems that allow a frequent collection, such as scraping and flushing, was expected to preserve the specific methane potential. Instead, significant differences ($p<0.001$) in the specific production of methane of flushed manure was observed. This was probably due, as discussed in the previous paragraph, to use of stabilised liquid separated fraction of slurry during flushing, that lower the specific production of methane. Statistical significant differences ($p<0.001$) can be observed among raw flushed manure, flushed liquid fraction and flushed solid fraction. This behaviour was probably due to the washing process and, in particular, to the separation process. Highly biodegradable VS appeared to be concentrated in the separated liquid fraction that had a high specific production of methane. This result suggests that the solid-liquid separation process did not distribute VS equally, but operated a selection: the most productive fraction of VS appeared to be contained in the liquid fraction. This result was already observed by other authors (e.g. Liao et al., 1984; El-Mashad and Zhang, 2010), that supported their findings considering the composition of the separated fraction. It was observed that fibrous (poorly degradable) compounds tend to accumulate in the separated solid fraction, lowering the specific production of methane. The valorisation of the liquid separated fraction cannot be performed in CSTR, since the low concentration of solids. Other authors, e.g. Wilkie et al. (2004) and Rico et al. (2007), obtained interesting results using fixed-film anaerobic digesters.

Table 6.2. Specific methane production (NL kg_{SV}^{-1}, mean value±standard deviation of the six samples taken during each season).

Effluents	Summer 2012	Autumn 2012	Winter 2013	Spring 2013	Summer 2013	Autumn 2013	Mean
Scraper	175±22	188±12	177±25	193±34	192±12	183±23	185±22
Slatted floor	152±14	160±11	166±17	161±23	168±23	168±24	162±19
Flushing (raw)	174±15	129±12	163±15	186±31	173±21	188±30	169±26
Flushing (liquid fraction)	193±28	205±29	200±22	209±21	209±35	217±37	205±28
Flushing (solid fraction)	141±16	145±27	139±31	155±23	144±11	156±32	147±27

Slatted floor manure was expected to produce a lower amount of methane. In fact, the observed specific production was 162±19 NL_{CH_4} kg_{VS}^{-1}, significantly ($p<0.001$) lower in comparison with scraped manure (185±22 NL_{CH_4} kg_{VS}^{-1}). As previously discussed, this was probably due to the long retention time in the deep pit. The slow but constant production of small bubbles of gas and foam was always observed in the deep pit. However, when considering the concurrent reduction of VS in the slatted floor manure (Table 6.1), we observed a more pronounced depletion in the methane yield. If the methane production is expressed as a function of TS (in order to include in the analysis also the variation of VS), scraper and slatted floor manure produced 144±17 and 118±14 NL_{CH_4} kg_{TS}^{-1}, respectively. This aspect is not clearly visible if only specific biogas production (referred to the mass of VS) is considered. Nevertheless, when a substantial change in the characteristics of the solids occurs (especially when dealing with a transformation of similar manures), the specific production could be a misleading parameter during a farm scale evaluation. For example, considering negligible the effect of evaporation in the deep pit during the period of storage (Costa et al., 2015), the calculated methane yield of 1 kg of raw scraped manure was 18.8±4.3 NL_{CH_4}, while for 1 kg of slatted floor manure was 13.2±3.3 NL_{CH_4}.

The methane content in biogas is reported in Table 6.3. The values remained between 50 and 58%, an interval that is comparable with that in the literature (Hill, 1984; Møller et al., 2004b; El-Mashad and Zhang, 2010). Again, the lower values were observed in manure that was partially stabilised (slatted manure). No evident seasonal effects were observed.

Table 6.3. Methane concentration in biogas (%, mean value±standard deviation of the six samples taken during each season).

