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Environmental impact of livestock activities in the Po Valley and mitigation pathways evaluation

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Abstract

In Italy, livestock production is of great importance within the agricultural sector, driven by social, economic and cultural factors, with the Po Valley hosting approximately 80% of the national dairy and beef cattle, and pig populations. The link between animal production and the environment is increasingly recognized as a critical issue, requiring a thorough understanding of their interaction.

This thesis aimed to evaluate different mitigation strategies to reduce on-farm emissions of greenhouse gases and air pollutants, specifically examining air treatment technologies in pig housing facilities and some waste management measures for the cattle sector. Focus was made on the two above environmental aspects due on the one hand to the strong relationship that these have with the sector under analysis and on the other to their relevance in the Italian and the broader European context. To provide a comprehensive understanding of these issues, the approach included (i) direct measurements during field campaigns on farms to assess emissions; (ii) life cycle assessment (LCA) methodology using primary production data collected through surveys, complemented by secondary data from literature and estimation models; and (iii) large-scale modeling based on regional/national statistics and general activity factors, to estimate the potential resulting from widespread adoption of mitigation practices.

The relationship between livestock in Italy and air pollution, with a focus on ammonia emissions, was explored through three different studies. The first focused on the evaluation of the evolution of atmospheric concentrations of ammonia (NH_3) in Northern Italy during the COVID-19 pandemic. Ground-based and satellite data were analyzed for the year 2020 and compared with data from previous years, and it was concluded that, unlike some other air pollutants, the anti-COVID-19 measures affecting human activities did not lead to a reduction in NH_3 air concentrations, confirming its strong relationship with agriculture, whose activities were not significantly interrupted during the pandemic.

The other two studies focused instead on the adoption of abatement strategies in pig farms to reduce ammonia and particulate emissions from housing facilities. This phase of the agricultural cycle is very influential on the total emissions of these pollutants, but mitigation technologies aimed at it are not widespread in the Italian context. LCA has been used to

quantify the environmental impact of pig production in a "standard" production context and to compare it with the production when implementing two air treatment technologies aimed at reducing pollutant emissions: i) wet scrubber; ii) dry scrubber. Inventories have been made of the technical performances of farms in Italy and Spain. Operation of the two technologies have been obtained from experimental trials carried out during the Life-MEGA project. Both technologies tested showed their potential to reduce emissions in the pig housing phase, which had an impact on all the categories affected by air pollutant emissions, such as particulate matter formation, acidification and eutrophication. At the same time, different trade-offs were observed with impact categories related to energy and resource use. The dry scrubber was found to be the most favorable option when considering the balance with the trade-offs.

The third study explored the effect that the large-scale implementation of air cleaning technologies (with a focus on wet acid scrubbers) for pig housing facilities could have in and endpoint perspective in the European Union. Emissions related to the housing stage of NH_3, PM10, NMVOC and indirect N_2O from large pig farms (>1000 heads of sows or fattening pigs) were first estimated in the actual situation (current scenario - CS), considering implementation rates and removal efficiencies of the different emission abatement techniques available. Subsequently, alternative scenarios were simulated with a growing implementation rate of the wet acid scrubber (35% and 65% of the concerned pig farms in all Member States). In conclusion, scrubber technologies are promising for mitigating the environmental impacts of pig production and the emission reduction achievable with increased implementation rates across the EU would have a largely positive endpoint effect on human health, and lead to significant alleviation of current environmental costs on society of air pollution related to intensive pig farming. Future research should focus on more issues related to these technologies, including optimizing cost-effectiveness, impact on animal welfare and production performance, on farm worker working conditions, and effluent management.

Subsequently, two studies were developed focused on the analysis of potential strategies for reducing greenhouse gas emissions from cattle farming waste management, one on the dairy and one on the beef sector. For the dairy sector, a direct measurement approach was adopted, through a campaign that included an on-farm trial lasting several months, adopting slurry treatment through the commercial additive SOP Lagoon and specific technologies for on-site control of emitted gases from slurry storage tanks. Although the effectiveness of emission reduction varied throughout the trial, the results were positive, particularly for methane, as three months after the first applications of the additive, the treated storage tank showed statistically significant lower emissions than the untreated tank.

As for the beef cattle sector, on the other hand, a case study was developed to analyze a beef cattle farm integrating on-farm renewable energy production technologies by combining life cycle assessment and emissions modeling. The implementation of anaerobic digestion system on farm is shown to considerably enhance the environmental and energy performance of beef production. This study, to the authors' knowledge, was the first to quantify the possible environmental benefit deriving from such integrated management of livestock and energy production on a beef farm.

Overall, this work has provided valuable context-specific insights into the technologies evaluated, highlighting the benefits and limitations of their application. It goes without saying that none of them can fully address the environmental challenges of animal production if implemented in isolation, as integrated interventions are required.

LCA was confirmed as a valuable tool for identifying areas for improvement within the value chain and for highlighting trade-offs, ensuring that improvements in one aspect of a product's life cycle do not inadvertently shift environmental burdens to another phase or impact category. However, due to some of its limitations discussed in the course of the work, it cannot stand alone in guiding decision making and environmental policy development. This again underlines the importance of a holistic approach to assessing the environmental impacts of livestock production, as a key challenge is to capture the complexity and variability of farming systems where local practices and conditions vary widely.

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CHAPTER 1 - Introduction

1.1. Brief overview of the livestock sector in Italy

In Italy, livestock play an important role in the agricultural sector for social, economic and cultural reasons. They provide food security, high value-added typical products, landscape conservation, rural development and contribute to energy supply through renewable agro-energy chains. Livestock supply chains are spread throughout the country, albeit with differences in terms of species reared, production systems and productivity. The main production area in quantitative and economic terms is the Po Valley, a vast plain that stretches from west to east for 650 km from the western Alps to the Adriatic Sea, crossing all the northern regions of the country. This area hosts a large part of the Italian livestock and is characterized by highly specialized intensive supply chains (Zucali et al., 2017; Lovarelli et al., 2020). In particular, the regions of Lombardy, Emilia-Romagna, Piedmont and Veneto host around 80% of national dairy and beef cattle and pigs.

As regards the dairy sector, in 2020 the total number of dairy cows in milk in Italy was around 1.5 million, and the number of total dairy cattle (including heifers and calves) around 2.7 million, distributed in 25,915 farms. The national production of cow's milk exceeded 13 million tons in 2021, the sector is well developed and achieves self-sufficiency rates above 80%. In 10 years from 2010 to 2020 there has been a significant reduction in the number of farms, almost 11,000 units, while the number of animals has remained stable, and production has steadily increased. In practice, in parallel with what happens in the rest of Europe, the dairy sector is experiencing a significant phenomenon of production facilities concentration, with the average size going from 78 to 106 heads per farm in the period 2012-2021. As already mentioned, farms are mostly concentrated in the northern regions of the country, with Lombardy which alone has 5,392 dairy farms and 45% of national milk production (ISMEA, 2022). Important DOP (Denomination of Protected Origin) cheese products are produced by the dairy industry of the Po Valley regions (such as Grana Padano, Parmigiano Reggiano, Gorgonzola, Taleggio, etc.) and represent an essential cultural heritage of this area, as well as a main destination of the milk produced.

Beef cattle farming represents only about 4% of the agro-industrial turnover and suffers from a strong dependence on imports, with a degree of self-sufficiency of about 50%.

Nevertheless, the sector is well structured, involves many stakeholders and is widespread throughout the country. In 2019, there were approximately 94.6 thousand farms specialized in this production, with a total of 2.635 million animals slaughtered per year. Moreover, the number of animals reared is increasing (+ 8.6% of the beef cattle population in 2015-2020), although the apparent per capita consumption of beef in Italy, 16.8 kg in 2019, is observing a decreasing trend (ISMEA; 2021). In the Po Valley in particular, the open cycle rearing system is widespread, which means that the farms manage only a part of the rearing cycle. Weaned calves are bought externally from pasture-based systems, mainly in France, and Italian farms directly manage only the fattening part, where animals are fed a mixed diet of self-produced fodder and commercial feed and supplements purchased externally. The incidence of this rearing system on national beef production is around 44-48% (ISMEA, 2021), and has been extensively described in literature, for example by Berton et al. (2017) and Bragaglio et al. (2018).

Pig farming in Italy is also an important sector, where around 9 million pigs are produced for slaughter yearly. In Italy in 2019 the pig production system generated a revenue of over 3,000 million €, equal to the 5.7 % of the national agricultural output (Mipaaf, 2020). The Italian pig meat supply chain is complex and articulated. In the livestock segment there are over 32,000 farms, which are unevenly distributed in the country, with a strong concentration in the Po Valley. These pig farms are mostly small organizations (with revenues inferiors to 10 million €) acting as suppliers of companies downstream of the supply chain. These are gradually integrating pig farms through the contractual arrangement of the "Soccida"[1], whose diffusion has steadily increased in the last 10 years (Mipaaf, 2020).

All these supply chains also include many actors in the upstream stages related to the production of the raw materials consumed (e.g., feed and energy) and, increasingly, in all the processes related to manure management, which may be carried out directly by the farmer/cooperative or even externally. This issue is becoming more and more important due to the environmental impacts that it entails, as will be explored in the following

[1] Contract aimed at setting up an agricultural enterprise with an associative nature, in which an economic collaboration is implemented between the one who has the livestock (rescuer, *soccidante)* and who must raise it (breeder, *soccidiario)*

subchapters and in the whole thesis. As an example, Figure 1.1 provides an overview of the main sub-processes, and consequently stakeholders, commonly involved in the pig production chain.

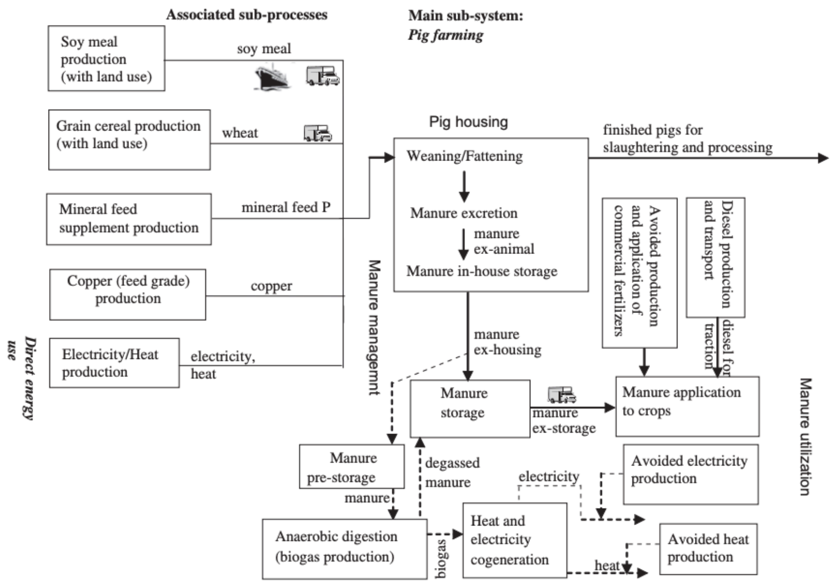

Figure 1.1 - Overview of the pig product chain, including upstream processes, up to the farm gate. Source: Nguyen et al. (2012)

Farmers are therefore actors in an intricate supply chain and, while they suffer from the variations in the price of meat and milk which undergoes continuous fluctuations also based on factors that do not depend on them, and the constant increase in the cost of production factors, they are increasingly called, for via market trends and European and/or national regulations, to transform production processes by increasing environmental and social sustainability.

1.2. Livestock production and the environment

Nowadays, the relationship between animal production and the environment is increasingly recognized as a critical issue. As the global demand for animal products continues to grow, understanding the multiple interactions between animal production and the environment is essential for sustainable resource management. In fact, livestock production has a number of positive effects on the environment, but at the same time there are several major negative impacts, such as climate change, land use and land use change, release of pollutants in air, soil and water bodies, antibiotic resistance, and in some cases, competition for natural and artificial resources between feed and food. As awareness of these environmental issues grows, the need for collaboration among stakeholders to develop and implement mitigation strategies becomes more pressing. The practices studied and adopted to mitigate these impacts are manifold, including agroecological approaches, precision farming and precision livestock farming, alternative feeding strategies, and circular economy models that aim to minimize waste and promote nutrients recycling.

This work focuses mainly on two aspects that link livestock production and the environment: greenhouse gas emissions and the release of pollutants into the air, with a focus on ammonia. These are two issues that are particularly important in the Italian production context and, more generally, in Europe. As for Italy, the Po Valley is the macro-area with the highest livestock density, due to the conditions that make it particularly suitable for agriculture and livestock breeding, which causes significant environmental pressures.

Table 1.1 provides an overview of total Italian national GHG emissions as reported by ISPRA (Higher Institute for Environmental Protection, in Italian) in the national GHG inventory prepared according to the IPCC guidelines. Looking at this data, the first thing that stands out is the decline in emissions over time. Indeed, total GHG emissions from the agricultural sector have decreased significantly (-10.9%) over the last twenty years. At the same time, however, its relative contribution to total national emissions has increased, from 6.4% in 2000 to 8.3% in 2020. A significant decrease (more than 30%) in emissions from the energy sector over the same period, together with the increase in carbon sequestration by LULUCF, has led to an increase in the relative contribution of the agricultural sector. Although the energy sector is currently still by far the largest source of CO_2 eq in Italy, its reduction trend is expected to continue in the future, which would bring more and more attention to agriculture.

Table 1.1 - Italian national greenhouse gas emissions, expressed in Gg CO_2 eq, and trend from 2000 to 2020 for the agricultural sector, divided into subcategories according to the IPCC calculation and reporting structure. Source: Eurostat, 2023.

Gg CO_2 eq /year	2000	2005	2010	2015	2020
Enteric fermentation	15.13	13.29	12.88	13.04	13.53
Manure management	7.19	7.10	6.87	6.35	6.22
Rice cultivation	1.66	1.75	1.82	1.67	1.58
Managed agricultural soils	12.12	11.47	9.57	9.67	10.82
Field burning of agricultural residues	0.02	0.02	0.02	0.02	0.02
Liming	0.00	0.01	0.02	0.01	0.01
Urea application	0.53	0.51	0.34	0.42	0.47
Total	36.68	34.19	31.56	31.21	32.68

Looking then at the contributions within the agricultural sector itself, livestock turns out to be very influential on total emissions, considering that “enteric fermentation”, “manure management” and part of “managed agricultural soils” (which is mainly represented by direct and indirect N_2O emissions from fertilizers field application) are attributable to this sector.

It should also be noted that, according to the IPCC reporting approach, only direct emissions geographically located in Italy are accounted for here. However, in a life cycle vision, animal production involves a series of others indirect greenhouse gases emissions, including those linked to energy and feed consumption. Furthermore, in this way, any breeding phases that are not held within the country are also excluded. In the case of beef cattle, for example, part of the Italian production derives from fattening of weaned calves imported from other countries (e.g., from France).

In conclusion, the overall emissions linked to the livestock activity in Italy, including also the indirect ones and the production phases of some supply chains that take place outside the country, would be even higher. To the authors' knowledge, there is currently no estimate of this overall value. Zucali et al. (2017) performed a similar study but focused only on Lombardy, one of the leading regions for the livestock sector as already mentioned, and based on life cycle approaches, they estimated the total GWP of milk and meat production in Lombardy to be 14.6 Mt CO_2 eq.

The adoption of GHG reporting frameworks that use a life-cycle perspective allows for a clearer and more complete view of actual GHG emissions, along with other environmental impacts, from Italian agricultural supply chains. This topic will be further explored in the next sub-chapter.

The agricultural sector is also the main contributor to ammonia (NH_3) emissions in the EU, accounting for more than 90% of them, despite the reduction achieved in absolute terms since 1990 (EEA, 2019). In particular, the livestock sector contributes to about 80% of the agricultural share due to NH_3 emissions from effluents that occur during permanence in housing facilities, storage and field application (Figure 1.2 provides the detailed breakdown of NH_3 emission sources in Italy for 2020). The challenge in managing livestock effluent is that while it may be possible to retain ammoniacal nitrogen at one point, such as during its time in housing facilities, it remains susceptible to volatilization at subsequent stages, such as during handling, storage, or field application (Reis et al., 2015). NH_3 causes a series of cascading negative effects that damage both ecosystem biodiversity due to acidification and nitrogen enrichment, and human health, being a precursor of secondary fine particulate matter (PM2.5). The damages to public health and ecosystems have been evaluated in 10–25 € per kg of emitted NH_3 (Executive Body of the Convention on Long-Range Transboundary Air Pollution, 2019). For all these reasons, in recent decades, European international integrated policies have been increasingly interested in quantifying, monitoring, and limiting NH_3 emissions from livestock.

Crucially from an environmental perspective, preventing the loss of ammonia (NH_3) can help retain valuable nutrients in the manure or slurry, allowing a richer nutrient mix to be applied to the soil when these organic materials are spread. This in turn reduces the need for synthetic mineral fertilizers, which not only benefits environmental sustainability, but also maximizes the use of an existing free resource, adding value to the process. This holistic approach contributes to both environmental protection and resource optimization.

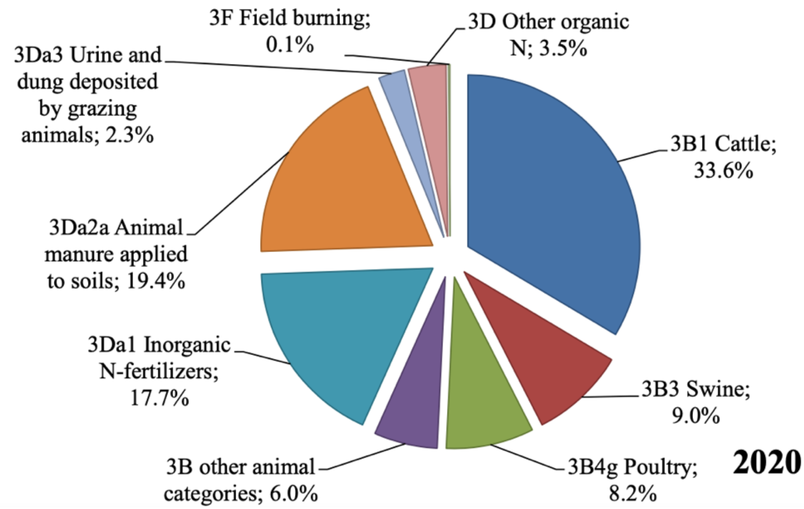

Figure 1.2 – Relative contribution of the various emission sources to the total NH_3 emitted by agriculture in Italy in 2020. Source: ISPRA (2022)

Agriculture also contributes to PM pollution through direct emissions from livestock and mechanization, and indirectly through NH_3. Indeed, the latter can react with sulfur dioxide (SO_2) and nitrogen oxides (NOx) in the atmosphere, leading to the formation of secondary sulfate and nitrate particles, the main components of particulate matter (PM2.5) (Lovarelli et al., 2020).

As mentioned above, the efforts of the European Commission (EC) under the Convention on Long-range Transboundary Air Pollution and its extension protocols have already led to significant improvements in NH_3 emissions, with a 24% reduction from 1990 to 2018 (EEA, 2019). For the livestock sector, the reduction is mainly due to a decrease in livestock numbers (especially cattle), changes in manure management, and improved feeding techniques (Jacobsen et al., 2019). However, the downward trend in NH_3 emissions has slowed down in recent years (EEA, 2019). Moreover, international policies adopted in recent decades to reduce anthropogenic emissions of SO_2 and NOx, which are also involved in PM2.5 formation, have led to greater reductions in relative terms than those of NH_3 (Reis et al., 2015), which favors a greater focus on the latter.

1.3. The Life Cycle Assessment approach for analyzing the environmental impact of agricultural and livestock supply chains

Life cycle assessment (LCA) is a holistic approach for evaluating the environmental impact during the life cycle of products or processes. LCA has long been used for environmental metrics of food products (Andersson et al., 1994), as well as a decision-making tool for environmental management in the same area. It is internationally standardized by ISO 14040:2006 and 14044:2018, which define the four founding phases (goal and scope definition, inventory, impact assessment and interpretation of results) and guide an application as harmonized as possible among practitioners. LCA also finds application in Type III certification programs (regulated by ISO 14025:2010) to produce environmental product declarations (EPD), which are increasingly being used by enterprises in the agri-food sector for reasons of transparency, marketing and eco-labeling (Cimini & Moresi, 2018). The different certification programs provide sector-specific guidelines, called product category rules (PCR), for the compilation of EPDs (Minkov et al., 2015).

FAO, through the Livestock Environmental Assessment and Performance Partnership (LEAP), has recently published a series of guidance documents on how to, by means of the LCA method, measure, avoid and mitigate environmental impacts associated with livestock chains, including large and small ruminants, pigs and poultry, as well as some focus documents on feed and feed additive, nutrient flows, biodiversity and water use.

The advantage of LCA over other environmental assessment methods is that it allows the simultaneous assessment of different environmental impacts of the same product or process, in order to consider any trade-offs and avoid shifting the burden between different impact categories when implementing mitigation measures. In any case, while acknowledging that agri-food production involves many other environmental aspects, a special focus in this thesis is made on global warming, as it is an environmental issue of common and urgent relevance, involving actors at all economic, social and political levels, regardless of location, which translates into a necessary intervention for its mitigation.

A first overview of the application of LCA to livestock systems was made by De Vries & De Boer in 2010, and since then the adoption of this method has further increased. Indeed, in more recent years LCA reviews have been published in specific fields of livestock, including milk (Baldini et al., 2017), pig (McAuliffe et al., 2016) and beef (de Vries et al., 2015) production systems. These studies recapitulated the environmental criticalities of each, but also highlighted some limits of the use of the LCA. As a first literature screening showed that a specific review on poultry was missing, one of the first activities of this work was to deepen the LCA studies published on poultry production, which resulted in a publication in Trends in Food Science and Technology (Costantini et al., 2021). This was motivated by the fact that poultry productions have experienced impressive growth in the last decades and poultry meat (represented mostly by chicken meat) is the most produced worldwide (FAOSTAT, 2023). According to OECD-FAO (2019), chicken meat is expected to increase by 40 Mt by 2028, representing about half of the total increase in meat production within that year. Beside this, poultry products play a major role in human nutrition, especially in developing countries, due to several factors including being relatively inexpensive, widely available, unaffected by religious restrictions and with a high nutritional value (FAO, 2013). In addition to reviewing the methodological approach to this sector through LCA, the study by Costantini et al. (2021) aimed to summarize the main findings and highlight the current shortcomings of the literature. The discussion focused on the production parameters that most influence environmental performance, as well as on possible mitigation measures, some of which are well known, while others are still partially unexplored. However, this work is not presented in detail in this thesis due to a matter of scope and space, for further information please refer to the publication.

It should always be kept in mind that making uncritical comparisons between the results of different LCA studies is unadvisable. In fact, in addition to the variability and uncertainty related to activity data themselves, some methodological choices can have significant influence. For this reason, particular attention when interpreting the results must be paid to the methodological choices adopted as well as to the assumptions made. For this reason, all the above reviews also draw attention to the different methodological approaches used in the literature with respect to functional units, system boundaries, inventory data collection, and management of multifunctionality.

For all livestock sectors, the vast majority of literature points to feed consumption, with all its indirectly related upstream impacts, as the main environmental hotspot across different environmental impact categories (energy, water and land use, acidification and eutrophication, greenhouse gas emissions). This trend does not change regardless of the different rearing systems (e.g., conventional vs organic, intensive vs extensive). In fact, animal performance related to feed, in particular feed intake, daily weight gain and/or average daily milk and/or egg production per hen day, and consequently the feed conversion ratio (FCR), have been identified as highly influencing factors on the environmental impact of the final products of the supply chains. Therefore, the search for continuous improvement of all these parameters on the farm is always mentioned as the first aspect to be pursued in order to achieve a more sustainable production. For meat products, the environmental results are strongly influenced by the age at slaughter. In addition to animal-related parameters, the quality of buildings and equipment and other farm characteristics have been shown to influence the variability of environmental performances.

In the case of meat, the slaughtering, processing and packaging stages were found to have a limited impact on most of the impact categories, except for cumulative energy demand, for which they have a weight in the order of 10-15%. Seeking continuous improvement also in these areas (e.g. through greater efficiency in energy consumption, proper management of waste and wastewater, adoption of recyclable packaging) can play a role in making the supply chain more sustainable, but it is clear that the priority mitigation must be sought in the agricultural phase, as it would be the most incisive.

Mitigation scenarios for existing production systems have been widely evaluated in the literature. Given the crucial role that it plays in the livestock overall impact, many of these have addressed the feed sector in various aspects. However, in addition to interventions in the search for more sustainable feeds, there are many pathways for mitigation, some already consolidated, others more experimental, tested and proposed in the literature. In fact, it is not possible to address the issues of sustainability of agri-food systems without a systemic vision of each proposal. These include, for example, increasing production efficiency, substituting inputs with lower-impact alternatives, and developing agroecological practices based on the mobilization of biological processes and circularity, which often require redesigning systems. The use of new technologies (biotechnologies such as genomics, epigenetics, microbiota; digital technologies; innovative biorefineries) and new governance to ensure business continuity so that employment is not threatened will contribute to the transition. The transition to more sustainable products and processes must be promoted by public policies and be rewarded, visible and economically valued. To achieve the goal, most of these approaches will need to be mobilized simultaneously.

Table 1.2 summarizes, in clusters, a wide range of management strategies that can be pursued for sustainable development of the livestock sector. It is important to keep in mind that while the main objective of these strategies is currently climate change mitigation, many of them also have benefits or at least no strong negative impacts on some other objectives (e.g. biodiversity, water quality, etc.).

Table 1.2 - Summary of mitigation measures proposed in the literature for the impacts of livestock production, grouped by macro-area and intervention method. Source: Peyraud & MacLeod (2020).

	Efficiency of herds	Substitution	Agro-Ecology	Circular economy
Increasing resources use efficiency	• Animal efficiency • Feeding strategies, feed additives • Herd management: less mortality, lifespan of reproductive females	• Reducing mineral N application • Take full account of manure N and symbiotic N from legumes	• Diversification of crop rotation for new plant protein sources • More robust animals	• New protein sources derived from wastes (biomass refinement, use of insects, etc.)
Reducing emissions	• Breeding for lower methanogenesis • Breeding for increased efficiency • Feeding strategies including use of methanogen inhibitor • Herd management: less mortality, lifespan of reproductive females	• Use of legumes and manure instead of mineral N fertilizer • Replace feeds associated with land use change with alternatives • Replacing fossil fuel with renewable energy	• Diversification of crop rotation • Manure storage and application • Grassland and hedges development • Soil C sequestration practices • Soil restauration practices • Agroforestry	• Manures transfer between farms and regions • Anaerobic digestion of manure
Closing nutrient cycles	• Breeding for increased efficiency • Precision feeding • Use of feed additive	• Use of legumes and manure instead of mineral N and P fertilizer	• Diversification of crop rotation • Grassland and hedges • Manure storage and application • Soil C sequestration practices • Mix farming systems	• Manure refinement • Reintroduction of livestock in cropping regions • Use of coproduct, waste streams and new protein sources as feed • New value of animal by products
Increasing biodiversity		• Use of legumes	• Crop diversification • Grassland and hedges development • Development of Agroforestry • Use of local breeds	• Integration of crop and livestock at territorial level • Use the ability of livestock to utilize a diverse range of biomasses
Controlling infectious diseases			• More robust animals • Integrated and preventive management of microbial ecosystems	
Improving animal welfare & health	• Prevention of production diseases such as lameness and mastitis	• Vaccines and plant secondary compounds instead of antimicrobials	• Animal robustness and adaptability • Suppression of painful practices • Improvement of living environment for expression of natural behavior • Integrated preventive management of animal health	
Ensuring food safety with less inputs		• Vaccines and plant secondary compounds instead of antimicrobials • Use of legumes	• Improved animal robustness • Integrated and preventive management of animal health	

In order to better contextualize the production sector under study and also to have a reference base for possible comparisons of results, a brief review of the published literature on the application of LCA to Italian animal production systems was carried out. The results are presented graphically in Figure 1.3 and in Table 1.3.

To perform the review, scientific manuscripts were retrieved by Scopus database covering the period 2015 to 2022. This period was selected to reflect the current state of the art and recent development, as well as the application of updated LCA methods. As far as cow's milk is concerned, only the papers published in the period 2018 to 2022 were analyzed since, being by far the most analyzed supply chain in the literature, the resulting papers in recent years were already considered complete and significant to represent that sector.

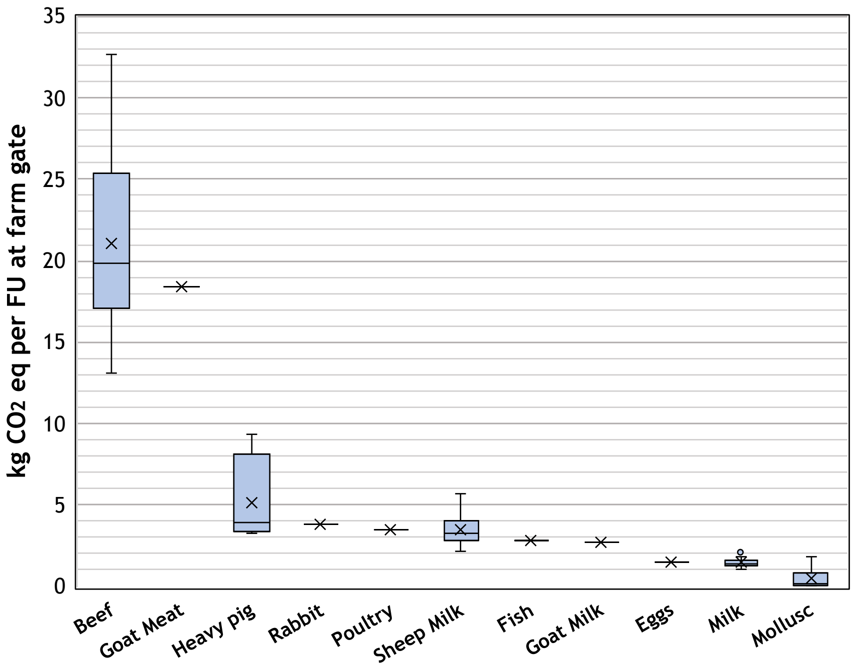

Figure 1.3 - Literature review of GWP impact of animal productions in Italy at farm gate. Only studies that collected and elaborated inventories that were at least partially composed of primary data from one or more farms were considered. For individual FUs, refer to Table 1.3. Source: author's elaboration on literature review results.

Table 1.3 - Literature review of GWP impact of animal productions in Italy at farm gate. Only studies that collected and elaborated inventories that were at least partially composed of primary data from one or more farms were considered. Source: author's elaboration on literature review results. Notes: LW – Live Weight; FPCM – Fat and Protein Corrected Milk; FW – Fresh Weight.

Product	FU	n° obs	Median	Average
Beef cattle	1 kg LW	11	19.80	21.07
Pig	1 kg LW	4	3.90	5.11
Rabbit	1 kg LW	1	3.86	3.86
Poultry	1 kg LW	1	3.50	3.50
Sheep Milk	1 kg FPCM	6	3.31	3.50
Fish (trout) - aquaculture	1 kg FW	1	2.83	2.83
Goat Milk	1 kg FPCM	1	2.67	2.67
Egg	1 kg FW	1	1.54	1.54
Milk	1 kg FPCM	19	1.38	1.42
Mollusc	1 kg FW	6	0.13	0.46

Variability has been found in the coverage of impacts. The most commonly studied category is GWP. Some studies may present multiple impact assessment results due to the consideration of different rearing systems or production scenarios for comparison.

In all studies, feed production and supply was found to be a major contributor to the impact of poultry production for both global warming, acidification and eutrophication. Although a wide variety of feeds and feed components are used, protein feeds are often the most impactful in the different LCA studies reviewed, especially for GWP (mainly due to Land Use Change). Soybean and its derived products are the feeds more frequently identified as environmental hotspots, also because they are the main source of protein. Moreover, low self-sufficiency for this protein source exposes the continent to serious food security risks. For organic production in the EU, the GWP impact of soybean and its by-products is usually lower because genetically modified crops are not allowed and, consequently, locally produced feedstocks are used instead of imported ones.

Some specificities of the Italian agri-food supply chains emerge from the review, such as the fact that the impact of the pig is slightly higher than other European references due to the typically longer fattening cycle of the Italian heavy pig. Nonetheless, the trends of international productions and the related environmental performances are generally confirmed. Poultry products are recognized, together with milk, as the most environmentally efficient among the main livestock production chains, in particular with regard to the carbon footprint but also resources depletion (e.g., land and energy use).

Beef is confirmed as having the highest impacts compared to the rest of animal products, which is also accentuated in Italy by the widespread production and consumption of veal (which causes the peak that can be observed in the graph).

Aquaculture shows good results and will play an increasingly important role in the food chains of the near future, also in view of its constant growth. However, it falls outside the scope of this thesis and will therefore not be explored here.

1.4. References

Andersson, K., Ohlsson, T., & Olsson, P., 1994. Life cycle assessment (LCA) of food products and production systems. Trends in Food Science & Technology, 5(5), 134–138.

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.

Benato, A.; Macor, A. 2019. Italian Biogas Plants: Trend, Subsidies, Cost, Biogas Composition and Engine Emissions. *Energies* 12, no. 6: 979.

Berton M., Agabriel J., Gallo L., Lherm M., Ramanzin M., Sturaro E., 2017. Environmental footprint of the integrated France-Italy beef production system assessed through a multi-indicator approach. Agricultural Systems, 155, 33-42.

Bragaglio A., Napolitano F., Pacelli C., Pirlo G., Sabia E., Serrapica F., Serrapica M., Braghieri A., 2018. Environmental impacts of Italian beef production: a comparison between different systems. Journal of Cleaner Production, 172, 4033-4043.

Cimini, A., & Moresi, M., 2018. Are the present standard methods effectively useful to mitigate the environmental impact of the 99% EU food and drink enterprises? Trends in Food Science & Technology, 77, 42–53.

Clune S., Crossin E., Verghese K., 2017. Systematic review of greenhouse gas emissions for different fresh food categories. J. Clean. Prod., 140 (2), 766-783.

Costantini M., Ferrante V., Guarino M., Bacenetti J, 2021. Environmental sustainability assessment of poultry productions through life cycle approaches: a critical review. Trends in Food Science & Technology, 110, 201-212.

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

De Vries M., Van Middelaar C.E., De Boer I.J.M., 2015. Comparing environmental impacts of beef production systems: A review of life cycle assessments. Livestock Science; 178, 279-288.

Eea, European Environmental Agency, 2019. European union emission inventory report 1990-2017, under the UNECE convention on long-range transboundary air pollution (CLRTAP). EEA Report | No 08/2019

Eurostat, 2021. European Commission online statistical database (available online at: https://ec.europa.eu/eurostat/data/database, accessed on February 2021).

Executive Body of the Convention on Long-Range Transboundary Air Pollution, 2019. Draft of assessment report on ammonia. Available at: https://unece.org/fileadmin/DAM/env/documents/2019/AIR/EMEP_WGE_Joint_Session/Assessment_Report_on_Ammonia_20190827.pdf, Accessed September 2023.

FAO (Food and Agriculture Organization of the United Nations). Poultry development review. Italy, Rome (2013)

FAOSTAT. FAO online statistics database. http://faostat.fao.org

GSE, 2021. GSE (Gestore dei Servizi Energetici S.p.A.) online statistical database (available online: https://www.gse.it/dati-e-scenari/statistiche, accessed on February 2021).

ISMEA (Institute of Services for the Agricultural and Food Market, *in Italian*), 2021. Allevamento bovino da carne – Scheda di settore (*Beef cattle – Sector report*) 2020. Presentation available at https://www.ismeamercati.it/flex/files/1/a/2/D.2e8af1c92fc3e2b3b1fa/Skeda_bovino2021.pdf.

ISMEA ((Institute of Services for the Agricultural and Food Market, *in Italian*), 2022. ISMEA Mercati – Lattiero Caseari, Ottobre 2022. Presentation available at https://www.ismeamercati.it/flex/files/1/5/0/D.e3e7b0123873e4cf895a/Scheda_LATTE_2022.pdf

ISPRA (Higher Institute for Environmental Protection, *in Italian*), 2022. Italian Emission Inventory 1990-2020. Informative Inventory Report 2022. Report n. 361/2022. Text available on ISPRA website at http://www.isprambiente.gov.it

Jacobsen B.H., Latacz-Lohmann U., Luesink H., Michels R., Ståhl L., 2019. Cost of regulating ammonia emissions from livestock farms near Natura 2000 area – analysis of case farms from Germany, Netherlands and Denmark. J. Environ. Manag., 246, pp. 897-908, 10.1016/j.jenvman.2019.05.106

Lovarelli D., Conti C., Finzi A., Bacenetti J., Guarino M. 2020. Describing the trend of ammonia, particulate matter and nitrogen oxides: the role of livestock activities in Northern Italy during Covid-19 quarantine. Environmental Research, 110048.

McAuliffe G., Chapman D.V., Sage C.L., 2016. A thematic review of life cycle assessment (LCA) applied to pig production. Environmental Impact Assessment Review, 56, 12-22.

Minkov, N., Schneider, L., Lehmann, A., & Finkbeiner, M., 2015. Type III environmental declaration programmes and harmonization of product category rules: Status quo and practical challenges. Journal of Cleaner Production, 94, 235–246.

Mipaaf (Ministero delle politiche Agricole, alimentari e forestali), 2020. La competitività del settore suinicolo – il Quadro di settore, i trend emergenti e gli strumenti a support del rilancio della filiera nazionale.

Nguyen T.L.T., Hermansen J.E., Mogensen L., 2012. Environmental costs of meat production: the case of typical EU pork production. Journal of Cleaner Production, 28, 168-176.

Peyraud J.-L., MacLeod M., 2020. Study on Future of EU livestock: how to contribute to a sustainable livestock sector? – Final Report. Luxembourg: Publications Office of the European Union, 2020. PDF ISBN: 978-92-76-20624-8, doi: 10.2762/3440, KF-02-20-525-EN-N.

Reis, Sutton, Howard, 2015. Overview, aims and scope. S. Reis, et al. (Eds.), Cost of Ammonia Abatement and the Climate Co-benefits, Springer Science & Business Media Dordrecht (2015), pp. 1-6, 10.1007/978-94-017-9722-1_1.

Roma R., Corrado S., De Boni A., Bonaventura Forleo M., Fantin V., Moretti M., Palmieri N., Vitali A., De Camillis C., Life Cycle Assessment in the livestock and derived edible products sector, In: Notarnicola B. et al. (eds.), Life Cycle Assessment in the Agri- food Sector. Springer International Publishing Switzerland, 2015. 251-332.

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 - Aim and organization of the thesis

2.1. Aim of the work

As mentioned in the introduction, there is a growing interest among policymakers, sector workers, and consumers in identifying measures to mitigate the environmental impacts of livestock systems. In northern Italy, currently, particular interest is directed to the reduction above all in terms of greenhouse gas and atmospheric pollutant emissions. There are already many technologies and management practices that could contribute to significant emissions reductions, but not all of them are being widely adopted. In some cases, this is due to problems of low diffusion and awareness among farmers, in other cases it is due to limitations imposed by cost implications.