Effluents	Summer 2012	Autumn 2012	Winter 2013	Spring 2013	Summer 2013	Autumn 2013	Mean
Scraper	54.1±1.1	53.4±0.8	52.0±1.0	53.8±0.9	56.5±1.1	52.5±0.5	53.7±1.6
Slatted floor	48.1±0.9	48.6±1.2	50.9±0.8	53.2±1.0	55.5±0.7	51.3±1.3	51.3±2.8
Flushing (raw)	57.3±1.0	55.1±0.8	55.5±1.1	57.0±1.3	55.3±0.8	56.4±1.1	56.1±0.9
Flushing (liquid fraction)	50.3±0.7	49.1±1.3	51.3±1.0	51.0±1.1	49.7±0.8	50.8±1.1	50.4±0.4
Flushing (solid fraction)	56.9±0.9	55.8±1.2	60.5±1.1	57.4±1.1	59.3±0.9	58.7±1.1	58.1±1.7

Table 6.4 reports the Waller grouping from the ANOVA procedure, describing the statistical differences among different technologies and parameters, and supporting the previous discussion.

Table 6.4. Waller grouping of different technologies and parameters.

Effluents	TS	VS	Methane production	Methane concentration
Scraper	B	B	B	C
Slatted floor	C	C	C	D
Flushing (raw)	D	C	C	B
Flushing (liquid fraction)	D	D	A	D
Flushing (solid fraction)	A	A	D	A

In general it should be considered that some minor differences among manures can probably be explained by other factors like feedstuff quality, genetic variety, conservation, microclimate, geopedology and soil structure of the areas where feedstuffs were produced, which can slightly influence the amount of undigested residuals even if the amount of feed was constantly monitored.

6.3.3 Energy consumption

Different manure handling techniques requires the installation and operation of different technologies. The scrapers were moved by two 3 kW electrical engines, twice a day (overall operation time: 80 minutes). The daily consumption of energy was 4 kWh. Assuming an average live weight (LW) of 700 kg cow^{-1}, the daily specific consumption of energy can be estimated at 65 Wh t_{LW}^{-1}. Flushing was operated through a centrifugal pump of 15 kW, twice a day (overall operation time: 20 minutes). The overall flushed manure (300 m^3 d^{-1}) was then treated in a screw press solid-liquid separator (5 kW, operated 12 hours per day). The overall daily consumption was estimated at 65 kWh. Since the farm was subdivided into two barns, and we considered only one of them, the daily energy consumption of the studied section was 32.5 kWh. Therefore, the daily specific consumption of energy can be estimated at 515 Wh t_{LW}^{-1}, which is a much higher value with respect to scraper. It can also be observed that the management of the flushing process is discretional (see, for example, the brief review reported in Wilkie et al., 2004, where it is highlighted that differences of 2-4 times in flow rates are possible among farms with similar characteristics). Therefore, the value obtained in the present study should be considered as site-specific, even if the flushing can in any case considered as a technology with a high-energy and water consumption. In the studied farm, in fact, an average water consumption of ~2,500 L t_{LW}^{-1} was calculated, and can be compared with other literature values (e.g. 2,260 L t_{LW}^{-1}, Williams and Frederick, 2001; 935 L t_{LW}^{-1}, Chastain et al., 2001; 4,000 L t_{LW}^{-1}, Kay Camarillo et al., 2012).

Slatted floor handling system did not require any specific device, since it is based on gravity. The energy consumption for the transport of manure to the storage was not considered here, as the pump was operated every two weeks and the specific energy consumption was negligible. These values can slightly be varied as a function of the dimension of the farm, but the proportion between them should remain quite constant.