This work aimed at understanding the environmental performance of different mitigation strategies to reduce on-farm emissions of greenhouse gases and air pollutants. More specifically, for the swine sector, the implementation of air treatment technologies within the pig housing facilities were analyzed, while for the cattle sector, measures focused on the sustainable management of livestock waste, including the application of additives in livestock slurry storage and anaerobic digestion.

In order to obtain a more holistic understanding of the issues under study, the analytical approach was carried out at several levels: (i) through direct measurements carried out during field campaigns on farms, in order to measure the actual level of emissions from typical livestock farms emission sources; (ii) the application of life cycle assessment (LCA) methodology, starting from primary production data collected directly through questionnaires and/or measured in the field, combined with secondary data extracted from the literature, and/or derived from estimation models, to obtain a broader assessment of the environmental impacts associated with animal production, as well as an overview of the trade-offs associated with possible mitigation scenarios; (iii) large-scale modeling based on regional/national statistics and generic activity factors (e. e.g. farm characteristics, emission factors) to estimate the environmental impacts that could result from the widespread implementation of some mitigation practices.

2.2. Overview of the chapters

The body of the thesis has been divided into two main chapters related to the two main research lines followed during the doctoral thesis:

i) the impact of livestock on air pollution, discussed in Chapter 3;

ii) the impact of livestock on greenhouse gas emissions, discussed in Chapter 4.

It is important to emphasize that these are very broad and complex topics and that a Ph.D. thesis would not be sufficient to explore them in their entirety. The work falls within these specific fields of study and has been developed primarily considering the geographical scope of reference, i.e., the intensive animal production context of Northern Italy. Nevertheless, for several reasons, the work has focused on assessing some mitigation activities, one of which is certainly the involvement in research projects, such as in the case of air treatment technologies tested in field trials in the context of the Life-MEGA project coordinated by Prof. Guarino; other publications have instead been driven by an interest in deepening some topics that are still little treated in the literature.

Chapters 3 and 4 consist of a specific introduction that contextualizes the topic, followed by a collection of publications, selected from those produced (published or under review) by the candidate during the Ph.D. activity.

Chapter 3 is composed, in order, of the following three papers:

1) *Comparison of ammonia air concentration before and during the spread of COVID-19 in Lombardy (Italy) using ground-based and satellite data*. This is the only paper that does not deal with mitigation measures but focuses on evaluating the atmospheric concentrations of ammonia (NH_3) in Northern Italy during the COVID-19 pandemic. The study analyzes both ground-based and satellite measurement data for the year 2020 and compares it with data from previous years. In contrast to some other air pollutants, the study concludes that the anti-COVID-19 measures affecting human activities did not lead to a reduction in NH_3 air concentrations. The work confirms the strong relationship between agriculture, whose activities have not been significantly interrupted during the pandemic, and emissions of this pollutant. At the same time, it provides insight

into the importance of combining data from multiple sources to better understand atmospheric phenomena and inform pollution mitigation efforts.

2) *Life cycle assessment of air treatment techniques to reduce NH_3, GHG, VOCs and particulate matter emissions from pig housing*. The paper mainly focuses on quantifying the environmental impact of pig production using the Life Cycle Assessment (LCA) approach to compare conventional production conditions with two alternative scenarios implementing air treatment technologies: i) wet scrubber and ii) dry scrubber. The study included one transition farm in Spain and two fattening farms in Italy. The main results indicate that both tested scrubber technologies showed the potential to reduce emissions during the pig housing phase, affecting categories related to air pollutant emissions such as particulate matter formation, acidification and eutrophication. However, there were trade-offs between emission reductions and increased resource consumption associated with the scrubber technologies. The dry scrubber was found to be the more favorable option when considering the balance between emission reduction and trade-offs. The study observed similar environmental trends in Spain and the two Italian farms, although with some differences in absolute values. Scrubbers had a more pronounced impact, both positive in emission reduction and negative in trade-offs, in the Italian farms. A sensitivity analysis on characterization factors also highlighted regional differences in scrubber impacts, especially on acidification. In conclusion, scrubber technologies are promising for mitigating the environmental impacts of pig farming, especially in regions where eutrophication and particulate matter formation are of concern. However, these technologies alone cannot fully address the environmental challenges of pig production, which require interventions at different levels within the supply chain. In addition, there is room for improvement in scrubber efficiency to enhance emission reductions while optimizing resource consumption, particularly for wet scrubbers, to minimize trade-offs.

3) *Improvement of human health and environmental costs in the European Union by air scrubbers in intensive pig farming*. The paper addresses the environmental and health impacts of intensive pig production in the European Union (EU) due to air pollutant emissions from housing facilities, namely NH_3, PM10, NMVOC

and N_2O. It examines the potential impact of the large-scale implementation of air pollution control technologies, in particular wet acid scrubbers, in pig farms across the EU. Alternative scenarios with different levels of wet acid scrubber implementation are considered to assess their impact on emission reductions, human health and environmental costs. Implementation rates of 35% and 65% of scrubbers in relevant pig farms across the EU result in significant reductions in human health impacts and environmental costs associated with air pollution. In particular, the reduction in ammonia emissions effectively achieved by wet acid scrubbers plays a critical role in mitigating these impacts. While the study acknowledges the need for further assessment of issues such as cost-effectiveness, animal welfare, production performance, and more, the findings highlight the significant potential for improving the environmental sustainability of intensive pig production at the housing stage. The paper advocates increased encouragement and support from EU and/or national policies to promote the implementation of wet acid scrubber technology, especially in regions where its use is not yet common.

Chapter 4 is composed, in order, of the following two papers:

4) *Real-scale study on methane and carbon dioxide emission reduction from dairy liquid manure with the commercial additive SOP LAGOON*. The paper addresses the urgent need to reduce methane (CH_4) emissions as a significant contributor to climate change, with a focus on managing emissions from liquid manure in the dairy industry. The study evaluates the effectiveness of the commercial additive SOP LAGOON in reducing carbon-based greenhouse gas (GHG) emissions from manure storage. A full-scale trial was conducted at a commercial dairy farm over a period of approximately four months, comparing emissions from a manure tank treated with SOP LAGOON to an untreated control tank. Even if the effectiveness of the emission abatement during the tests fluctuated and the chemical-physical-biological principles through which the additive acts are not yet completely clear, the results of the tests are to be considered positive, as after three months from the initial additive applications, the treated storage tank demonstrated

substantially lower and statistically significant emissions compared to the untreated tank.

5) *The effects of incorporating renewable energy on the environmental footprint of beef production*. This paper addresses the environmental impacts of beef cattle production and evaluates how on-farm adoption of renewable energy systems, namely anaerobic digestion (AD) for biogas production and rooftop photovoltaic (PV) systems, affects the overall environmental and energy footprint of beef production. The study uses a life cycle assessment approach, comparing different baseline production scenarios with different energy mitigation systems. The implementation of anaerobic digestion systems is shown to considerably enhance the environmental and energy performance of beef production. The adoption of PV systems contributes to further improvements, albeit to a lesser extent. The best outcome was observed when both AD and PV systems are combined. The study highlights that displacing fossil fuel-based energy plays a critical role in reducing the overall environmental impact of beef production, and the integration of renewable energy systems such as anaerobic digestion and photovoltaics is a major step in this direction. While this study provides valuable insights, future research should explore additional technical and production parameters in conjunction with different farming systems to further enhance the understanding of this issue.

Chapter 5, on the other hand, presents a final discussion of the work, drawing general conclusions and setting the pace for further study and improvement.

CHAPTER 3 – Livestock and air pollution

Brief overview of the EU regulatory framework on air pollutant emissions and its relation to the livestock sector

The first official international cooperation agreement aimed at tackling the problem of air pollution dates to 1979, the year in which the Convention on Long-range Transboundary Air Pollution (CLRTAP) was signed by 31 Parties among which the European Community. The CLRTAP, which has now 51 Parties (UN, 2020), entered into force in 1983 and since then the EC and MS have pledged to respect constraints and achieve targets for the abatement of air pollutant emissions. In order to identify specific measures to be taken by Parties to cut their emissions of air pollutants, the CLRTAP has been extended over the years by eight protocols, among which the Protocol to Abate Acidification, Eutrophication, and Ground-level Ozone, also known as the Multi-effect Protocol or the Gothenburg Protocol, signed by 31 Parties in 1999 (UN, 2020). This protocol initially set ceilings on sulfur, NO_x, VOC and ammonia for national emissions of the Parties to be reached by 2010.

At the same time, the EU Directive 96/61/EC concerning Integrated Pollution Prevention and Control, commonly referred to as IPPC Directive, established common environmental constraints for operators in some economic sectors related to large-scale emissions of air pollutants. In order to obtain operating permits, the concerned operators must present reports on the environmental performance of their activities at specific time intervals, demonstrating to respect the constrains imposed and efforts to improve in sustainability over time by the adoption the so-called best available techniques (BAT). In this regard, the European IPPC Bureau (EIPPCB) was set up in 1997 with the aim of developing and validating BAT Reference Documents (BREF) for each sector of interest. In each BREF the state of art of techniques (both consolidated and emerging) applied to prevent and/or reduce emissions is defined and the best available ones among them are determined. Highlights and conclusions deriving from each BREF are finally summarized in documents called BAT Conclusions, which should be the reference for setting the operating permit conditions.

For MS, the common purpose of the Gothenburg protocol and the IPPC Directive has been combined in the EU Directive 2001/81/EC, which ratified the Gothenburg Protocol and set the National Emission Ceilings (NEC). This Directive requires BAT to be implemented in order to reduce national emissions and MS must encourage for their adoption by stakeholders at a country level, in accordance with the IPPC Directive, BREFs and national regulations.

Currently, the IPPC directive, already renewed by the Directive 2008/1/EC, has been integrated with 6 other European directives into the Directive 2010/75/EU on Industrial Emissions (IED). This Directive also gives the currently into force definition of *best available techniques*, according to which:

- ‘techniques’ refers to the technology used to prevent and/or reduce emissions and the way in which the installation is designed, built, maintained, operated and decommissioned;
- ‘available techniques’ means those developed on a scale which allows implementation in the relevant industrial sector, under economically and technically viable conditions, taking into consideration the costs and advantages, whether or not the techniques are used or produced inside the MS in question, as long as they are reasonably accessible to the operator;
- ‘best’ means most effective in achieving a high general level of protection of the environment as a whole.

In the IED it is also specified that the Commission should aim to update BREFs not later than 8 years after the publication of the previous version.

The emissions ceilings have been updated and renewed by a new NEC Directive (2016/2284/EU) entered on force in 2016. The new Directive transposes the reduction commitments for 2020 and for 2030 agreed by the EU and its MS under the 2012 revised Gothenburg Protocol (UN, 2020).

As concerns livestock production, a BREF was developed in 2003 for intensive rearing of poultry or pig (IRPP) and was subsequently reviewed and updated up to the currently adopted version of 2017 (Santonja et al., 2017). This document is intended for farms hosting more than 40000 places for poultry, or more than 2000 places for pigs over 30 kg, or more than 750 places for sows. This threshold was defined because it is estimated to represent the emission limit of about 10 tons of ammonia per year, a large amount that causes environmental pressure on the territory. In areas particularly suited to livestock farming, such as the Po Valley, where the concentration of livestock farms is very high, it is easy to imagine the environmental pressure to be even greater. The document describes specific measures that farmers need to take to get emissions down covering a wide range of on-farm processes and activities, from animal feeding to the adopted housing structures up to manure management. According to *Santonja et al.* (2017), farms that require a permit to operate because falling within the parameters of the IED are 20,018. As regards the pig sector in particular, farms with more than 2000 places for fattening pigs (live weight > 30 kg) are 6,580, while farms with more than 750 sow places are 1,863. However, these data refer to 2013 and there are no more recent updates in EU publications (data include UK in the calculation, while excluding Croatia). It is conceivable that the number of these farms has increased, given the trend observed in recent decades of a decrease in the number of livestock farms, while the number of animals has remained stable, which implies a progressive concentration of animals in ever larger farms (Eurostat, 2022).

Air scrubbers are air end-of-pipe cleaning devices used to control and remove pollutants from exhaust air, used extensively installed in pig and poultry housing facilities in North European countries like the Netherlands and Belgium (Van der Heyden et al., 2015). In the pig sector, the wet acid scrubber, which involves the capture of NH_3 and other pollutants (e.g., odorous compounds, dust) by means of an acid solution, is currently the most widely applied air cleaning technology. In the BAT conclusions of the IRPP BREF the air cleaning systems (including wet acid scrubber) are listed in the techniques to be considered BAT (Santonja et al., 2017). However, it is specified that this technique may not be generally applicable due to the high implementation cost.

The aim of this chapter is to examine the environmental concerns related to air pollution in Northern Italy, which are partly attributed to livestock activities. At the same time, it explores the untapped potential for mitigating these issues through the adoption of air treatment technologies in local pig farms.

References

Eurostat, 2022. European Commission online statistical database (available online at: https://ec.europa.eu/eurostat/data/database. accessed on December 2022).

Santonja, G.G., Georgitzikis, K., Scalet, B.M., Montobbio, P., Roudier, S., Sancho, L.D., 2017. Best available techniques (BAT) reference document for the intensive rearing of poultry or pigs. EUR 28674 EN. https://doi.org/10.2760/020485

Van der Heyden, C., Demeyer, P., Volke, E.I.P., 2015. Mitigating emissions from pig and poultry housing facilities through air scrubbers and biofilters: state-of-the- art and perspectives. Biosyst. Eng. 134, 74e93. https://doi.org/10.1016/ j.biosystemseng.2015.04.002

3.1. Comparison of ammonia air concentration before and during the spread of COVID-19 in Lombardy (Italy) using ground-based and satellite data

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Comparison of ammonia air concentration before and during the spread of COVID-19 in Lombardy (Italy) using ground-based and satellite data

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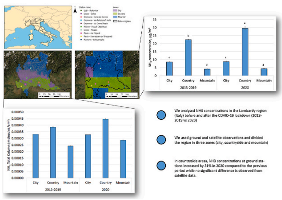
HIGHLIGHTS

- We used ground and satellite data to estimate NH3 concentrations in Lombardy.
- We analyzed the period before, during and after the COVID-19 lockdown.
- We divided both datasets in three zones: city, countryside and mountain.
- NH3 concentrations increased by 31% in countryside areas in 2020.
- No difference was found between pre- and post-lockdown NH3 from satellite data.

GRAPHICAL ABSTRACT



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ABSTRACT

Several anthropogenic activities have undergone major changes following the spread of the COVID-19 pandemic, which in turn has had consequences on the environment. The effect on air pollution has been studied in detail in the literature, although some pollutants, such as ammonia (NH_3), have received comparatively less attention to date. Focusing on the case of Lombardy in Northern Italy, this study aimed to evaluate changes in NH_3 atmospheric concentration on a temporal scale (the years from 2013 to 2019 compared to 2020) and on a spatial scale (countryside, city, and mountain areas). For this purpose, ground-based (from public air quality control units scattered throughout the region) and satellite observations (from IASI sensors on board MetOp-A and MetOp-B) were collected and analyzed. For ground-based measurements, a marked spatial variability is observed between the different areas while, as regards the comparison between periods, statistically significant differences were observed only for the countryside areas (+31% in 2020 compared to previous years). The satellite data show similar patterns but do not present statistically significant differences neither between different areas, nor between the two periods. In general, there have been no reduction effects of atmospheric NH_3 as a consequence of COVID-19. This calls into question the role of the agricultural sector, which is known to be the largest responsible for NH_3 emissions. Even if the direct comparison between the two datasets shows little correlation, their contextual consideration allows making more robust considerations regarding air pollutants.

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1. Introduction

Ammonia (NH_3) is an air pollutant of increasing environmental concern, whose emissions are primarily anthropogenic, released mainly from the agricultural sector by field application of synthetic fertilizers and manure management (Van Damme et al., 2015). NH_3 causes a series of cascading negative effects that damage both ecosystem biodiversity due to acidification and nitrogen enrichment (EMEP Centre on Emission Inventories and Projections, 2020; Erisman et al., 2007; European Commission, 2005) and human health (Van Damme et al., 2014), being a precursor of secondary fine particulate matter ($PM_{2.5}$) (Lovarelli et al., 2020; Perone, 2021). The damages to public health and ecosystems have been evaluated in 10–25 €/kg NH_3 (Executive Body of the Convention on Long-Range Transboundary Air Pollution, 2019). For these reasons, in recent decades, international integrated policies have been increasingly interested in quantifying, monitoring, and limiting NH_3 emissions. Despite the reduction obtained in absolute terms since 1990, in 2018, agriculture accounted for 93% of NH_3 emissions in relative terms in the European Union, still showing the criticality of this sector (EEA, 2019).

The reduction of NH_3 emissions is a complex process. First of all, measuring this compound is not easy, which also makes accurate monitoring difficult. The most widely used methodologies to date are measurements through ground-based instruments and satellite-based remote-sensing (Nair and Yu, 2020). However, each of these two techniques has advantages and problems. The ground-based measurements currently allow understanding the evolution of the atmospheric concentrations over time at ground level, but they are affected by local spatial and meteorological variability. Indeed, NH_3 volatilization from main primary sources (fertilizers and manure) is influenced by weather parameters such as wind speed, rainfall, and temperature, among others (Brentrup et al., 2000). Given the high variability related to NH_3 emissions, a geographically dense network of control units can be a useful method to fulfill monitoring requirements. However, due to the stickiness of NH_3 to observational instruments, the control units that collect NH_3 data are normally fewer per unit area than those used to measure other air pollutants (Van Damme et al., 2015). Satellite remote sensing, on the other hand, is based on the distinction of the NH_3 spectrum in the gas-phase by means of infrared spectrometers and allows to obtain a broad spatial coverage (both superficial and vertical) of NH_3 atmospheric concentrations, but the measurements available to date suffer from temporal discontinuity because deployed instrumentation is not onboard geostationary satellites (Nair and Yu, 2020). Furthermore, the reliability of the measurements at night or in the presence of clouds decreases. Ultimately, since the two methods compensate, at least partially, the respective limitations, considerations on atmospheric NH_3 pollution made by integrating both types of measurement can be more solid and comprehensive.

During the worldwide spread of Coronavirus Disease (2019) (COVID-19) in 2020, attention on air quality and the need to understand changes in the presence of pollutants in the atmosphere increased considerably. In literature, several studies have focused on the relationships among the COVID-19 outbreak, government actions to contain the spread of the infection and air pollution (Nuñez-Delgado et al., 2021; Zambrano-Monserrate et al., 2020). Among the European countries, Italy was the first in which the infection was detected in 2020, and which suffered a rapid spread of the infection in the first months of the year; this led the government to implement partial restrictions (e.g., establishment of a "red zone" in some municipalities in Northern Italy on 23 February, and subsequent interruption of school and university teaching in attendance). Then, heavy restrictions were introduced starting on 8 March, until on 23 March a nationwide lockdown was declared. Only industries deemed essential, such as food and pharmaceutical supply chains, and the agrifood sector were allowed to remain operational. This lasted officially until 3 May (DPCM, 2020).

As concerns the Po Valley in Northern Italy, where air pollution is recognized as being normally high (Raffaelli et al., 2020), the lockdown period has led to significant reductions in atmospheric concentrations of pollutants such as $PM_{2.5}$, PM_{10}, nitrogen oxides (NO_x) and others (e.g., carbon monoxide and benzene) (Buganza et al., 2020; Collivignarelli et al., 2020; Deserti et al., 2020). On the other hand, the effect of the pandemic on NH_3 concentration has received comparatively less attention (Gualtieri et al., 2020; Lovarelli et al., 2020), although Northern Italy, and in particular Lombardy, is one of the leading regions for agriculture. Here, livestock production accounts for around 52% of pigs (ISMEA, 2019a), 20% of meat and dairy cattle (ISMEA, 2019b, c), and 17% of poultry (ISMEA, 2020) of the whole Country. The lower interest towards NH_3, compared to PM concentrations could be explained by the identification of PM particles as possible vectors for transporting the SARS-CoV-2 virus and for their responsibility for respiratory and cardiovascular diseases (Li et al., 2018; Srivastava, 2021). However, NH_3 influence on secondary aerosol is significant as it is a recognized precursor (Perone, 2021). Moreover, Zheng et al. (2020) and Manigrasso et al. (2020) discussed the possibility that SARS-CoV-2 spread is favored by a mild alkaline pH of airborne particles, and thus related to ammonia-polluted environments such as the Po Valley. To support this hypothesis, in the world 28 hotspots were identified with an NH_3 column concentration above 0.5 mg/m^2, which were linked to either biomass burning and fires or (and especially) agricultural areas, in particular with agricultural valleys surrounded by mountains, such as the case of Po valley in Northern Italy. Perone (2021) also identified cities in Northern Italy as those with the highest mortality risk in the country.

The aim of this study is to analyze NH_3 air concentration in Lombardy, the region most affected by the pandemic (Altuwayjiri et al., 2021; Bonati et al., 2021; Perone, 2021), by using both data from control units at ground level and satellite observations to evaluate the temporal and spatial scale of NH_3 concentration before and during lockdown (i.e., the strict national lockdown occurred in Spring, 2020), highlighting relationships, similarities or differences between these two measurement solutions.

2. Methods

2.1. Data collection of ground-based observations

Ground-based observations were retrieved from the database of the Regional Environmental Protection Agency of Lombardy (ARPA Lombardia, 2020). Data from all the control units collecting NH_3 measurements in the Lombardy Region were considered; overall data from 12 control units were available. These latter measure NH_3 indirectly through special analyzers using the chemiluminescence technique, by which NH_3 is first oxidized to nitrogen oxide (NO) and its concentration in the air sample is measured alongside NO and NO_2 (nitrogen dioxide). In particular, hourly data on air NH_3 concentration (expressed as μg/m^3) collected by the control units in question were sourced.

Furthermore, from the same ARPA Lombardia control units and for the same period, the daily weather parameters of temperature (T; °C), relative humidity (RH; %), rainfall (R; mm) and wind speed (W; m/s) were downloaded to consider their effects on NH_3 air concentration. The main characteristics of the control units used to retrieve these data are shown in Table 1.

All control units were distinguished in "city", "country" (short for countryside) or "mountain" stations, to investigate the effect of COVID-19 on NH_3 air concentration in the city, most densely populated and characterized by traffic jams and industrial activities, in the countryside, where agricultural and livestock activities are most concentrated, and in mountain areas. This grouping was based on the zoning of Lombardy for air quality monitoring according to the Italian legislative decree 155/10 under the directive 50/08/CE on ambient air quality and cleaner air for Europe (ARPA Lombardia, 2020). According to the previous regulations, the stations located in *urban agglomerations* or *highly urbanized plains* have been considered as "city" stations, those located in the *plain area* as

Table 1
List of the control units used. Their location is provided as well as the grouping in different zones.

Province	Station Name	Analyzer model	Longitude (E°)	Latitude (N°)	Altitude (m)	Zoned as
Cremona	Corte de Cortesi	API 201E	10.0062	45.2785	57	Country
Cremona	Fatebenefratelli	TEI 17i	10.0438	45.1425	43	City
Cremona	Gerre Borghi	TEI 17i	10.0692	45.1095	36	City
Lecco	Colico	TEI 17i	9.3847	46.1381	229	Mountain
Lecco	Moggio	TEI 17i	9.4975	45.9128	1194	Mountain
Lodi	Bertonico	API 201E	9.6663	45.2335	65	Country
Mantua	Schivenoglia	TEI 17i	11.0761	45.0169	12	Country
Milan	Pascal	API 201E	9.2355	45.4790	122	City
Pavia	Folperti	TEI 17i	9.1646	45.1947	77	City
Pavia	Sannazzaro	ENVEA AC32e	8.9042	45.1028	87	Country

"country" and finally those located in the *pre-Alps, Apennines and mountains* as "mountain". The same grouping of stations was also adopted by Lonati and Cernuschi (2020). The zone shapefile was obtained from the Lombardy region geo-portal website (http://www.geoportale.regione.lombardia.it/) with the zone classification carried out at the municipality level. Fig. 1 shows the regional zoning for Lombardy and the geographical position of each control unit. To further investigate the land cover in each zone, the Corine Land Cover (CLC) classification (Feranec, 2016) from 2018 was reclassified to a binary layer with the classes agriculture/other. Agricultural land cover was 87% in the country class, 55% in the city class and 14% in the mountain class.

The temporal coverage of the collected data goes from the beginning of 2013 to the end of October 2020. This period was selected because the same data were available also from the satellite dataset. Some data were lacking or were characterized by excessive uncertainty; therefore, the dataset was cleaned before data processing. With respect to ground-based control units, NH_3 concentrations not included in ±3 standard deviations were excluded from the dataset. Data were averaged as daily measures in order to be merged with the weather data for further analyses. All data were grouped in 2 periods to allow comparisons between normal living conditions and the period of the pandemic: the first included data from 2013 to 2019, while the second those of 2020.

Statistical analyses were conducted using SAS version 9.4 (SAS Institute, Cary, NC, USA) statistical software. Descriptive and multivariate statistics were carried out on each meteorological variable and on NH_3 concentration. A general Linear Model procedure (GLM Proc) was carried out to identify a model predicting the air concentration of NH_3 based on the unit zoning, local weather, monitoring period, and their interactions. In particular, in the model the following class parameters were included: (i) the period, with the 2 levels of 2013–2019 and 2020, and (ii) the zone, with the 3 levels of city, country, and mountain; the weather variables of temperature, rainfall and wind speed and the months of the year were included as well, together with their interactions for (i) year, zone and month, and (ii) temperature, rainfall and wind speed.

2.2. Data collection of remote sensing observations

Remote sensing data were retrieved from the online freely available database of the IASI sensor (https://iasi.aeris-data.fr/nh3/), which is the Infrared Atmospheric Sounding Interferometer onboard the ESA's (European Space Agency) MetOp satellites. In particular, data observed by the MetOp-A and MetOp-B satellites were obtained for the period 2013–2020, as MetOp-B became operational in early 2013, and thus data from two satellites were available to obtain a higher number of observations. While a third satellite, MetOp-C has also been operational since 2019, IASI data from this platform were not used to avoid introducing a bias in the later period of the dataset. For both satellites, daily level 2 products were downloaded, which report total column NH_3 in

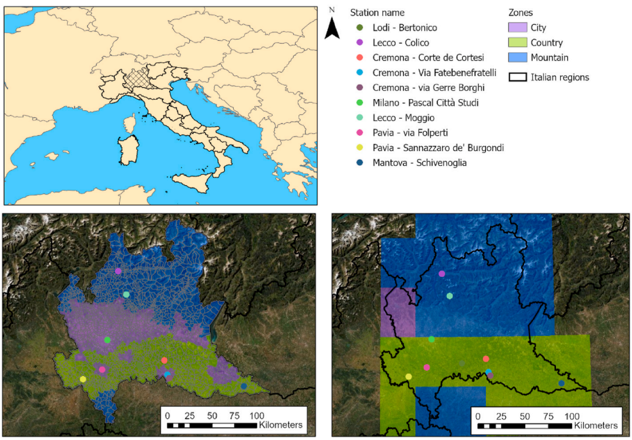

Fig. 1. Zoning of the Lombardy region (Northern Italy) in city (violet colored), country (green colored), and mountain (blue colored) zones. Colored dots represent the position of every ground-based control unit. Bottom-left: original zoning; bottom-right: the zoning resampled to IASI pixels. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

molecules/cm^2 and their relative uncertainty (in percentage) for discrete points observed by the satellite with an approximate footprint of 12 km (at nadir). Only daytime observations were used, as these are considered more accurate owing to the larger thermal contrast compared to nighttime observations (Van Damme et al., 2014). As cloud cover and thermal contrast in the atmospheric column greatly reduce the number of available measurements, daily observations lying partially or entirely within the Lombardy region were spatially re-gridded to a 0.5 ° × 0.5 ° grid (for a total of 26 grid points) and temporally averaged to obtain monthly means. This grid size is similar to the choice adopted by Van Damme et al. (2014) and allowed us to obtain on average 100 measurements per grid cell per month, thus producing more statistically robust NH_3 monthly estimations. The averaging was weighted based on the uncertainty of each measurement, following the procedure described by Van Damme et al. (2014), i.e. (equation (1)).

$$\overline{x} = \frac{\sum w_i x_i}{\sum w_i} \tag{1}$$

where x_i is a IASI measurement contained in the 0.5°x0.5° cell, w_i is the weighting factor, equal to $1/\sigma^2$ and σ is the error of the total column retrieval on a pixel basis.

The uncertainty of each monthly average was then expressed as

$$\overline{\sigma} = \frac{\sum \frac{1}{\sigma_i}}{\sum \frac{1}{\sigma_i^2}} \tag{2}$$

Moreover, it was established to discard monthly means for which the uncertainty was higher than 75% and measurements were fewer than 10 (30% of a month), as per recommendations by Van Damme et al. (2014).

Similar to the methodology adopted for the ground-based control units, satellite-based measurements were also divided into two periods, i.e., 2013–2019 and 2020, and classified as “city”, “country” or “mountain” based on the areas defined by ARPA Lombardia. In this case, the analysis was limited to the months from January to June owing to the unavailability of IASI data from June 2020 onwards at the time of writing. The zone shapefile was transformed to a raster and resampled to the same 0.5 × 0.5° grid used for IASI observations; each IASI pixel was then classified by assigning to it the zone with the largest count within the pixel (see Fig. 1). In addition, the CLC 2018 was resampled to the IASI grid and the percentage of agricultural land use was counted for each pixel. In the country zone, agricultural land use ranged between 50% and 83% except for one pixel (21%). In the mountain zone, it ranged between 4% and 21% while in the city zone it was 13%.

Statistical analyses were also conducted using SAS version 9.4 (SAS Institute, Cary, NC, USA) statistical software, with a General Linear Model procedure (GLM Proc) similar to the ground-based dataset; also in this case, a model predicting NH_3 air concentration based on the units zoning, local weather, monitoring period and their interactions was carried out for satellite data. The statistical model used for satellite data was the same as for ground-based data, to allow the best comparability of results; therefore, the parameters included in the model were class parameters for the period (levels of 2013–2019 and 2020) and for the zone (3 levels of city, country, and mountain) and the weather variables of temperature, rainfall and wind speed, the months of the year and the interactions among year, zone and month, as well as temperature, rainfall and wind speed.

To complement satellite observations with meteorological data in a similar way as done for ground NH_3 measurements, daily temperature, rainfall and wind speed from the reanalysis model ERA5 (European center for meteorology and weather forecast reanalysis, (Hersbach et al., 2020)) at 0.25° × 0.25° resolution were obtained. The values were then regridded to the same 0.5° × 0.5° grid of IASI observations and averaged to monthly values to conduct the GLM procedure. The steps of the adopted methodology are summarized for both ground-based and remote sensing observations in Fig. 2.

3. Results

3.1. Ground-based measurement

For what concerns NH_3 air concentrations, Fig. 3 reports the average concentration (μg/m^3) for the control units according to the classification in the zones of city, country, and mountain, with the mean and standard error for each month of the two periods 2013–2019 and 2020.

In both periods, the highest NH_3 concentrations can be observed for countryside stations. These stations are located in areas near agricultural

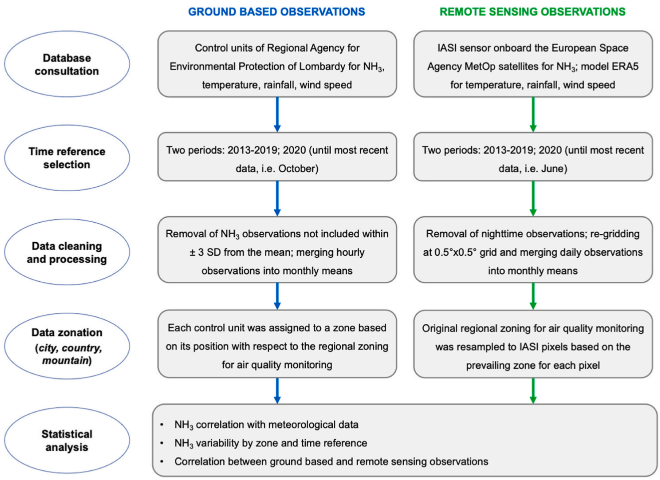

Fig. 2. Flow chart summarizing the main phases of the methodology adopted to organize and analyze the dataset.

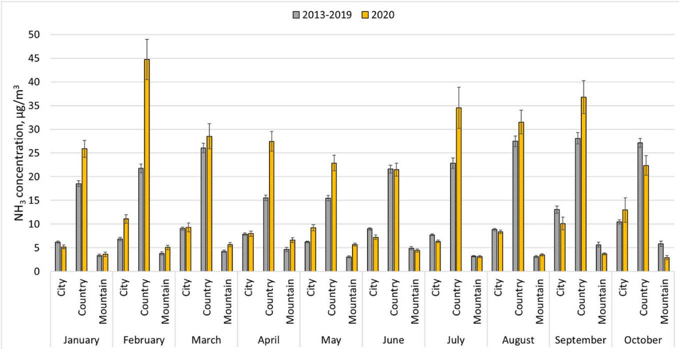

Fig. 3. Mean and standard error of NH_3 concentration, expressed in μg/m^3, per zone, month and period.

production sites, whose seasonal practices affect mostly NH_3 emissions. Statistically significant differences can be found among the three zones, with country zones showing much higher concentrations than stations located in city and mountain zones. In particular, the mean yearly value for country zones was 22.5 and 29.5 μg/m^3 for 2013–2019 and 2020, respectively. The highest concentrations were recorded in August and September for 2013–2019 (27.5 and 28.1 μg/m^3, respectively) and in February and September for 2020 (44.7 and 36.8 μg/m^3, respectively). The lowest average value of NH_3 concentration in the country zone (21.5 μg/m^3, June 2020) was higher than the highest of city and mountain zones (13.0 and 6.6 μg/m^3, October and April 2020, respectively).

In all cases, mountain stations show the lowest concentrations, with 2020 higher than 2013–2019 in all months except for June, July, September, and October. The yearly average is equal to 4.2 and 4.4 μg/m^3 in 2013–2019 and 2020, respectively. The city zone has intermediate values, with a yearly average equal to 8.6 and 8.8 μg/m^3 for 2013–2019 and 2020, respectively. Although for half the months considered, the city zone NH_3 concentration was higher in 2020 than in the previous period (i.e., February, March, April, May, and October), the annual average is still slightly higher in 2020. Interestingly, the months in which NH_3 in the city was higher than in the previous periods were the same months in which the strictest lockdown was in progress (i.e., spring 2020). A similar observation can be done with respect to the mountain zone.

Both country and city zones showed higher NH_3 concentration in 2020 than in the previous period in February, March, April, and May. Such higher concentration, especially the peak observed in February, can be attributed to manure management and manure field spreading. In particular, because field application of manure can be carried out depending on the constraints fixed by crops cultivation (sowing periods), by laws (European nitrates directive 91/676/EEC and the regional action programs for the protection of water pollution, Nitrates Directive (Eu), 2016), as well as by weather conditions (field spreading can be carried out when field conditions permit it, such as when no rainfall occurs), quite strict temporal windows can be identified during the year. Therefore, slurry application, and the related NH_3 air concentration were higher at the beginning of 2020 than in previous years probably because the unfavorable weather conditions of the previous autumn (i.e., in October 2019 high rainfall was observed at country zone stations with average rainfall of 115 mm, equal to 15% of the year) did not favor slurry spreading. Hence, a massive field application of slurry occurred at the beginning of 2020, which led to the consequent observation of high NH_3 concentrations. Moreover, during the months of lockdown, transport and industrial activities mostly stopped, and consequently also the related emissions decreased. The reduction of other air pollutants, such as sulfuric acid and nitric acid, with which NH_3 combines to form secondary PM (Ge et al., 2020), may have contributed to binding less NH_3 and a higher chance to find it in its free form.

The main zone differences mentioned above are reported in Fig. 4. Here, NH_3 air concentrations are reported split into 2 periods and 3 zones. In 2020, NH_3 concentration recorded at country stations were 69% of the total and were 31% higher than in 2013–2019. No significant differences between NH_3 values in 2013–2019 and 2020 can be observed for the city zone (+2%). In the mountain zone, instead, NH_3 concentration participates for 10% of the total, with +6% in 2020 compared to 2013–2019. The three zones of city, country, and mountain show statistically significant differences.

3.1.1. *Meteorological observations in relation to NH_3 air concentration*

Fig. 5 reports the average meteorological data of the analyzed provinces, with control units grouped by zone in the two selected periods for temperature (Fig. 5-top), rainfall (Fig. 5-middle), and wind speed (Fig. 5-bottom). Considering that Po Valley is located in Northern Italy and that it is mostly characterized by a temperate climate, without a dry season and with a hot summer (Beck et al., 2018), the meteorological data show a pattern consistent with these characteristics. In particular, regarding temperature a cold winter and a warm summer can be identified; rainfall is concentrated in spring and autumn and wind speed is low on average throughout the year.

In general, it can be observed that both city and country station units

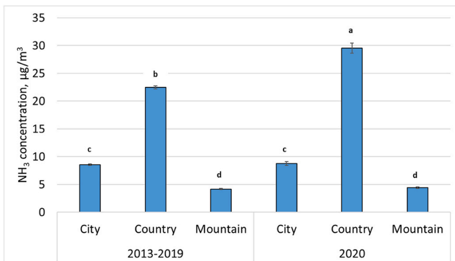

Fig. 4. Mean and standard error of NH_3 concentration, expressed in μg/m^3, per zone and period.