6.4 Conclusions

Manure handling can have an effect on the overall energetic balance of anaerobic digestion process. Scraping appears to be the most effective technology, as it doesn't significantly affect the characteristics of manure (that is adequate to be digested "as is") nor its energy content, and requires a minimal energetic consumption for collection. Slatted floor is a simpler technology that does not require the operation of any specific equipment, but a significant loss of methane can occur during the period of storage of manure in the deep pit. Finally, flushing requires much more energy than the other technologies, and the liquid fluxes produced are not fit to be directly introduced in the digesters commonly installed in Europe (mesophilic, wet technologies, CSTRs), since too diluted. The relatively high specific methane production of the liquid separated fraction could suggest its utilisation in other types of reactors, such as fixed film anaerobic digesters, even if the low solid concentration remains a problem. The solid separate fraction from flushing tends to accumulate the VS with the lower methane potential and therefore could be considered as suitable co-substrate only under particular circumstances, such as the adjusting of the humidity. In general, flushing appears to be a technology scarcely compatible with conventional anaerobic digestion processes: the unavoidable dilution makes the characteristics of the slurry unfit to be conveniently converted into methane.

References

APHA, AWWA, WEF 2012. Standard Methods for the Examination of Water and Wastewater - 22nd Edition. Alexandria, VA, USA.

Appels L., Lauwers J., Degrève J., Helsen L., Lievens B., Willems K., Van Impe J., Dewil R. 2011. Anaerobic digestion in global bio-energy production: Potential and research challenges. Renew. Sust. Energ. Rev. 15:4295-301.

Chastain J.P., Vanotti M.B., Wingfield M.M. 2001. Effectiveness of liquid-solid separation for treatment of flushed dairy manure: a case study. Appl. Eng. Agric. 17:343-54.

Costa A., Gusmara C., Gardoni D., Tambone F., Guarino M. 2015. The effect of anaerobic digestion and storage on indicator microorganism inswine and dairy manure. T. ASABE Submitted:

El-Mashad H.M., Zhang R. 2010. Biogas production from co-digestion of dairy manure and food waste. Bioresour. Technol. 101:4021-28.

Fabbri C., Piccinini S. 2012. Bovini da latte e Biogas. Linee guida per la costruzione e la gestione di impianti. C.R.P.A., Reggio Emilia, Italy.

Fritsche U.R., Sims R.E.H., Monti A. 2010. Direct and indirect land-use competition issues for energy crops and their sustainable production – an overview. Biofuels Bioprod. Biorefin. 4:692-704.

Garlipp F., Hessel E.F., van den Weghe H.F.A. 2011. Characteristics of Gas Generation (NH3, CH4, N2O, CO2, H2O) From Horse Manure Added to Different Bedding Materials Used in Deep Litter Bedding Systems. J. Equine Vet. Sci. 31:383-95.

Gissén C., Prade T., Kreuger E., Nges I.A., Rosenqvist H., Svensson S.-E., Lantz M., Mattsson J.E., Börjesson P., Björnsson L. 2014. Comparing energy crops for biogas production – Yields, energy input and costs in cultivation using digestate and mineral fertilisation. Biomass Bioenerg. 64:199-210.

Gopalan P., Jensen P.D., Batstone D.J. 2013. Biochemical Methane Potential of Beef Feedlot Manure: Impact of Manure Age and Storage. J. Environ. Qual. 42:1205-12.

Hill D.T. 1984. Methane Productivity of the Major Animal Waste Types. Trans. ASAE 27:530-34.

Holm-Nielsen J.B., Al Seadi T., Oleskowicz-Popiel P. 2009. The future of anaerobic digestion and biogas utilization. Bioresour. Technol. 100:5478-84.

Jørgensen K., Jensen L.S. 2009. Chemical and biochemical variation in animal manure solids separated using different commercial separation technologies. Bioresour. Technol. 100:3088-96.

Kay Camarillo M., Stringfellow W.T., Jue M.B., Hanlon J.S. 2012. Economic sustainability of a biomass energy project located at a dairy in California, USA. Energy Policy 48:790-98.

Kirk D., Faivor L. 2012. The impact of dairy housing and manure management on anaerobic digestion. Got manure? Enhancing Environmental and economic Sustainability conference agenda, Liverpool, New York, 34-42.

Larney F.J., Buckley K.E., Hao X., McCaughey W.P. 2006. Fresh, Stockpiled, and Composted Beef Cattle Feedlot Manure. J. Environ. Qual. 1844-54.