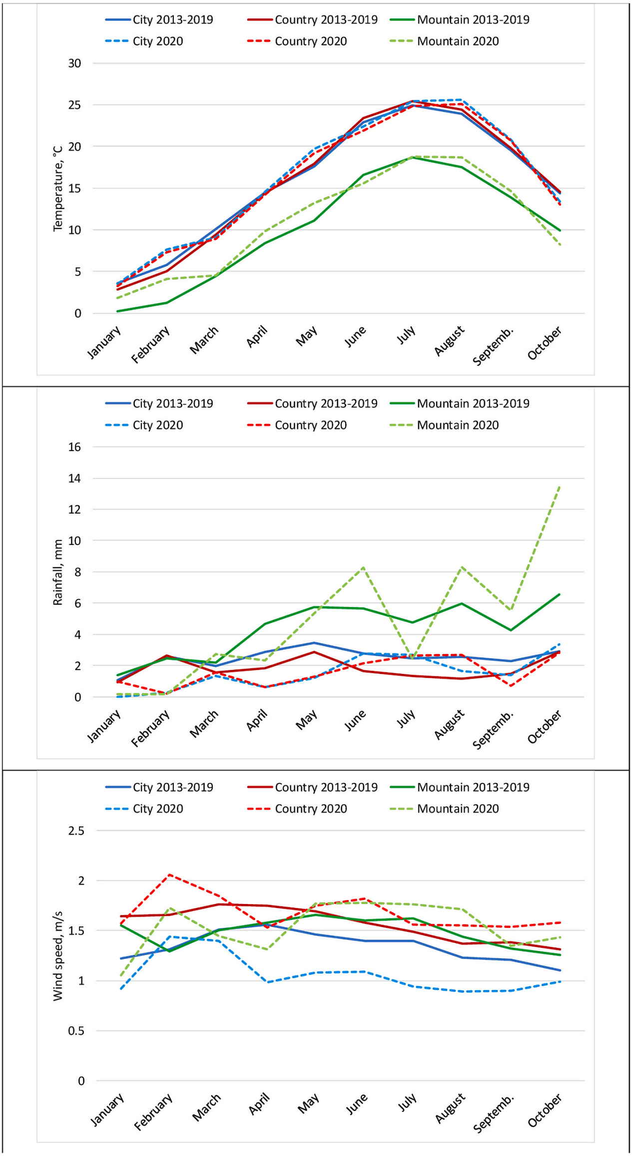

Fig. 5. Mean meteorological data (temperature -top, rainfall – middle, and wind speed -bottom) per zone (city, country, and mountain), month (Jan–Oct) and period (2013–2019, 2020).

recorded similar average yearly values for temperature, with 2020 showing slightly higher values (15.7 and 16.2 °C in 2013–2019 and 2020 in the city zone and 15.7 and 15.8 °C in the country zone, respectively). In the mountain zone they were lower (10.2 and 10.9 °C in 2013–2019 and 2020, respectively). The average rainfall was lower in 2020 compared to the previous period in the city (2.5 and 1.9 mm/d in 2013–2019 and 2020, respectively) and country (1.8 and 1.6 mm/d) zones, while in the mountain zone rainfall was higher (4.4 and 4.9 mm/d). As regards wind speed, all values were low: 1.3 and 1.1 m/s as average of 2013–2019 and 2020 in the city zone, country averaging 1.6 and 1.7 m/s and mountain averaging 1.5 m/s in both periods. Pearson correlations were calculated (data not shown) and significant differences were found among all the meteorological parameters considering the different zones and months. Although the linear relationship is not very strong, temperature is significantly negatively correlated with rainfall and wind speed. In contrast, rainfall and wind speed are positively correlated with each other. The changes in NH_3 air concentration and meteorological parameters during the period of observation are reported in Table 2.

The relative difference between the periods 2020 and 2013–2019 is carried out by using the data reported in Figs. 3 and 5. From these data, it emerges that NH_3 concentration in air was higher in 2020 than in the previous period from February to May for all zones and this occurred, in most cases, together with higher temperatures. Except for these four months and October, in city zones NH_3 was lower in 2020 than over the previous years, which occurred with changes in the meteorological parameters that, however, do not allow to identify clear links between NH_3 and weather patterns, and especially with rainfall/wind speed.

To clarify the relationships among these variables, a GLM procedure was performed to obtain a predictive model for NH_3 concentration in air. The model was generated on the classes of year (2 levels: 2013–2019 and 2020) and zone (3 levels: city, country, and mountain) and was found statistically significant, although with a very low coefficient of determination ($R^2 = 0.19$). All variables were also significant with "year" and "zone" very significant ($p < 0.0001$) and month, weather parameters and their interactions significant ($p < 0.05$). Among the meteorological parameters, rainfall was the variable with the lowest significance for the prediction of NH_3. The calculated Least Squares Means (LS Means) were all significant ($p < 0.0001$). However, the statistical significance of LS Means for the effect year*zone showed no difference between the 2 periods for the city and the mountain zones as reported in Table 3.

This result shows that, in the two studied periods, both city zones and mountain zones have comparable results. Since NH_3 is mostly released in countryside areas and normally deposited within a short radius from the emissive source, it can be expected that lower concentrations reach cities and mountain zones and that these latter do not show significant

Table 2
Percentage change between 2020 and 2013–2019 for NH_3 air concentration, temperature (T), rainfall (R) and wind speed (W). When the resulting changes are positive (year 2020 > years 2013–2019) cells have a green background.

Month	Zone	NH_3	T	R	W
January	City	-15.4%	-3.3%	-34.0%	-24.6%
	Country	39.6%	15.5%	1.0%	-4.3%
	Mountain	6.5%	682.6%	-87.7%	-32.3%
February	City	62.4%	32.6%	-90.0%	9.9%
	Country	105.6%	44.4%	-91.3%	24.1%
	Mountain	32.8%	231.2%	-92.7%	34.1%
March	City	2.8%	-10.3%	-31.0%	-7.3%
	Country	9.5%	-5.7%	1.3%	5.1%
	Mountain	33.2%	1.6%	26.0%	-3.3%
April	City	1.1%	0.5%	-78.7%	-37.2%
	Country	76.7%	-1.0%	-64.9%	-12.6%
	Mountain	41.5%	16.1%	-50.2%	-17.1%
May	City	48.2%	11.9%	-64.9%	-26.0%
	Country	48.2%	6.9%	-54.3%	3.6%
	Mountain	83.8%	19.1%	-7.3%	6.6%
June	City	-19.7%	-2.3%	1.1%	-22.1%
	Country	-0.6%	-6.6%	28.1%	15.2%
	Mountain	-9.8%	-6.1%	47.0%	11.3%
July	City	-17.5%	2.2%	9.7%	-32.9%
	Country	51.4%	-2.2%	94.8%	4.7%
	Mountain	-1.9%	0.8%	-49.2%	8.6%
August	City	-5.2%	7.1%	-34.6%	-27.6%
	Country	14.7%	2.5%	133.9%	13.1%
	Mountain	11.9%	6.8%	39.0%	18.8%
September	City	-22.3%	6.3%	-40.7%	-25.6%
	Country	31.0%	4.5%	-52.7%	11.6%
	Mountain	-33.7%	5.5%	28.7%	2.3%
October	City	24.2%	-7.3%	15.3%	-10.0%
	Country	-17.6%	-10.3%	-1.4%	20.6%
	Mountain	-49.7%	-16.8%	104.1%	13.5%

Table 3
Statistical significance of Least Squares Means resulting from the GLM procedure for the effect year*zone and NH_3 as dependent variable.

i/j	2013–2019 country	2013–2019 mountain	2020 city	2020 country	2020 mountain
2013–2019 city	***	***	n.s.	***	***
2013–2019 country		***	***	***	***
2013–2019 mountain			***	***	n.s.
2020 city				***	***
2020 country					***

Notes: *** = p < 0.0001; n.s. = not significant.

differences in the analyzed periods. Since agricultural activities are more subject than others to annual variability and seasonality, the fact that NH_3 in countryside areas is significantly different between 2013 and 2019 and 2020 prompted us to investigate the effects of seasonality on NH_3 every year. Thus, the model was relaunched evaluating the effect of each year on NH_3, focusing only on the country zone. This model was found significant with the coefficient of determination $R^2 = 0.29$ and all weather parameters significant ($p < 0.05$). Significant differences for NH_3 concentration emerged in some years, while no differences were found between (i) 2013 and 2016, (ii) 2014 and 2015, and (iii) 2017, 2018 and 2020. The results of this second analysis suggest that meteorological variability can play a role on NH_3 air concentration in country zones.

3.2. *Satellite measurements*

The NH_3 total column observations from IASI data reported in Fig. 6 show slightly different patterns compared to ground measurements, although it is a shared feature that country zones show the highest values in all months during 2013–2019 and in 2020. The highest total column NH_3 was recorded for grid points classified as "country" in June, both over 2013–2019 ($5.89e^{-4}$ mol/cm^2) and in 2020 ($5.31e^{-4}$ mol/cm^2); over 2013–2019, May was the second highest month with respect to NH_3 values ($4.59e^{-4}$ mol/cm^2), while in 2020 it was March, when the lockdown started ($5.22e^{-4}$ mol/cm^2). In March 2020, mountain grid points also show very high values ($5.11e^{-4}$ mol/cm^2), possibly because of the actual inclusion of agricultural areas within grid points classified as mountain at the IASI scale (maximum agricultural cover was 21% in mountain grid points from CLC, 2018). As a further hint of this, compared to ground observations, city zones show generally lower total column NH_3 than mountain zones, with the lowest values overall recorded for city zones in January 2020 ($1.34e^{-4}$ mol/cm^2). For country zones, all months except June show an increase in total column NH_3 compared to the average of 2013–2019. For the other zones, the pattern is more variable: city grid points show a higher NH_3 total column in February, March, April, and June 2020 while for mountain stations the months with a higher NH_3 total column are January, March (with an increase of 108% compared to the same month over 2013–2019), April and June.

As shown in Fig. 7, averaging all monthly observations between January and June, mountain and country zones show an increase in NH_3 total column in 2020 compared to 2013–2019, with similar values (18% for mountain zones and 15% for country zones).

As for ground observations, the NH_3 total column for city zones is unvaried between the two periods (−0.73%). In contrast with ground observations, however, the share of NH_3 for the different zones is more uniform: in 2020, country zones accounted for 42% of the total column, mountain zones for 27% and city zones for 31%. This might be caused by the coarse spatial grid of IASI measurements and mix between different land use classes at the IASI scale.

3.2.1. *Meteorological observations in relation to NH_3 air concentration*

As for ground observations, monthly meteorological data and NH_3 total column divided in the two periods were compared. Table 4 reports such results.

In most months, temperature was higher in 2020 than in 2013–2019, particularly in the mountain zones where it was always higher except for March, with an increase of 449.99% in February 2020 compared to 2013–2019. In contrast, precipitation was lower in most months, except for mountain zones in March and all zones in June, while wind speed showed a more varied pattern with a decrease in January 2020 and an increase in May 2020 in all zones.

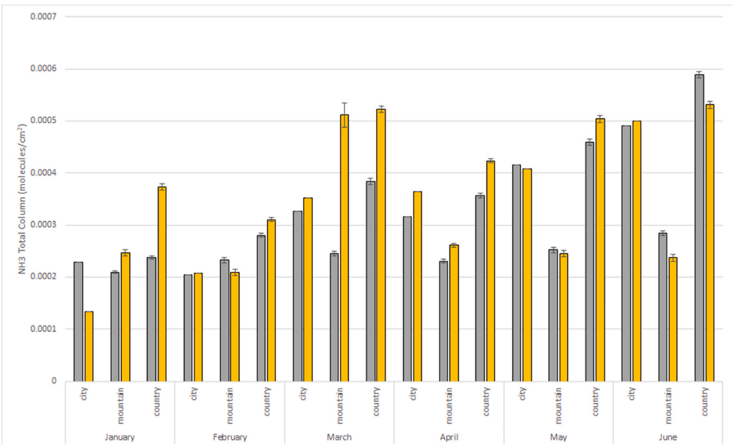

Fig. 6. Mean and standard error of total column NH_3 from IASI observations for each month (January–June) over the period 2013–2019 and 2020 for each of the three zones considered (city, mountain, country).

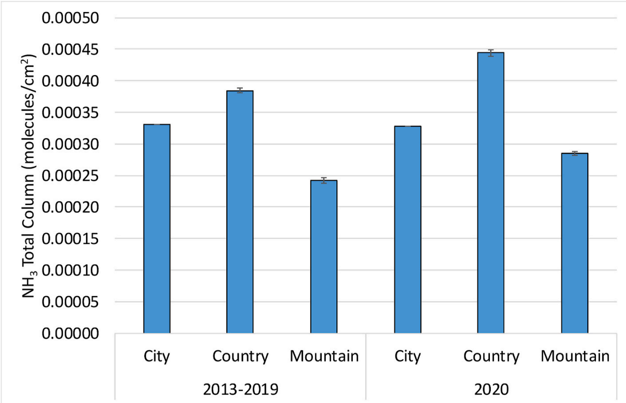

Fig. 7. Mean and standard error of NH_3 total column for January–June in 2013–2019 and 2020 for the three zones considered (city, mountain, country).

Table 4
Percentage change between 2020 and 2013–2019 for NH_3 total column, temperature (T), rainfall (R) and wind speed (W) from IASI and ERA-5 observations. When the resulting changes are positive (year 2020 > years 2013–2019) cells have a green background.

Month	Zone	NH_3	T	R	W
January	City	-41.30%	12.53%	-81.74%	-15.22%
	Country	57.09%	8.67%	-66.28%	-5.94%
	Mountain	17.62%	50.67%	-69.93%	-2.42%
February	City	1.53%	50.45%	-87.94%	-1.07%
	Country	10.67%	48.31%	-90.94%	48.47%
	Mountain	-9.82%	449.99%	-66.97%	32.27%
March	City	7.72%	-5.11%	-4.30%	4.73%
	Country	36.02%	-4.51%	-22.97%	46.01%
	Mountain	108.41%	-5.71%	14.73%	-9.26%
April	City	15.62%	3.15%	-57.28%	-47.02%
	Country	18.64%	-0.05%	-52.21%	-24.42%
	Mountain	13.32%	9.78%	-57.93%	28.69%
May	City	-1.68%	9.32%	-10.92%	24.81%
	Country	9.80%	9.55%	-36.25%	46.39%
	Mountain	-2.54%	16.17%	-12.84%	26.45%
June	City	1.84%	-7.49%	49.63%	-11.88%
	Country	-9.85%	-5.65%	47.79%	41.90%
	Mountain	-16.59%	9.58%	44.44%	32.56%

A GLM procedure was carried out to identify a prediction model for NH_3 total column, in a similar way as done for ground-based measurements. The model was found significant with a coefficient of determination of 0.55. The main difference with the GLM for ground-based observations is related to the fact that none of the meteorological parameters was found statistically significant, and only the zone parameter and month*zone effect were significant in the prediction of NH_3. In addition, no statistical difference emerged from the evaluation of LS Means for year*zone, as reported in Table 5.

The result of the GLM procedure with satellite NH_3 observations and meteorological parameters shows no statistical difference with respect to the NH_3 concentration in the air between the two periods. This is in contrast with the ground-based measurements, which showed a significant difference for NH_3 concentrations in country stations, between 2013 and 2019 and 2020. The reason for this result may be due to the

Table 5
Statistical significance of LS Means resulting from the GLM procedure for the effect year*zone and NH_3 as dependent variable.

i/j	2013–2019 country	2013–2019 mountain	2020 city	2020 country	2020 mountain
2013–2019 city	n.s.	n.s.	n.s.	n.s.	n.s.
2013–2019 country		n.s.	n.s.	n.s.	n.s.
2013–2019 mountain			n.s.	n.s.	n.s.
2020 city				n.s.	n.s.
2020 country					n.s.

susceptibility of control units to a series of site-specific aspects that, instead, cannot be measured in the satellite coarse grid, and the fact that satellites measure total column NH_3, which might be the result of emissions originating in a different grid point than the footprint observed by the satellite at the time of data acquisition.

4. Discussion

This study was carried out with a focus on comparing the period (i.e., from year 2013–2019) before the lockdown determined by the emergency of COVID-19 and the subsequent period during the pandemic (i.e., year 2020, with the strict lockdown lasting from March–June 2020), adopting two measurement systems. Remote sensing datasets show no significant difference in NH_3 air concentration between the period before the spread of COVID-19 and the one during the pandemic. From the ground based analysis, NH_3 was observed higher than other years in city and mountain, and significantly in country zones. As NH_3 is strongly dependent on agricultural and livestock activities, its concentration was found subject to seasonality, weather conditions and to agricultural management. This aspect has been underlined also in the study of Lonati and Cernuschi (2020) and Deserti et al. (2020). The same result was observed also by Gualtieri et al. (2020) who state that, in 2020, NH_3 concentration increased in Italy compared to 2019. They also report that more than 90% of NH_3 is of agricultural origin in Milan, while this contribution decreases to 71% in Rome and 62% in Bologna, suggesting once more the relevance of livestock activities in Northern Italy. Therefore, the role of agriculture and livestock appears to be the largest influence on NH_3 air concentrations. Given the relationship of weather and seasonality with agricultural field activities, and the characteristics of NH_3emissions (Brentrup et al., 2000; Sutton et al., 2013), it can be expected that NH_3 concentration is higher in summer (high temperatures) than in winter (when regulations prohibit field spreading due to nitrate leaching), and that peaks are observed in periods before crop cultivation (generally from February to June and from August to October in the analyzed area, where double cropping is widespread) when base-dressing fertilizers are applied on the field (Guido et al., 2020; Pedersen et al., 2020). All these aspects are confirmed also by the present study. Independently from the period considered, the highest NH_3 concentration can be observed in February–March and July–October, compared to the other months.

Moreover, different values in NH_3 concentration are strictly linked to manure storage (Zilio et al., 2020) and to field application, since this operation can be carried out when no rainfall occurs and in agreement with the European nitrates directive 91/676/EEC and the regional action programs for the protection of water pollution caused by nitrates from agricultural sources. These aspects have been widely investigated in literature, such as by Skjøth et al. (2011), Ramanantenasoa et al. (2018), and Ge et al. (2020).

4.1. Limitations of ground-based and satellite observations

In this study, measurements from both ground control units and satellite sensors were used to investigate air concentrations of NH_3. The two techniques showed similar patterns, such as the higher yearly average concentrations in country zones in 2020 compared to 2013–2019 but also some differences for individual months and in the relative contribution of the different zones to total NH_3 concentrations, which are related to the limitations inherent in measurement techniques.

For the entire Lombardy region (surface area about 23000 km^2), the availability of NH_3 measured data for the analyzed period was limited to 10 control units for the assessment using ground stations. However, NH_3 concentration may vary considerably within few kilometers (Lonati and Cernuschi, 2020). This confirms the scarcity of NH_3 control units compared to other air pollutants; in fact, in the same area available stations amount to 90 units for PM_{10} and 38 units for $PM_{2.5}$ (ARPA Lombardia, 2020). Moreover, for the same control units, air concentrations of different pollutants, such as NH_3 and PM, are not always available, making comparisons more difficult.

As concerns satellite images, the number of measurements, after filtering for observations with high uncertainty, did not allow us to create a grid finer than $0.5 \times 0.5^{\circ}$, which would have introduced a large number of data gaps. The coarse grid however complicates the assessment of local variability; the comparison between data gathered from each ground based control unit and satellite observations is problematic because of 1) the different spatial resolution; in fact, ground stations classified as "city" might be included in country zones in IASI pixels, with a much larger contribution of NH_3 air concentration from agricultural areas; 2) the different unit of measure and assessment method, as one is a ground level observation, the other is a column observation. An attempt to correlate monthly data from ground and satellite observations showed an r^2 of 0.21–0.29; this is in line with Van Damme et al. (2015), who report Pearson's correlations of 0.28 comparing monthly data from IASI and ground observations from a global network, ranging from 0.81 (Russian Fyodorovskoye site, with very high NH_3 concentrations owing to fire events) to negative correlations at several sites in Finland, probably caused by the low concentrations at these sites in association with low temperatures and thus the low thermal contrast of IASI.

Given the limitations of both measurement approaches, among which the temporal and spatial resolution, improvements should be introduced both in the number and density of ground stations, especially in areas where agriculture is widespread, and in the availability of satellite data with a higher spatial resolution, which might also make validation efforts easier in comparison with ground observations.

4.2. Prospects for improving air quality and opportunities for further study

In the future, a significant reduction in NH_3 air concentrations is expected as a consequence of the abatement measures that are being introduced in agricultural and livestock farms (e.g., closed tank storages, manure and slurry treatments, precision application of slurry on field) (Finzi et al., 2019; Zilio et al., 2020), although these measures are currently present only in few contexts. Considerable improvements have already been introduced in the European Union, which brought to a strong reduction of NH_3 air concentration, reaching −24% from 1990 to 2017 (Costantini et al., 2020). However, this trend of reduction is proceeding further; in fact, in some countries, regions, and farms additional improvements are being introduced or are under study. For example, Miranda et al. (2021) investigated through the Life Cycle Assessment method, the environmental implications of treating slurry and found positive results for the reduction of gases, among which NH_3, with the addition of sulfuric acid to slurry. Slurry acidification was found effective in reducing NH_3 and other greenhouse gases (GHGs) also by Fangueiro et al. (2015). However, supporting farmers in the direction of abating NH_3 is fundamental to increase the spread of such ameliorative techniques and solutions. The chief aspects on which to focus include efficient livestock rearing techniques supported by technology, balanced animal feed rations, air scrubbers in barns, adequate manure management with frequent removal from the barn, manure treatments such as anaerobic digestion, solid-liquid separation, slurry acidification, proper manure storage with closed tanks and proper field application with precision application equipment (Guido et al., 2020; Regueiro et al., 2016).

Importantly from the environmental point of view, avoiding NH_3 losses can lead to maintaining nutrients in the manure/slurry and therefore applying more nutrients to the soil when spreading manure or slurry, thus requiring fewer mineral fertilizers and bringing benefits to the environmental sustainability, plus valorizing an already available resource that is free of charge. Finally, while high NH_3 concentration in air may represent a local issue, damages to ecosystems from NH_3 are not only local and are widely investigated due to the main effects of acidification and eutrophication on biodiversity loss in coastal and estuarial areas, such as in the study by Vetterli et al. (2016).

Despite the improvements in agricultural practices, this study has shown that NH_3 concentrations in air remained high in areas such as the Po Valley even during the lockdown caused by the spread of COVID-19. To further validate the considerations made regarding NH_3 air concentrations during 2020, future research should consider a wider area (e.g. Italy, Europe), as well as expand the analysis to other air pollutants. To this end, it would also be interesting to integrate ground-based and satellite datasets with physico-chemical dispersion models (e.g. LOTOS-EUROS, Schaap et al., 2008; also employed by Van Damme et al., 2014, 2015) to better understand their origin and interactions, while in this study a purely statistical comparative approach was adopted.

5. Conclusions

This study aimed to evaluate the atmospheric concentrations of NH_3 during the spread of the COVID-19 pandemic in Northern Italy and the related changes in anthropogenic activities. For this purpose, ground-based and satellite measurement data relating to 2020 were analyzed and compared with previous years (2013–2019). Ground-based measurements showed statistically significant differences between the two periods in the country areas, where NH_3 was found higher in 2020 (+31% compared to the 2013–2019 average); on the other hand, no significant differences emerged for the city and mountain areas. Satellite data show similar patterns, but no significant differences between the two periods and less spatial variability between city, country, and mountain areas, probably due to the coarse spatial grid of IASI measurements. Contrary to other air pollutants, it can be concluded that no NH_3 reduction effect has occurred as a consequence of the anti-COVID-19 measures, which can be explained by the non-interruption of agricultural activities, the main emissive source of this pollutant. The integration between datasets from different measurement sources allows having a broader understanding of atmospheric phenomena, since both methods alone have their limitations. These considerations offer insights into the physico-chemical modeling of this pollutant and the actions aimed at its mitigation.

CRediT authorship contribution statement

Daniela Lovarelli: Conceptualization, Methodology, Formal analysis, Investigation, writing: original daft, writing: review and editing. **Davide Fugazza:** Methodology, Formal analysis, Investigation, writing: original daft, writing: review and editing. **Michele Costantini:** Investigation, writing: original daft, writing: review and editing. **Cecilia Conti:** Investigation, writing: original daft, writing: review and editing. **Guglielmina Diolaiuti:** Supervision. **Marcella Guarino:** Conceptualization, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

References

Altuwayjiri, A., Soleimanian, E., Moroni, S., et al., 2021. The impact of stay-home policies during Coronavirus-19 pandemic on the chemical and toxicological characteristics of ambient PM2.5 in the metropolitan area of Milan, Italy. Sci. Total Environ. 758 (1), 143582. https://doi.org/10.1016/j.scitotenv.2020.143582.

Beck, H., Zimmermann, N., McVicar, T., Vergopolan, N., Berg, A., Wood, E.F., 2018. Present and future Köppen-Geiger climate classification maps at 1-km resolution. Scientific Data 5, 180214. https://doi.org/10.1038/sdata.2018.214.

Bonati, M., Campi, R., Zanetti, M., Cartabia, M., Scarpellini, F., Clavenna, A., Segre, G., 2021. Psychological distress among Italians during the 2019 coronavirus disease (COVID-19) quarantine. BMC Psychiatr. 21, 20. https://doi.org/10.1186/s12888-020-03027-8.

Brentrup, F., Küsters, J., Lammel, J., Kuhlmann, H., 2000. Methods to estimate on-field nitrogen emissions from crop production as an input to LCA studies in the agricultural sector. Int. J. Life Cycle Assess. 5, 349–357. https://doi.org/10.1007/BF02978670.

Buganza, E., Donzelli, M., Gurrieri, G.L., Lazzarini, M., Scotto Di Marco, E., Bravetti, E., Dal Santo, U., Di Leo, A., Lanzani, G., Cazzuli, O., Minardi, G.P., 2020. Preliminary analysis of air quality in Lombardy during the COVID-19 emergency (in Italian). Agenzia Regionale per la Protezione dell'Ambiente – Lombardia. Available at: https://www.arpalombardia.it/sites/DocumentCenter/Documents/Aria%20-%20Relazioni%20approfondimento/Analisi%20preliminare%20QA-COVID19.pdf. (Accessed 9 November 2020).

Collivignarelli, M.C., Abbà, A., Bertanza, G., Pedrazzani, R., Ricciardi, P., Miino, M.C., 2020. Lockdown for CoViD-2019 in Milan: what are the effects on air quality? Sci. Total Environ. 732, 139280. https://doi.org/10.1016/j.scitotenv.2020.139280.

Costantini, M., Bacenetti, J., Coppola, G., Orsi, L., Ganzaroli, A., Guarino, M., 2020. Improvement of human health and environmental costs in the European Union by air scrubbers in intensive pig farming. J. Clean. Prod. 275, 1244007. https://doi.org/10.1016/j.jclepro.2020.124007.

Deserti et al.,, 2020. Preliminary Study of the Effects of COVID-19 Measures on Emissions in the Atmosphere and on Quality of the Air in the Po Valley (In Italian). Report from the LIFE PrepAIR Project, August 2020. Report 2 Covid-19. Available at: https://www.lifeprepair.eu/wp-content/uploads/2020/09/COVIDQA-Prepair-2-17Settembre2020.pdf. (Accessed 14 December 2020).

Dpcm, 2020. Decree of the President of the Council of Ministers of the Italian Republic of March 22, 2020 - Further Implementing Provisions of the Decree-Law of February 23, 2020, N. 6, Containing Urgent Measures Regarding the Containment and Management of the Epidemiological Emergency from COVID-19 (In Italian). President of the Council of Ministers of the Italian Republic, Rome, Italy. Available at: https://www.gazzettaufficiale.it/eli/id/2020/03/22/20A01807/sg. (Accessed 14 December 2020).

Eea, European Environmental Agency, 2019. European union emission inventory report 1990-2017, under the UNECE convention on long-range transboundary air pollution (CLRTAP). EEA Report | No 08/2019.

Emep Centre, 2020. On Emission Inventories and Projections. Available at: https://www.ceip.at/webdab-emission-database. (Accessed 2 February 2021).

Erisman, J., Bleeker, A., Galloway, J., Sutton, M., 2007. Reduced nitrogen in ecology and the environment. Environ. Pollut. 150, 140–149. https://doi.org/10.1016/j.envpol.2007.06.033.

European Commission, 2005. Communication from the commission to the council and the European Parliament. "Thematic Strategy on air pollution.". Available at: https://ec.europa.eu/environment/archives/cafe/pdf/annex_sec_2005_1132_en.pdf. (Accessed 2 February 2021).

Executive Body of the Convention on Long-Range Transboundary Air Pollution, 2019. Draft of assessment report on ammonia. Available at: https://unece.org/fileadmin/DAM/env/documents/2019/AIR/EMEP_WGE_Joint_Session/Assessment_Report_on_Ammonia_20190827.pdf. (Accessed 2 February 2021).

Fangueiro, D., Pereira, J., Bichana, A., Surgy, S., Cabral, F., Coutinho, J., 2015. Effects of cattle-slurry treatment by acidification and separation on nitrogen dynamics and global warming potential after surface application to an acidic soil. J. Environ. Manag. 162, 1–8. https://doi.org/10.1016/j.jenvman.2015.07.032.

Feranec, I., Soukup, T., Hazeu, G., Jaffrain, G. (Eds.), 2016. European Landscape Dynamics. CRC Press, Boca Raton.

Finzi, A., Riva, E., Bicoku, A., Guido, V., Shallari, S., Provolo, G., 2019. Comparison of techniques for ammonia emission mitigation during storage of livestock manure and assessment of their effect in the management chain. Journal of Agricultural Engineering 50, 12–19. https://doi.org/10.4081/jae.2019.881.

Ge, X., Schaap, M., Kranenburg, R., Segers, R., Reinds, G.J., Kros, H., de Vries, W., 2020. Modeling atmospheric ammonia using agricultural emissions with improved spatial variability and temporal dynamics. Atmos. Chem. Phys. 20, 16055–16087. https://doi.org/10.5194/acp-20-16055-2020.

Gualtieri, G., Brilli, L., Carotenuto, F., Vagnoli, C., Zaldei, A., Gioli, B., 2020. Quantifying road traffic impact on air quality in urban areas: a Covid19-induced lockdown analysis in Italy. Environ. Pollut. 267, 115682.

Guido, V., Finzi, A., Ferrari, O., Riva, E., Quílez, D., Herrero, E., Provolo, G., 2020. Fertigation of maize with digestate using drip irrigation and pivot systems. Agronomy 10 (10), 1453. https://doi.org/10.3390/agronomy10101453.

Hersbach, Hans, Bell, Bill, Berrisford, Paul, Hirahara, Hoji, et al., 2020. The ERA5 Global Reanalysis. Q. J. R. Meteorol. Soc. 146, 1999–2049. https://doi.org/10.1002/qj.3803.

Ismea, 2019a. Settore lattiero caseario - Scheda di settore (Last accessed. http://www.ismeamercati.it/flex/cm/pages/ServeBLOB.php/L/IT/IDPagina/3521. (Accessed 14 December 2020).

Ismea, 2019b. Allevamento bovino da carne – Scheda di settore (Last accessed. http://www.ismeamercati.it/flex/cm/pages/ServeBLOB.php/L/IT/IDPagina/3515. (Accessed 14 December 2020).

Ismea, 2019c. Settore suinicolo – Scheda di settore (Last accessed. http://www.ismeamercati.it/flex/cm/pages/ServeBLOB.php/L/IT/IDPagina/3516. (Accessed 14 December 2020).

Ismea, 2020. Filiera avicola – Scheda di settore. Last accessed. http://www.ismeamercati.it/carni/avicoli-uova. (Accessed 14 December 2020).

Li, T., Hu, R., Chen, Z., Li, Q., Huang, S., Zhu, Z., Zhou, L.-F., 2018. Fine particulate matter (PM2.5): the culprit for chronic lung diseases in China. Chronic Diseases and Translational Medicine 4, 176–186.

Lombardia, Arpa, 2020. Agenzia Regionale per la Protezione dell'Ambiente – Lombardia. Last accessed. https://www.arpalombardia.it/Pages/ARPA_Home_Page.aspx. (Accessed 9 November 2020).

Lonati, G., Cernuschi, S., 2020. Temporal and spatial variability of atmospheric ammonia in the Lombardy region (Northern Italy). Atmospheric Pollution Research 11, 2154–2163. https://doi.org/10.1016/j.apr.2020.06.004.

Lovarelli, D., Conti, C., Finzi, A., Bacenetti, J., Guarino, M., 2020. Describing the trend of ammonia, particulate matter and nitrogen oxides: the role of livestock activities in northern Italy during Covid-19 quarantine. Environ. Res. 191, 110048. https://doi.org/10.1016/j.envres.2020.110048.

Manigrasso, M., Protano, C., Guerriero, E., Vitali, M., Avino, P., 2020. May SARS-CoV-2 diffusion be favoured by alkaline aerosols and ammonia emissions? Atmosphere 11, 995. https://doi.org/10.3390/atmos11090995.

Miranda, C., Soares, A.S., Coelho, A.C., Trinidade, H., Teixeira, C.A., 2021. Environmental implications of stored cattle slurry treatment with sulphuric acid and biochar: a life cycle assessment approach. Environ. Res. 194, 110640. https://doi.org/10.1016/j.envres.2020.110640.

Nair, A.A., Yu, F., 2020. Quantification of atmospheric ammonia concentrations: a review of its measurement and modeling. Atmosphere 11, 1092. https://doi.org/10.3390/atmos11101092.

Nitrates Directive, Eu, 2016. 2016/2284 of the European Parliament and of the Council of 14 December 2016 on the reduction of national emissions of certain atmospheric pollutants, amending Directive 2003/35/EC and repealing Directive 2001/81/EC. Orkesterjournalen L 344/1, 12–17.

Nuñez-Delgado, A., Zhou, Y., Domingo, J.L., 2021. Editorial of the VSI "Environmental, ecological and public health considerations regarding coronaviruses, other viruses, and other microorganism potentially causing pandemic diseases.". Environ. Res. 192, 110322. https://doi.org/10.1016/j.envres.2020.110322.

Pedersen, I.F., Rubaek, G.H., Nyord, T., Sorensen, P., 2020. Row-injected cattle slurry can replace mineral P starter fertiliser and reduce P surpluses without compromising final yields of silage maize. Eur. J. Agron. 116, 126057. https://doi.org/10.1016/j.eja.2020.126057.

Perone, G., 2021. The determinants of COVID-19 case fatality rate (CFR) in the Italian regions and provinces: an analysis of environmental, demographic, and healthcare factors. Sci. Total Environ. 755 (1), 142523. https://doi.org/10.1016/j.scitotenv.2020.142523.

Raffaelli, K., Deserti, M., Stortini, M., Amorati, R., Vasconi, M., Giovannini, G., 2020. Improving air quality in the Po valley, Italy: some results by the LIFE-IP-PREPAIR project. Atmosphere 11, 429. https://doi.org/10.3390/atmos11040429.

Ramanantenasoa, M.M.J., Gilliot, J.M., Mignolet, C., Bedos, C., Mathias, E., Eglin, T., Makowski, D., Genermont, S., 2018. A new framework to estimate spatio-temporal ammonia emissions due to nitrogen fertilization in France. Sci. Total Environ. 645, 205–219. https://doi.org/10.1016/j.scitotenv.2018.06.202.

Regueiro, I., Coutinho, J., Gioelli, F., Balsari, P., Dinuccio, E., Fangueiro, D., 2016. Acidification of raw and co-digested pig slurries with alum before mechanical separation reduces gaseous emission during storage of solid and liquid fractions. Agric. Ecosyst. Environ. 227, 42–51. https://doi.org/10.1016/j.agee.2016.04.016.

Schaap, M., Timmermans, R., Roemer, M., Boersen, G., Builtjes, P., Sauter, F., Velders, G., Beck, J., 2008. The LOTOS-EUROS model: description, validation and latest developments. Int. J. Environ. Pollut. 32, 270–290. https://doi.org/10.1504/IJEP.2008.017106.

Skjøth, C.A., Geels, C., Berge, H., Gyldenkærne, S., Fagerli, H., Ellermann, T., Frohn, L. M., Christensen, J., Hansen, K.M., Hansen, K., Hertel, O., 2011. Spatial and temporal variations in ammonia emissions – a freely accessible model code for Europe. Atmos. Chem. Phys. 11, 5221–5236. https://doi.org/10.5194/acp-11-5221-2011.

Srivastava, A., 2021. COVID-19 and air pollution and meteorology – an intricate relationship: a review. Chemosphere 263, 128297. https://doi.org/10.1016/j.chemosphere.2020.128297.

Sutton, M.A., Reis, S., Riddick, S.N., Dragosits, U., Nemitz, E., Theobald, M.R., Tang, Y.S., Braban, C.F., Vieno, M., Dore, A.J., Mitchell, R.F., Wanless, S., Daunt, F., Fowler, D., Blackall, T.D., Milford, C., Flechard, C.R., Loubet, B., Massad, R., Cellier, P., de Vries, W., 2013. Towards a climate-dependent paradigm of ammonia emission and deposition. Phil. Trans. Roy. Soc. Lond. B Biol. Sci. 368 (1621), 20130166. https://doi.org/10.1098/rstb.2013.0166.

Van Damme, M., Clarisse, L., Dammers, E., Liu, X., Nowak, J.B., Clerbaux, C., Flechard, C.R., Galy-Lacaux, C., Xu, W., Neuman, J.A., Tang, Y.S., Sutton, M.A., Erisman, J.W., Coheur, P.F., 2015. Towards validation of ammonia (NH3) measurements from the IASI satellite. Atmospheric Measurement Techniques 8, 1575–1591.

Van Damme, M., Wichink Kruit, R.J., Schaap, M., Clarisse, L., Clerbaux, C., Coheur, P.-F., Dammers, E., Dolman, A.J., Erisman, J.W., 2014. Evaluating 4 years of atmospheric ammonia (NH3) over Europe using IASI satellite observations and LOTOS-EUROS model results. J. Geophys. Res.: Atmosphere 119, 9549–9566. https://doi.org/10.1002/2014JD021911.

Vetterli, A., Hietanen, S., Leskinen, E., 2016. Spatial and temporal dynamics of ammonia oxidizers in the sediments. Mar. Environ. Res. 113, 153–163. https://doi.org/10.1016/j.marenvres.2015.12.008.

Zambrano-Monserrate, M.A., Ruano, M.A., Sanchez-Alcade, L., 2020. Indirect effects of COVID-19 on the environment. Sci. Total Environ. 728, 138813. https://doi.org/10.1016/j.scitotenv.2020.138813.

Zheng, G., Su, H., Wang, S., Andreae, M.O., Pöschl, U., Cheng, Y., 2020. Multiphase buffer theory explains contrasts in atmospheric aerosol acidity. Science 369, 1374–1377.

Zilio, M., Orzi, V., Chiodini, M.E., Riva, C., Acutis, M., Boccasile, G., Adani, F., 2020. Evaluation of ammonia and odour emissions from animal slurry and digestate storage in the Po Valley (Italy). Waste Manag. 103, 296–304. https://doi.org/10.1016/j.wasman.2019.12.038.