Levine A.D., Tchobanoglous G., Asano T. 1985. Characterization of the size distribution of contaminants in wastewater: treatment and reuse implications. Journal (Water Pollution Control Federation) 805-16.

Liao P.H., Lo K.V., Chieng S.T. 1984. Effect of liquid—solids separation on biogas production from dairy manure. Energy in Agriculture 3:61-69.

Martinez J., Dabert P., Barrington S., Burton C. 2009. Livestock waste treatment systems for environmental quality, food safety, and sustainability. Bioresour. Technol. 100:5527-36.

Meyer D., Price P.L., Rossow H.A., Silva-del-Rio N., Karle B.M., Robinson P.H., DePeters E.J., Fadel J.G. 2011. Survey of dairy housing and manure management practices in California. J. Dairy Sci. 94:4744-50.

Møller H.B., Sommer S.G., Ahring B.K. 2004a. Biological Degradation and Greenhouse Gas Emissions during Pre-Storage of Liquid Animal Manure. J. Environ. Qual. 33:27-36.

Møller H.B., Sommer S.G., Ahring B.K. 2004b. Methane productivity of manure, straw and solid fractions of manure. Biomass Bioenerg. 26:485-95.

Moset V., Cambra-López M., Estellés F., Torres A.G., Cerisuelo A. 2012. Evolution of chemical composition and gas emissions from aged pig slurry during outdoor storage with and without prior solid separation. Biosys. Eng. 111:2-10.

Rico J.L., García H., Rico C., Tejero I. 2007. Characterisation of solid and liquid fractions of dairy manure with regard to their component distribution and methane production. Bioresour. Technol. 98:971-79.

Rigolot C., Espagnol S., Robin P., Hassouna M., Béline F., Paillat J.M., Dourmad J.Y. 2010. Modelling of manure production by pigs and NH3, N2O and CH4 emissions. Part II: effect of animal housing, manure storage and treatment practices. Animal 4:1413-24.

Song Z., GaiheYang, Liu X., Yan Z., Yuan Y., Liao Y. 2014. Comparison of Seven Chemical Pretreatments of Corn Straw for Improving Methane Yield by Anaerobic Digestion. PLoS ONE 9:e93801.

Tait S., Tamis J., Edgerton B., Batstone D.J. 2009. Anaerobic digestion of spent bedding from deep litter piggery housing. Bioresour. Technol. 100:2210-18.

Wilkie A.C., Castro H.F., Cubinski K.R., Owens J.M., Yan S.C. 2004. Fixed-film Anaerobic Digestion of Flushed Dairy Manure after Primary Treatment: Wastewater Production and Characterisation. Biosys. Eng. 89:457-71.

Williams D.W., Frederick J.J. 2001. Microturbine operation with biogas from a covered dairy manure lagoon. ASAE Meeting Presentation, Paper, 016154.

CHAPTER 7

7 General discussion and conclusions

7.1 Discussion

This thesis is particularly dedicated to a comprehensive evaluation of manure handling solutions commonly adopted in the Po Valley area. The investigation was conducted at several levels, taking into account different aspects related to the applied housing solutions: their influence on GHG and NH_3 emissions; a Life Cycle approach for the quantification of their environmental performance; the implication of the selected strategies in the downstream anaerobic digestion process.

The review performed in Chapter 3 allowed a deep analysis of the literature related to LCA applied to milk production. In particular, some interesting aspect emerged, such as the statistically significant differences resulting from the choice of different functional units (FPCM or ECM) or the importance of conducting a sensitivity analysis to understand the reliability of the estimated burdens and identify the most relevant input parameters affecting the LCA outcomes. The difficulties in comparing results obtained from different studies, strongly related to the practitioners' decisions, underlined the need of a wider level of standardization among procedures. The recent guidelines on environmental performance of large ruminant drawn up by Livestock Environmental Assessment and Performance Partnership (FAO, 2016), confirmed this shortage of harmonization and hopefully will fill this gap in the future.

Chapter 4 was dedicated to the measurement of GHG and NH_3 from different housing solutions widely spread in dairy farms of the Po Valley. The results of this study showed that shed areas within the barns have different emissive patterns: feeding alleys are major sources of NH_3 emissions, while GHG are predominantly emitted from straw-cubicles. The adopted manure removal system affects the emissions from the barns, for the micro-conditions that floor type creates in the temporarily indoor-stored manure. Furthermore, some trade-offs among different gases emerged. Scrapers constitute a good choice to control GHG, but increase NH_3 emissions as a consequence of the urine spreading and of the increased air-exchanging surface. Otherwise, since urine is quickly drained-off via openings, slatted floors reduce NH_3 but promote the CH_4 emissions, fostering anaerobic conditions in the pit underneath the pavement.

Results obtained in Chapter 4 were then used for a broader analysis of the environmental burdens associated to dairy farm as a whole. In Chapter 5 a full LCA study was conducted to compare emission inventories compiled with measured or estimated emission factors for the manure management phase. Findings outlined differences between measured and estimated approach, highlighting the need for more flexible and precise emission factors, in particular for GHG estimation. Local conditions (e.g. temperature) and manure characteristics (e.g. volatile solids) strictly affect gaseous emissions (Aguirre-Villegas et al. 2017). It would be worth to develop suppler emission factors, able to take into account local conditions, in order to improve estimates and thus environmental assessment of single farms. Furthermore, a special care of replacement animals, with proper housing and feeding choices, could be crucial to improve the sustainability of milk production.

In the last few decades, anaerobic digestion plants constituted a win-win-win solution for dairy farms: they have contributed to achieve climate change mitigation, they are renewable energy sources using manure to produce biogas, and have increased farm profitability. The biogas potential contained in manure samples obtained from farms with different manure handling systems were evaluated in Chapter 6. Trade-offs between biogas yield and energy consumption were also considered. Scraper technologies showed a high level of biogas production with low energy requirements. It assures a frequent and efficient removal of manure, without altering its physical and chemical properties, providing a suitable matter for anaerobic digestion plants. Otherwise, flushed manure, although has registered the highest values in the Biochemical Methane Potential (BMP) tests, does not fit to be directly introduced in digesters commonly installed in Europe since too diluted. Furthermore the flushing requires a higher energy amount.

As emerged from this thesis, manure handling tactics constitute an important choice for the reduction of livestock environmental impacts on air. Floor construction has a significant effect on GHG and NH_3 emissions and the suggestions collected here could be useful to develop guidelines driving incentive mechanisms for new buildings or refurbishment of dairy barns in the European context. In particular, the prompt cleaning of the flooring surfaces with scrapers (better if running on rubber flooring to achieve higher efficiency) seems a good compromise between environmental protection (removing quickly the emission sources from the barns) and the achievable biogas yield. However, a part with these investment choices, also the good daily-management of the farms (i.e. modulating the frequency of scraping, frequency of renewing cubicles, etc.) is very important for the improvement of the farm environmental sustainability.

Synergies and antagonistic effects were observed among different gaseous emissions both in the barns and in other steps of the manure management continuum (i.e. anaerobic digestion reduces GHG but increases NH_3 emissions). Thus becomes clear that the modulation of environmental impacts of livestock farming can succeed only through the adoption of the proper combination of measures to control the whole-chain GHG and NH_3 emissions (Hou et al., 2015).

7.2 Further development

As emerged in Chapter 5, measured emissions are fundamental to provide accurate indications on the actual environmental burdens of dairy farms. Even considering the possible errors associated to measurements, the opportunity to substitute an estimated value with a measured one assumes wider relevance in driving the choice of mitigation options to implement. For this reason, it would be worth continuing this research line. An approach to verify could be the substitution of B_0 default values proposed by IPCC with those obtained in Chapter 6 from BMP tests. Furthermore, particular attention should be paid to storage emissions, searching for value of emissions measured in the Italian context.