3.2. Life cycle assessment of air treatment techniques to reduce NH_3, GHG, VOCs and particulate matter emissions from pig housing

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Abstract

This study focuses on the work carried out on the quantification of the environmental footprint of pig production, with a specific focus on housing emissions. Life Cycle Assessment (LCA) was used to quantify the environmental impact of pig production under “standard” production context and compare it with production when implementing two technologies for air treatment aimed at abating pollutant emissions, i) Wet scrubber; ii) dry scrubber. The study focuses on one transition farm in Spain and two fattening farms in Italy. Inventories were built regarding farms’ technical performances and productive and environmental parameters, and regarding the two technologies’ operation and emissions abatement efficiency. This allowed subsequently to (i) outline the environmental footprint of pig production in the different scenarios by means of an established characterization method; (ii) highlight environmental hotspots; (iii) compare a baseline scenario with two alternative ones for each farm. Both tested technologies showed their potential to reduce emissions in the pig housing stage, which influenced all those categories affected by air pollutant emissions, such as particulate matter formation, acidification and eutrophication. At the same time, various trade-offs have been observed between the categories that are

affected by the emission abatement and those that are instead more linked to energy and resource use. In fact, both scrubbers need consumables for their operation, and these involve an additional impact on the system compared to the base scenario. When considering the balance between emissions avoided and trade-offs generated, the dry scrubber was found to be the best solution. The results in Spain and in the two farms in Italy showed similar environmental trends in the different scenarios, albeit with slightly variable results in absolute terms. Scrubbers had a greater influence (both positively in mitigation and negatively in trade-offs) in farms in Italy, probably due to their use during phases with longer duration in these farms rather than in the Spanish one, which involved only one phase of the pig lifecycle. A final sensitivity analysis on the characterization factors also made it possible to investigate aspects related to the methodology with respect to some impact categories, including acidification, concluding that the impact of scrubbers is also variable based on the geographical location in which they are implemented. In conclusion, scrubbers are both environmentally interesting technologies and can bring benefits especially in areas where eutrophication and particulate matter formation are locally relevant issues. At the same time, these alone do not solve the problem of the environmental impact of pig farming, which requires various interventions at different levels of the supply chain. Furthermore, there is ample room for improvement in scrubbers' efficiency to be able to achieve greater emissions reductions on the one hand and to optimize the use of consumables on the other, which would be crucial, especially for the wet scrubber, to limit trade-offs.

Introduction: the Life Cycle Assessment approach in the Life-MEGA project

The general objective of the project Life-MEGA was to provide scientifically-informed decision support to help actors in the pig production sector to decide for technologies that can potentially help to reduce emissions in animal housing. In general terms, a potential emission reduction by incorporating these technologies to the farm could help to improve sustainability of the pig production sector. However, the use of added technologies to the standard farm needs extra infrastructure and energy consumption which can, for example, increase consumption of natural resources. Therefore, a holistic scientific approach is needed to assess the environmental performance of these technologies from a global perspective. Life Cycle Assessment (LCA) has become increasingly employed in recent years in the agricultural sector since it provides a useful and valuable tool for agricultural systems environmental evaluations and comparisons. LCA has been widely used to assess environmental impact from pig production globally (Dourmad et al., 2014; Poore & Nemecek, 2018). This tool was selected due to its standardised quantitative approach to estimating environmental impacts as well as its holistic vision, including multi criteria environmental indicators. LCA is an internationally recognized methodology, regulated by ISO standards (ISO 14040 2006; ISO 14044 2018), that aims to analyse products, processes, or activities from an environmental perspective throughout their entire life cycle, or even part of it. This methodology considers all the inputs (resources and energy consumed), and outputs (emissions and wastes) generated. LCA as a multicriteria tool which can include more than 20 environmental indicators. In summary, a LCA includes all aspects that could potentially affect human health, ecosystem quality and depletion of resources.

In particular, this deliverable corresponds to Action C.3 on “Life cycle assessment of techniques to reduce NH3, GHG, VOCs and particulate matter”. The baseline scenario corresponds to a representative transition pig farm in Spain and two fattening pig farms in Italy. The technologies assessed comprise a wet and a dry scrubber installed in the housing of the pig farms. An environmental assessment of the baseline in comparison to the use of the technologies (“alternative scenario”) evaluated in this project was carried out to test their effectiveness to reduce emissions in the pig housing as well as their overall environmental performance. As previously mentioned, this environmental assessment was conducted using the LCA approach. In this study, the 16 indicators recommended by the European Commission (CE) through the Product Environmental Footprint (Zampori & Pant, 2019) initiative, were used to quantify potential impact to climate change, acidification, and eutrophication, among others.

The scenarios evaluated were:

a) Baseline scenario, where no technology was used to reduce emissions in the housing;

b) Alternative scenario - Technology 1: Wet scrubber;

c) Alternative scenario - Technology 2: Dry scrubber.

These scenarios (baseline and technologies 1 and 2) were assessed in three pig farms:

i) Transition farm, Catalonia, Spain

ii) Fattening farm A, Lombardy, Italy

iii) Fattening farm B, Lombardy, Italy

These correspond to the farms where the project's experimental field tests took place. The objective of this selection was to test the technologies in question in contexts where, to the authors' knowledge, they had never yet been applied, despite being areas with a strong livestock production vocation as well as the particularly acute environmental problem of ammonia emissions. Specifically, the two regions where these farms are located are those that host the largest number of pigs in their respective countries. The selected farms are representative of the respective context in technical-productive and economic terms.

Hereafter the report is divided in the following sections:

- The "methodology" section provides a detailed description on the study scope and methodological choices and models used to perform the environmental analysis as well as the limitations of the study. It describes the studied system and the type and source of the data used (inventory). This is followed by a description of the assessed scenarios (wet and dry scrubber) and a description of the sensitivity analysis performed to test the robustness of the results.
- Section on "results" describes environmental outcomes obtained for both the baseline and the alternative scenarios on each of the farms assessed, as well as for the sensitivity analysis performed.
- Section on "discussion and conclusions" includes the relevant aspects and challenges encountered in this task (discussion) as well as a summary of the remarkable findings and the aspects requiring further research (conclusions).

Methodology

The analysis was carried out following the guidelines on the *Environmental Performance of Pig Supply Chains* published by the Food and Agriculture Organization of the United Nations (FAO, 2016). For each one of the scenarios the environmental assessment was conducted in accordance with the four phases by the reference ISO standards (ISO 14040, 2006): Goal and Scope, Life Cycle Inventory, Life Cycle Impact Assessment, and Interpretation.

The goal of this LCA was to quantify the environmental footprint of a pig production system, with a focus on technologies to reduce housing emissions. We conducted an attributional LCA, which means performing an assessment of the current situation without considering consequences in other systems (what would be considered a consequential LCA).

The intended application of the results was to assess the overall environmental impact of the assessed technologies applied to reduce emissions in the pig housing.

The analysis was carried out with a cradle-to-farm gate approach, the functional unit adopted was 1 kg of live weight produced, in accordance with the FAO guidelines (FAO, 2016). The system boundaries were *cradle-to-farm gate*. The system boundaries are shown schematically in Figure 1. In line with the system boundaries adopted, downstream processes were not included in the LCA. As the focus of this study was in the pig housing inside the farm, downstream processes would not change between scenarios. The potential reduction of veterinary treatments possibly given by a better air quality in animal housing was excluded by the assessment due to the lack of information, in particular of the characterization factors (CF).

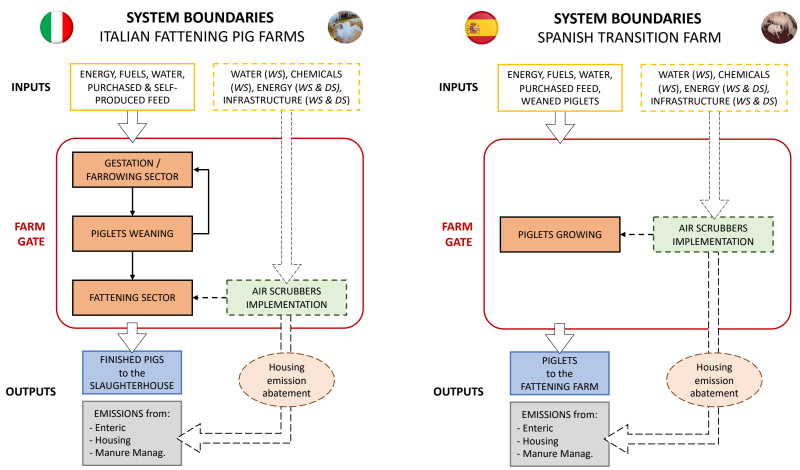

Figure 1. Diagram of the system boundaries for fattening farms in Italy and for the transition piglet farm in Spain. Outputs at the farm gate are represented by the live weight

of animals sold (Functional unit) and emissions from enteric fermentations, animal housing and manure management. The dotted lines represent the inputs and processes present only in the alternative scenarios, i.e. those which involve the on-farm implementation of air scrubbing technologies. These in fact involve additional inputs consumption, while having at the same time an effect on the system's emissions, acting on the reduction of housing emissions.

At this stage, no impact allocation procedure was carried out because the only outputs having an economic value of the system are represented by the animals leaving the farm for fattening (in Spain) or slaughter (in Italy).

The target audience of the study is represented by Catalonian/Spanish and Italian pig producers advisors, other workers or entities involved in the supply chain (e.g., feed producers, ham production and distribution brands), and policy makers, as well as the general scientific community.

Inventory analysis

As for Italy, the analysed farms are in Lombardy, Northern Italy. These are two intensive closed cycle (or farrowing-to-finishing) farms, meaning that produce piglets and raise them up to market weight. Specifically, heavy pigs for PDO dry-cured ham consortia are produced. Mixed-livestock farms are widespread in northern Italy, which means that it is common for these to have some arable land available, where energy-intensive crops are usually grown (most commonly maize). Consequently, the animals are fed partially with self-produced crops, complemented by purchased commercial feeds and supplements.

The animals are housed in an indoor system, with different specific conditions depending on their life stage. During lactation, sows are kept in farrowing crates where they are confined between bars to reduce the risk of the sow crushing her newborn piglets. After 3 weeks piglets are weaned and placed in a nursery, whereas the sow is returned to the gestation barn. Here, all females are artificially inseminated and remain housed in the gestating housing section for the gestation period. After the piglets reach approximately 25-35 kg, they are placed in a growing-finishing barn where they remain until they reach 160 kg, which corresponds to a period of approximately 9 months (minimum live weight and age required by PDO regulation). Boars are used to collect semen for artificial insemination.

The reared pig categories are housed in pens, more specifically farrowing pigs in closed mechanically ventilated buildings, whereas fattening pigs in closed naturally ventilated buildings. Electricity is consumed both in farrowing and fattening sections for illumination, feeding processes, and manure management, handled as slurry. Also, the feeding process requires diesel fuel consumption for grinding, mixing, and distribution operations.

As for Spain, the analysed farm is in Santa Eulalia de Riuprimer, Catalonia. The farm in Spain includes the transition stage, thus pigs from post-weaning up to 30-40 days of life, when they move on to a new stage (a fattening farm). The assessed farm is of intensive conventional management. Animals are housed in an indoor system. Pigs are fed with commercially purchased compound feed. There is an empty cycle in between pig batches. The duration of the "empty cycle" indicates how long is the period during which the installation remains empty (of animals) to carry out an appropriate cleaning and sterilisation of the facilities. The duration of this empty cycle has an impact on the total number of pigs that the farm produces per year. This is a common practice in Catalonian pig farms, and normally this "empty cycle" can vary from 2 to 3 weeks. For this study we consider 2 weeks. Regarding heating, in Catalonia (temperate Mediterranean climates), for transition farms some heating is used in winter. In this case, the thermal energy to heat the buildings is produced by a diesel boiler. Regarding cooling, it is not essential, as there is good air circulation.

The final inventory for the analysis consisted of collected data referring to the farm productive performance (stocking rate, production rate, etc.), to the consumption of the different resources (e.g., feed, water, energy use, cleaning products, etc.) as well as the waste (plastic, water, etc.) and emissions produced (enteric fermentation, manure management). Regarding the dry scrubber, the following information was collected: filter material, working time, energy consumption. The same applies to the wet scrubber scenario where water and citric acid consumption, ammonia abatement (and consequently nitrogen recovered in the solution) were also collected.

The farm's infrastructure (stables) have been excluded from the system boundaries. Their impact on the pork supply chain has in fact been assumed negligible due to their long life span (tens of years), as widely reported in the literature for livestock. On the other hand, the infrastructure of the technologies used has been included. In fact, these have much shorter life spans (about 10 years) and therefore it cannot be taken for granted that the contribution to the impacts is low, together with the fact that there is little knowledge on the influence of air treatment technologies on livestock impacts, especially pork.

All inventory data (both inputs and outputs) were related to a time horizon of one year, as is usually the case in LCA studies in agri-food chains. More details are presented in the following paragraphs.

Primary data

Primary data regarding farming activities were collected by means of questionnaires provided to farmers regarding inputs and outputs of production processes. Particular attention was given to: average annual pig population and mortality, divided into different sub-categories (e.g. piglets, lactating sows, gestating sows, fattening pigs); annual purchase (piglets for transition in Spain) and sales of heads (piglets for fattening in Spain and pigs for slaughter in Italy) and relative average weight; composition and consumption of feeds; possible internal production of feed components; housing facilities and manure management (necessary for the subsequent estimation of GHG, NH_3 and other pollutant emissions); energy and water consumption. Researchers of the University of Milan collected data on the farms involved in testing the tested technologies in Italy, while the IRTA researchers took care of the inventory of the farm involved in Catalonia, Spain.

The primary technical-productive data collected relating to the different farms according to the production cycles described above are shown in Table1 for Italy and Table 2 for Spain.

Table 1. Main technical-productive primary data collected for the Italian assessed pig farms.

		Farm A	**Farm B**
Average animal population by category			
Piglets present	*heads*	3000	1370
Fatteners present (31-50 kg)	*heads*	3000	-
Fatteners present (30-80 kg)	*heads*	-	2460
Fatteners present (80-120 kg)	*heads*	-	1780
Fatteners present (51-160 kg)	*heads*	2793	-
Fatteners present (120-160 kg)	*heads*	-	1900

Sows (160-200 kg) lactating		*heads*	215	250
Sows (160-200 kg) nursery & dry period		*heads*	515	450
Replacement male & female		*heads*	220	110
Average exit live weight/fattener		*kg head^{-1}*	160	160
Inputs				
Water	Tap	*m^3 year^{-1}*	82024	80236
Energy	Electricity	*kWh year^{-1}*	689868	312000
	Diesel	*l year^{-1}*	86160	70122
	Animal purchase	*tkm year^{-1}*	n/a	n/a
Feed	Compound feed	*tonnes year^{-1}*	5069	6598
Outputs				
Animals (fatteners and spent sows to the slaughterhouse)		t of LW sold *year^{-1}*	1775	1736

Table 2. Main technical-productive primary data collected for the Spanish assessed piglet transition farm.

Average animal population by category			
Average animal heads present		*heads*	7573
Animal spots per year		*heads*	9690
Average entry live weight/head		*kg head^{-1}*	5
Average exit live weight/head		*kg head^{-1}*	15
Inputs			
Water	Tap	*m^3 year^{-1}*	3779
Energy	Electricity	*kWh year^{-1}*	91958
	Diesel	*l year^{-1}*	29942
	Animal purchase	*tkm year^{-1}*	9411
Feed	Compound feed	*tonnes year^{-1}*	1118
Transport	Animal purchase	*tkm year^{-1}*	20011
	Manure	*tkm year^{-1}*	1322
	Waste	*tkm year^{-1}*	884
Outputs			
Animals (piglets to the fattening phase)		units *year^{-1}*	61634

Secondary data: Baseline Emissions

Primary data were supplemented with secondary data regarding air pollutant emissions, which were estimated using different established models available in the literature. In detail, methane emissions due to enteric fermentation and methane and dinitrogen monoxide emissions due to manure management were considered following the IPCC guidelines (IPCC 2019). Since animal feeds did not change across scenarios, we applied TIER I emission factors. Regarding ammonia emissions at the manure management stage, Tier II was used according to EEA guidelines.

Table 3 presents an overview of the emissions calculated and the methods used.

Table 3. Methods and Tier levels used for the determination of emissions in the baseline scenario, divided by gas and origin of emissions.

Emission type	Substance	Units	Method	Tier
Enteric Fermentation	CH_4	METHANE (kg CH_4)	IPCC 2019	Tier I
Manure management	CH_4	METHANE (kg CH_4)	IPCC 2019	Tier I
	N_2O	NITROUS OXIDE (kg N_2O) (Direct + Indirect (NH_3+NO_x))	IPCC 2019	Tier I
	NH_3	AMMONIA (kg NH_3)	EEA 2019	Tier II
	NO	NITRIC OXIDE (kg NO)	EEA 2019	Tier II
	NMVOC	Non-methane volatile organic compounds (kg NMVOC)	EEA 2019	Tier I
	PM2,5	Particulate (kg $PM_{2,5}$)	EEA 2019	Tier I
	PM10	Particulate (kg PM_{10})	EEA 2019	Tier I

Enteric methane emissions were calculated according to the IPCC 2019 Tier I based on emission factors according to type and number of animals. For the Spanish farm, these emission factors were adapted to national specific conditions for transition pigs as the emission factor by IPCC was only valid for fattening pigs.

Emissions of ammonia, methane, and nitrous oxide from manure management, including storage, were calculated according to EEA 2019 and IPCC 2019. In more detail:

- Methane emissions to the air compartment were quantified following the methodology suggested by Tier I from IPCC 2019. The main factors affecting methane emissions are the amount of manure produced and the portion of the manure that decomposes anaerobically. The former depends on the rate of waste production per animal, and the latter on how the manure is managed. When manure is stored or treated as a liquid as it occurs in this case (e.g., ponds, pits), it decomposes anaerobically and can produce a significant quantity of methane. The temperature and the retention time of the storage unit greatly affect the amount of methane produced. Default values were used depending on type of animal, manure management and climate conditions. We extracted the values for the potential IPCC climate zones of both countries, Italy and Spain.
- For emissions of nitrous oxide to the air compartment methodology suggested by IPCC 2019 Tier I was followed. Direct N_2O emissions occur via combined nitrification and denitrification of nitrogen contained in the manure. The emission of N_2O from manure during storage and treatment depends on the nitrogen and carbon content of manure, and on the duration of the storage and type of treatment. Nitrification (the oxidation of ammonia nitrogen to nitrate nitrogen) is a necessary prerequisite for the emission of N_2O from stored animal manures. Nitrification is likely to occur in stored animal manures provided there is a sufficient supply of oxygen. Indirect emissions result from volatile nitrogen losses that occur primarily in the forms of ammonia and NOx.
- For the calculation of ammonia emissions to the air compartment EMEP / EEA Tier II (European Monitoring and Evaluation Programme / European Environment Agency) was followed, which requires the number of animals, total nitrogen excretion rates (calculated according to IPCC guidelines); proportion of nitrogen excreted in buildings; proportion of nitrogen excreted as total ammoniacal nitrogen (TAN) and proportion of excretion site; amount of manure handled as liquid or solid manure; use of animal bedding; slurry storage system.

Background data

Background data regarding raw materials and some feed ingredients were retrieved from the established Ecoinvent database v3.8, Cut-Off system model (Wernet et al., 2016). Where available, datasets with specific geographic representativeness have been used (e.g., national electricity mixes for Italy and Spain), or otherwise European ("Europe without Switzerland" dataset) or world ("GLO" datasets) average ones. In some cases, especially for feed ingredients, average datasets have been modified considering local conditions to better represent the reference Italian and Spanish production context.

Alternative scenario modelling

As previously mentioned, the baseline scenario (i.e., without air treatment technologies) was compared with two alternative scenarios. In each of these the implementation of a different air treatment technology for pig housing in the production cycle is considered. The following paragraphs describe the methodological setting of the alternative scenarios analyzed, detailing the relative inventory data.

Wet scrubber

The wet scrubber treats the polluted indoor air, sucked in thanks to a vacuum generated by a blower, and recirculates purified air inside the barns. For its operation, it consumes a citric acid solution and energy for the blower. In this scenario data related to scrubber energy, water, and acid consumption as well as raw materials and energy needed for the construction of the machinery were included in the system boundaries.

A single wet scrubber prototype unit weighs 2000 kg, and it's made entirely of stainless steel (personal communication with Rota Guido staff). The scrubber is also equipped with 30 m of corrugated polyethylene pipe as ventilation duct for air inlet and outlet. A depreciation rate of 10 years (based on De Vries & Melse, 2017) was considered to model infrastructure inventory by year, as shown in Table 4.

Table 4. Wet scrubber infrastructure inventory.