Within this thesis framework, it is even more clear that mitigation strategies should consider the whole-chain of manure management continuum in order to be successful. The quantification of the environmental benefits achievable implementing mitigation options at several levels of the manure management continuum deserves attention.

Finally, to include circular economy ideas in the farm management becomes an important strategy. With currently available technologies, such as anaerobic digestion, manure is re-turning from being a waste to a valuable co-product of the dairy system. LCA studies have so far considered that manure is a waste and thus, has no environmental impacts associated to its production (all environmental impacts are assigned to the main product of milk). Future LCA studies could model the farm system considering manure as a co-product.

7.3 Main conclusions

- Assure a prompt cleaning of the flooring surface of the barns is the most effective way to control GHG and NH_3 emission levels. The results of this thesis indicate that the use of scrapers can be a win-win solution in terms of emission reductions and yield of the anaerobic digestion process.

- GHG and NH_3 emissions are influenced by different parameters, and trade-offs are observed between solutions proposed for their control. A combination of solutions is often needed to control the whole-chain GHG and NH_3 emissions.

- Little is known about lifecycle impacts of manure management and processing practices. This thesis provides an attempt to compare the emissive patterns of different housing solutions for dairy farms. However, the LCA framework could help in identifying the best management practices only if it is able to catch differences also in the manure management continuum. Results of this thesis underline that further research on this topic is merited.

References

Aguirre-Villegas H. A., Larson R. A., 2017. Evaluating greenhouse gas emissions from dairy manure management practices using survey data and lifecycle tools. Journal of Cleaner Production; 143: 169-179.

Hou Y., Velthof G. L., Oenema O., 2015. Mitigation of ammonia, nitrous oxide and methane emissions from manure management chains: a meta- analysis and integrated assessment. Global Change Biology; 21: 1293-1312.

FAO, 2016. Environmental performance of large ruminant supply chains: Guidelines for assessment. FAO, Rome, Italy.

List of publications

7.4 Publications in peer review journals

Coppolecchia, D., Gardoni, D., Baldini, C., Borgonovo, F., Guarino, M., 2015. The influence on biogas production of three slurry-handling systems in dairy farms. Journal of Agricultural Engineering 46(1), 30-35. doi:10.4081/jae.2015.449.

Baldini, C., Borgonovo, F., Gardoni, D., Guarino, M., 2016. Comparison among NH3 and GHGs emissive patterns from different housing solutions of dairy farms. Atmospheric Environment 141, 60-66. doi: 10.1016/j.atmosenv.2016.06.047.

Baldini, C., Gardoni, D., Guarino, M. 2017. A critical review of the recent evolution of Life Cycle Assessment applied to milk production. Journal of Cleaner Production 140, 421-435. doi: 10.1016/j.jclepro.2016.06.078.

Baldini, C., Bava, L., Zucali, M., Guarino, M. 2017. Milk production Life Cycle Assessment: a comparison between estimated and measured emission inventory for manure handling. Science of the Total Environment 625, 209-219. doi: 10.1016/j.scitotenv.2017.12.261.

7.5 Publications in conference proceedings

Baldini, C., Beretta, F., Colosio, C., Guarino, M., 2015. Exposure to organic dust in a farrowing-weaning farm as a possible risk for human and pigs: One Health Approach. Proceeding of “International congress on Rural Health as a social, economic and cultural engine”, Lodi, 8-11 September 2015.

Baldini, C., Gardoni, D., Guarino, M., 2015. The evolution of LCA in milk production. Proceedings of “LCA for Feeding the planet and energy for life- International conference on Life Cycle Assessment as reference methodology for assessing supply chains and supporting global sustainability challenges”, Stresa, 6-7 October 2015, edited by Scalbi, S., Dominici Loprieno, A., Sposato, P. published by ENEA, p 258-261. ISBN:978-88-8286-321-0.

Baldini, C., Gardoni, D., Zucali, M., Bava, L., Guarino, M., 2016. Gas Emissions from Manure Management: Measurements vs Estimates. Proceedings of “10th International Conference on Life Cycle Assessment on Food 2016”, Dublin, 19-21 October 2016.

Baldini, C., Borgonovo, F., Tullo, E., Guarino, M., 2017. Environmental and economic evaluation of slaughterhouse waste used as source of biomass for energy production. Proceedings of “Emissions of Gas and Dust from Livestock”, Saint-Malo, 21-24 May 2017.

Baldini, C., Borgonovo, F., Tullo, E., Fontana, I., Guarino, M., 2017. Greenhouse Gas Emissions from Dairy Farms with Different Manure Management. Proceedings of “2017 International Symposium on Animal Environment and Welfare”, Chongquing, 23-26 October 2017.

Abbreviations and acronyms

AD	Anaerobic Digestion
ANOVA	Analysis Of Variance
AP	Acidification Potential
BMP	Biochemical Methane Potential
CF	Carbon Footprint
CH_4	methane
CLRTAP	Convention on Long-Range Transboundary Air Pollution
CO_2	carbon dioxide
CSTR	Completely Stirred Reactors
ECM	Energy Corrected Milk
EEA	European Environmental Agency
EF	Emission Factor
EMEP	European Monitoring and Evaluation Programme
EP	Eutrophication Potential
EU	Energy Use
FAO	Food and Agriculture Organization
FPCM	Fat and Protein Corrected Milk
FSS	Farm Structure Survey
FU	Functional Unit
GDP	Gross Domestic Product
GHG	Greenhouse gases
GWP	Global Warming Potential
IDF	International Dairy Federation
ILCD	International Reference Life Cycle Data System
IPCC	Intergovernmental Panel on Climate Change
ISO	International Standards Organisation
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment

LEAP	Livestock Environmental Assessment and Performance Partnership
LSU	Livestock Unit
LU	Land Use
LW	Live Weight
MCS	Monte Carlo Simulation
N	nitrogen
N_2	molecular nitrogen
N_2O	nitrous oxide
NH_3	ammonia
N_L	Normal liter
NOx	nitrogen oxides
PM	Particulate Matter
SETAC	Society of Environmental Toxicology and Chemistry
S-LCA	Social LCA
TS	Total Solids
UAA	Utilized Agricultural Area
USEPA	United States Environmental Protection Agency
VS	Volatile Solids

Ringraziamenti

Innanzitutto grazie al prof. Navarotto, senza il quale tutta questa storia non sarebbe mai iniziata. Anche se ormai non “bazzica” più in Università vorrei che sapesse che nutro una grande stima nei suoi confronti, per la passione e l’onestà con cui ha sempre svolto il suo lavoro.

Alla mia tutor prof. Guarino, che da sempre si lascia chiamare Marcella. Per la sua capacità di vedere sempre il bicchiere mezzo pieno, che va a braccetto con il pessimismo cosmico che mi pervade quando mi vedo costretta a fare affidamento sulle mie capacità. Grazie per questa opportunità, per avermi accompagnato e spronato, per avermi spinto oltre, ma anche per aver rispettato i miei tempi “da mamma”.

Alle mie colleghe, senza le quali avrei abbandonato molto prima di raggiungere la meta: Federica, la Bipolare, Ilaria, la Bonton….distaminkia!, Emanuela, la Tuttofare. Ad onor di cronaca riporto anche il nickname della sottoscritta, la Scombinata. Grazie per il clima sereno che si respira in ufficio e per la vostra amicizia sincera.

A Maddalena&Luciana per la loro pazienza, con le quali non finirò mai di scusarmi per il mio essere casinara.
Alle persone conosciute in questo percorso: Margherita con cui ormai ci diamo del tu, il prof. Provolo per i suoi consigli, il diversamente Jacopo, Valentina, Lorenzo, Susanna, Guido. E poi gli irlandesi: il prof. Holden, Ming Jia, Tom, Paul, Jeremy, Yuan-Yuan e Qing Song, Joe, Rachel, Damién, Friedrich, Whenao, Pooja.

A mamma e papà. Per il loro inesauribile amore. Se da grande ne avessi anche solo un decimo di quello che avete voi da regalare, credo che sarebbe sufficiente.
Al resto della tribù, che nomino perché voglio che sappiano che per me sono tutti importanti. Paola, Roberto, Enrico, Chiara, Silvia, Maurizio. Grazie per aver creduto più di me in questa possibilità e per aver dato man forte alla famiglia con il servizio

baby-parking. Ai nipotozzi Pietro, Fede, Franci, Agnes, Tommy, Lori, AliMoni. Grazie perché da sempre è bello crescere insieme a voi.

Ad Angela e Piero che silenziosamente sopportano le innumerevoli spigolature del mio carattere.

Al mio super Zacca. Grazie semplicemente perché ci sei e rendi semplice la mia vita, che altrimenti altro che UCCC (Ufficio Complicazione Cose Complicate). Grazie perché so che mi ami e su questa certezza si fonda tutta la mia vita. Grazie anche perché mi sopporti, cosa non facile quando hai una tesi da scrivere.

A quei pezzettini di cuore che sono Teresa, Luigi e Giacomo. L'unico motivo per cui vale la pena provare ad essere ogni giorno più forte.

Annex I

Supplementary Materials of Chapter 4

Table SM 1. Results of the Kruskal-Wallis test for the feeding alley and the resting area. Further analysis are applied only if the p-value of the Kruskal-Wallis test resulted lower than 0.05.

Gas	Feeding alley	Resting area
NH_3	<0.0001	0.1387
N_2O	<0.0001	<0.0001
CH_4	<0.0001	<0.0001
CO_2	0.0007	<0.0001

Table SM 2. Results of the Steel-Dwass-Critchlow-Fligner test for the feeding alley.

NH_3	Farm 1	Farm 2	Farm 3	Farm 4
Farm 1	1	0.039	0.001	0.042
Farm 2		1	<0.0001	1
Farm 3			1	<0.0001
Farm 4				1
N_2O	Farm 1	Farm 2	Farm 3	Farm 4
Farm 1	1	<0.0001	<0.0001	<0.0001
Farm 2		1	0.045	0.866
Farm 3			1	0.260
Farm 4				1
CH_4	Farm 1	Farm 2	Farm 3	Farm 4
Farm 1	1	0.407	0.064	0.000
Farm 2		1	0.494	0.001
Farm 3			1	0.502
Farm 4				1
CO_2	Farm 1	Farm 2	Farm 3	Farm 4
Farm 1	1	0.664	0.008	0.199
Farm 2		1	0.006	0.404
Farm 3			1	0.059
Farm 4				1

Table SM 3. Results of the Steel-Dwass-Critchlow-Fligner test for the resting area.

N_2O	Farm 1	Farm 2	Farm 3	Farm 4
Farm 1	1	<0.0001	0.430	<0.0001
Farm 2		1	<0.0001	0.165
Farm 3			1	<0.0001
Farm 4				1
CH_4	Farm 1	Farm 2	Farm 3	Farm 4
Farm 1	1	0.226	0.002	0.001
Farm 2		1	0.000	0.012
Farm 3			1	<0.0001
Farm 4				1
CO_2	Farm 1	Farm 2	Farm 3	Farm 4
Farm 1	1	<0.0001	0.563	<0.0001
Farm 2		1	<0.001	0.003
Farm 3			1	<0.0001
Farm 4				1

Table SM 4. Results of Wilcoxon test between clean and dirty surfaces of the feeding alley.

Farm	**NH_3**	**N_2O**	**CH_4**	**CO_2**
2	0.3769	0.5738	0.0149	0.0673
4	0.7596	0.0105	0.0065	0.1017