Material	Amount/Unit	Life Span	Amount /Unit/Year
Chromium steel pipe	2000 kg	10 years	200 kg

Polyethylene pipe	30 m	10 years	3 m

The same amount/unit/year has been inventoried as waste (scrap steel and waste polyethylene). For polyethylene pipes it was considered the conversion 1 m (corrugated polyethylene pipe - DN75) = 0.347 kg (Source: Ecoinvent).

Consumables for scrubber operation, namely citric acid, water and electricity, have been modelled in order to express them in relation to 1 kg of ammonia removed by the machinery.

Water and citric acid consumption values used as inventory data are reported in Table 5, which correspond to the medians of the measurements made during the field trials foreseen within the Life-MEGA project.

Table 5. Wet scrubber consumables inventory.

Item	**Water**	**Citric acid**
Unit	dm^3/kg of removed NH_3	kg/kg of removed NH_3
Median	279.42	13.81
*N° obs**	20	17

**in this context, by observation it is meant a period in which the difference in filling of water and citric acid in the scrubber was measured, which was on average one week, although variable between 3 and 13 days. Observations include data from the different farms where the scrubber was tested.*

As regards electricity consumption, the average hourly consumption of 0.48 kWh per scrubber unit was used, measured during field trials thanks to an energy meter, and considered in the alternative scenario assuming 100% annual operation of the machinery.

In the trials carried out on farms during the project the scrubbers were used in some rooms of the farms, not in the whole farm. This means that the treatment of animal housing emissions during the trials was only partial. However, in the present Life Cycle Assessment, we wanted to test the influence of scrubbers in a hypothetical full-scale farm implementation. Therefore, a scaling calculation was necessary. The ammonia abatement capacity was considered as a basis for scaling the use of the scrubber to the entire farm, since this is the main pollutant targeted by this technology. The variables considered to do so were:

- the ventilation capacity → 6700 m3/h (personal communication with Rota Guido staff)
- a reference ammonia concentration inside pig barns → 10 mg/m3
- ammonia emission factor from housing, per pig place → 2.5 kg/year (reference value for fattening pig farms in Europe, source EEA)

It follows that:

$$\boldsymbol{Total\ inlet\ ammonia} = 10\left(\frac{mg_{NH3}}{m^3}\right) \times 6700\left(\frac{m^3}{h}\right) \times 8760\left(\frac{h}{year}\right) = 586.9\left(\frac{kg_{NH3}}{year}\right)$$

Therefore, assuming maximum operation (100% of the year) and the reference air concentrations, regardless of the abatement efficiency achieved, a scrubber unit as the one tested in the project is able to treat 586.9 kg of NH_3 per year.

The number of heads for which a scrubber of this size would be suitable is then estimated by the relationship between the total inlet ammonia and the ammonia emission factor from housing, per pig place:

$$\boldsymbol{Sizing} = \frac{586.9\left(\frac{kg_{NH3}}{scrubber\ unit\ *\ year}\right)}{2.5\left(\frac{kg_{NH3}}{pig\ place\ *\ year}\right)} = 234.76 \cong 230\left(\frac{pig\ place}{scrubber\ unit}\right)$$

Consequently, for Italy, the installation of a number of scrubber units equal to the number of fattening pig places divided by 230 for the two farms was assumed.

Table 6 shows the abatement efficiencies considered for the wet scrubber for the various air pollutants, resulting from the field trials of this project. Emission reduction was estimated based on punctual measures. Those depend on several factors (such as temperature, ventilation, measuring point, etc) that can vary across days or even along the same day. Therefore, there is a large uncertainty attached to the emission reduction values obtained. This was addressed by assessing the maximum potential reduction as the reduction attainable under the best operating conditions for the technology, as well as using the median reduction as a way to reduce noise in the data and better reflect the reductions generally obtained during the field trials carried out.

Table 6. Pollutants abatement efficiency obtained during the field trials in Italy and Spain regarding the *wet scrubber*, used to model the alternative scenario. Both the maximum abatement obtained, and the median are reported.

Item	**Spain**		**Italy**	
Air pollutant	Max	Median	Max	Median
Ammonia	-79%	-42%	-58.8%	-22.1%
PM10	-100%	-100%	-27%	ineffective

Note: Abatement of direct emissions of nitrous oxide has not been measured. However, a reduction of indirect emissions was considered as a rebound effect of the reduction of ammonia emissions. However, a fixed factor was not applied for this as it depended on the total emissions of each farm.

Dry scrubber

The dry scrubber is a fan inside a box mounted inside the pig housing facilities which blows air towards polyester fiber panels (about half a square meter of the total surface per scrubber unit). The flow rate can vary between 3000-6000 m^3/h. The same lifetime of the wet scrubber (10 years) was considered for the analysis and the same principle of sizing and scaling, these having been used in field trials on rooms of similar size.

As for the infrastructure, the Ecoinvent process "*Blower and heat exchange unit, central, 600-1200 m3/h {GLO}| market for | Cut-off, U*" was used as a proxy. Considering that the flow of dry scrubber is higher than the one of dataset (3000-6000 m^3/h vs 600-1200 m^3/h), it has been multiplied by 4.5 to scale it, and than divided by 10 to reflect its annual virtual consumption.

Considering that the polyester fiber has a very low weight and that, according to the manufacturer, the panels, under correct periodic maintenance, can also be replaced every 3-5 years, the filtering material has been considered negligible for the purposes of the life cycle analysis and excluded.

Consequently, the only consumable included for the wet scrubber scenario was electricity, whose average hourly consumption, according to the manufacturer, was considered 0.55 kWh.

Table 7 shows the abatement efficiencies considered for the dry scrubber for the various air pollutants, resulting from the field trials of this project. As for the wet scrubber, both the maximum abatement potential observed and the median one were reported and used for the current environmental assessment.

Table 7. Pollutants abatement efficiency obtained during the field trials in Italy and Spain regarding the *dry scrubber*, used to model the alternative scenario. Both the maximum abatement obtained, and the median are reported.

Item	**Spain**		**Italy**	
Air pollutant	Max	Median	Max	Median
Ammonia	-48%	-35%	-62%	-19%
PM10	-100%	-100%	-45%	-8.6%

Note: Abatement of direct emissions of nitrous oxide has not been measured . However, a reduction of indirect emissions was considered as a rebound effect of the reduction of ammonia emissions. However, a fixed factor was not applied for this as it depended on the total emissions of each farm.

Life cycle impact assessment

Life Cycle Impact Assessment (LCIA) is defined as the phase in the LCA aimed at assessing the magnitude of the potential environmental impact of a production system, in this case production of pigs and piglets at farm gate. In LCIA, impact models are used to calculate characterisation factors that connect elementary flows (resource consumptions, emissions) to the corresponding environmental impacts in different indicators (impact categories).

All the 16 indicators recommended by the European Commission (CE) through the Product Environmental Footprint (Zampori & Pant, 2019) initiative, were used. LCIA impact indicator results are calculated for the recommended impact categories according to the EF 3.0 Method (adapted) V1.00 (Fazio et al., 2018), which were derived from the International Life Cycle Data system, International Reference Life Cycle Data System (ILCD) scheme (E.U.-J.R.C., 2011).

The LCA Software tool SimaPro (PRéConsultants, 2020) was used.

RESULTS: BASELINE SCENARIO

The different impacts were grouped into processes to ease the result discussion, below a summary of results is given together with a clarification of what impact is included in each of these grouped processes.

For both Italy and Spain, the results of the baseline scenario as well as those of the alternative ones are shown in tables which show the absolute values deriving from each single item/contributor (inputs or outputs) to facilitate the contribution analysis interpretation. In the same tables, the hotspots are also highlighted graphically thanks to the use of colored bars which, for each impact category, indicate how large the cell values are compared to the other values on the same line based on the length of the bars. In practice, a longer bar represents a larger impact, and a shorter bar represents a relatively smaller impact, for a given item on each impact category.

As for Italy, the two fattening farms analyzed show different results in absolute terms but aligned in relative terms. In fact, the impact per kg of live weight is somewhat variable between the two mainly due to the different feeds used, and secondly to the different management of the breeding phases and growing performances. On the other hand, the contribution analysis clearly shows what the impact hotspots are for the different categories, and these maintain consistency between the two (Figures 2 and 3). Feed consumption and supply plays an important role in all categories and is the main hotspot of the process. This is in line with the scientific literature on livestock production. For climate change, particulate matter formation and terrestrial eutrophication the main contribution is instead given by pollutant emissions on farm.

Impact categories	Units	Farm emissions (enteric and manure management)	Feed	Diesel	Electricity, gas & other	Water
Climate change	kg CO_2 eq	3.25E+00	2.48E+00	2.29E-01	6.92E-02	1.53E-02
Ozone depletion	kg CFC11 eq	0.00E+00	1.02E-07	3.75E-08	9.35E-09	1.02E-09
Ionising radiation	kBq U-235 eq	0.00E+00	5.51E-02	1.34E-02	9.39E-03	5.45E-03
Photochemical ozone formation	kg NMVOC eq	2.98E-03	5.83E-03	2.20E-03	1.55E-04	5.08E-05
Particulate matter	disease inc.	7.21E-07	1.36E-07	9.44E-09	1.38E-09	8.19E-10
Human toxicity, non-cancer	CTUh	4.89E-09	4.53E-08	7.72E-09	6.94E-10	9.03E-10
Human toxicity, cancer	CTUh	0.00E+00	1.54E-09	2.04E-10	2.56E-11	6.41E-11
Acidification	mol H+ eq	3.93E-03	1.81E-02	1.94E-03	3.35E-04	8.50E-05
Eutrophication, freshwater	kg P eq	0.00E+00	6.03E-04	3.39E-05	1.76E-05	1.10E-05
Eutrophication, marine	kg N eq	3.02E-03	1.87E-02	7.24E-04	4.94E-05	1.64E-05
Eutrophication, terrestrial	mol N eq	2.74E-01	7.31E-02	7.90E-03	5.51E-04	1.56E-04
Ecotoxicity, freshwater	CTUe	4.44E+00	6.37E+01	3.65E+00	8.46E-01	2.77E-01
Land use	Pt	0.00E+00	2.06E+02	3.60E+00	3.49E-01	6.21E-02
Water use	m^3 depriv.	0.00E+00	1.65E+01	1.84E-02	4.65E-02	1.99E+00
Resource use, fossils	MJ	0.00E+00	9.93E+00	2.92E+00	1.04E+00	2.62E-01
Resource use, minerals and metals	kg Sb eq	0.00E+00	9.77E-06	2.43E-06	6.71E-07	7.55E-08

Figure 2. Environmental results and hotspots for the baseline scenario of the farm A in Italy.

Impact categories	Units	Farm emissions (enteric and manure management)	Feed	Diesel	Electricity, gas & other	Water
Climate change	kg CO_2 eq	2.57E+00	2.07E+00	1.48E-01	1.50E-01	1.53E-02
Ozone depletion	kg CFC11 eq	0.00E+00	1.15E-07	3.37E-08	2.02E-08	1.02E-09
Ionising radiation	kBq U-235 eq	0.00E+00	6.44E-02	9.12E-03	2.03E-02	5.45E-03
Photochemical ozone formation	kg NMVOC eq	2.40E-03	5.67E-03	1.83E-03	3.36E-04	5.08E-05
Particulate matter	disease inc.	5.65E-07	1.96E-07	2.32E-09	2.99E-09	8.19E-10
Human toxicity, non-cancer	CTUh	3.87E-09	3.02E-08	1.24E-09	1.50E-09	9.03E-10
Human toxicity, cancer	CTUh	0.00E+00	1.17E-09	1.36E-11	5.54E-11	6.41E-11
Acidification	mol H+ eq	3.03E-03	2.80E-02	1.48E-03	7.24E-04	8.50E-05
Eutrophication, freshwater	kg P eq	0.00E+00	1.69E-03	1.57E-06	3.80E-05	1.10E-05
Eutrophication, marine	kg N eq	2.33E-03	1.07E-02	6.50E-04	1.07E-04	1.64E-05
Eutrophication, terrestrial	mol N eq	2.11E-01	1.19E-01	7.13E-03	1.19E-03	1.56E-04
Ecotoxicity, freshwater	CTUe	3.43E+00	7.08E+01	1.08E+00	1.83E+00	2.77E-01
Land use	Pt	0.00E+00	1.94E+02	2.59E-01	7.54E-01	6.21E-02
Water use	m^3 depriv.	0.00E+00	4.25E+01	3.18E-04	1.01E-01	1.99E+00
Resource use, fossils	MJ	0.00E+00	9.16E+00	2.03E+00	2.25E+00	2.62E-01
Resource use, minerals and metals	kg Sb eq	0.00E+00	7.43E-06	2.65E-08	1.45E-06	7.55E-08

Figure 3. Environmental results and hotspots for the baseline scenario of the farm B in Italy.

Main process contribution in the Spanish farm (Figure 4) comes from three processes: purchasing of weaned piglets, emissions at farm and compound feed. Main differences between the Italian and the Spanish farm comes from the difference in the farm stages considered (transition for Spain, fattening for Italy). On one hand, the purchase of weaned piglets has a large weight in the Spanish farm as it includes only the transition farm. On the other hand, the consumption of diesel means a large contribution to the total impact, compared to if the whole fattening system would have been assessed (as grower and finisher pigs often do not use heating in Spain).

Impact from weaned piglets, which were purchased at farms in the region, included all the necessary processes for the rearing of the animals up to the weaning stage. Therefore, weaned piglets impact carried the weight from the compound feed impact (thus, crops used on the feed for the sows), farm emissions from enteric fermentation and manure storage, energy and water consumption and infrastructure. The impact from the weaned piglets contributed with over a 5% on all impact categories. This was the case for climate change (50% of impact for this category), Ozone depletion (43%), Ionising radiation (72%), Photochemical ozone formation (38%), Particulate matter (66%), Human toxicity no cancer (36%), Human toxicity cancer (47%), Acidification (80%), Eutrophication freshwater (43%), Eutrophication, marine (52%), Eutrophication, terrestrial (81%), Ecotoxicity freshwater (38%), Land use (55%), Water use (13%), Resource use, fossils (50%) and Resource use, minerals and metals (8%). If the fattening phase would be included, contribution of this process to the overall impact would be significantly reduced.

Emissions associated with the pig farm included emissions from enteric fermentation and manure management (housing and storage) at the transition farm. These emissions had a significant contribution on the impact for the categories CC (10%), PM (in particular, particulates between 2.5 µm, and 10µm, followed in a considerably minor amount by the emissions of ammonia and, lastly, from particles smaller than 2.5µm; 16%), Eutrophication, terrestrial (4%). The impact from emissions for the rest of impact categories where this process was contributing was 1%. Emissions during the weaned piglets' production are excluded here, as they are included in the impact from producing the piglets at the sow farm. Compound feed included production from the different raw materials in the countries of origin, processing, and transport to Catalonia. The feed from the production of weaned piglets is excluded from this process as it is included in the impact from producing the weaned piglets at the sow farm. Compound feed was the major contributor in most impact categories. This was the case for climate change (31% of impact for this category), Ozone depletion (39%), Ionising radiation (24%), Photochemical ozone formation (41%), Particulate matter (15%), Human toxicity no cancer (52%), Human toxicity cancer (42%), Acidification (16%), Eutrophication freshwater (43%), Eutrophication marine (43%), Eutrophication terrestrial (13%), Ecotoxicity freshwater (55%), Land use (44%), Water use (84%), Resource use, fossils (38%) and Resource use, minerals and metals (69%). From the compounds of the feed major contributors were wheat, maize, barley, and soybean due to those being the components present in greater amounts. Regarding the diesel contribution, it has importance to climate change (4%), ozone depletion (10%), photochemical ozone formation (15%), categories related to human toxicity (5%) and use of resources (8% for fossils, 21% for mineral and metals). At last, in relation to the incineration of carcasses, its contribution is over 5% of the global impact to the following impact categories: climate change and ozone depletion (5 and 6% respectively), categories related to human toxicity (5%), and eutrophication freshwater (9%).

Impact categories	Units	Farm (enteric fermentation+manure management emissions)	Farm 0, weaned piglets	Compound feed	Diesel	Electricity, gas &oil	Water	Transport	Carcasses Incinerated
Climate change	kg CO_2 eq	4,0E+05	2,0E+06	1,2E+06	1,7E+05	2,1E+03	1,1E+03	1,5E+04	2,1E+05
Ozone depletion	kg CFC11 eq	0,0E+00	1,2E-01	1,0E-01	2,7E-02	6,3E-04	2,2E-04	3,0E-03	1,6E-02
Ionising radiation	kBq U-235 eq	0,0E+00	4,0E+05	1,3E+05	9,8E+03	6,1E+03	1,6E+03	1,3E+03	4,4E+03
Photochemical ozone formation	kg NMVOC eq	2,4E+02	4,1E+03	4,5E+03	1,6E+03	3,2E+00	2,9E+00	8,6E+01	3,8E+02
Particulate matter	disease inc.	7,7E-02	3,1E-01	6,9E-02	6,3E-03	1,0E-04	6,6E-05	1,4E-03	4,7E-03
Human toxicity, non-cancer	CTUh	5,8E-04	3,6E-02	5,3E-02	5,7E-03	3,4E-05	2,2E-05	2,9E-04	5,2E-03
Human toxicity, cancer	CTUh	0,0E+00	1,3E-03	1,2E-03	1,3E-04	1,1E-06	1,1E-06	2,0E-05	1,4E-04
Acidification	mol H+ eq	2,0E+02	4,5E+04	9,0E+03	1,4E+03	1,4E+01	8,4E+00	8,0E+01	5,2E+02
Eutrophication, freshwater	kg P eq	0,0E+00	3,3E+02	3,3E+02	2,8E+01	3,4E-01	4,3E-01	2,5E+00	7,2E+01
Eutrophication, marine	kg N eq	2,5E+02	1,1E+04	8,9E+03	5,2E+02	1,4E+00	1,1E+00	2,4E+01	8,4E+01
Eutrophication, terrestrial	mol N eq	9,2E+03	2,0E+05	3,1E+04	5,7E+03	4,8E+01	1,8E+01	2,7E+02	9,1E+02
Ecotoxicity, freshwater	CTUe	3,6E+05	4,7E+07	6,9E+07	2,8E+06	4,2E+04	2,2E+04	2,3E+05	5,1E+06
Land use	Pt	2,8E+05	1,5E+08	1,2E+08	2,7E+06	4,0E+03	6,2E+03	1,2E+05	2,8E+05
Water use	m^3 depriv.	0,0E+00	2,6E+06	1,8E+07	1,2E+04	2,7E+03	5,9E+05	8,9E+02	2,6E+04
Resource use, fossils	MJ	0,0E+00	1,4E+07	1,1E+07	2,1E+06	1,3E+05	3,9E+04	2,2E+05	8,3E+05
Resource use, minerals and metals	kg Sb eq	0,0E+00	3,0E+00	2,6E+01	8,0E+00	2,7E-02	1,9E-02	3,4E-01	6,8E-01

Figure 4. Environmental results and hotspots for the baseline scenario of the transition farm in Spain.

RESULTS: ALTERNATIVE SCENARIO

The following paragraph reports the results of the alternative scenarios, where the two different technologies previously presented are implemented in the production process of piglets in Spain and fattening pigs in Italy. A comparison with the respective baseline scenarios and a discussion of the results also follows.

It should be borne in mind that for the modeling of the alternative scenario in the Italian farms, the technologies were considered as if implemented only in the fattening phase facilities (pigs weighing from 50-80 kg onwards) and not in the sow reproduction and piglet growth phases, therefore affecting only partially the farms emissions.

There were several indicators affected by the emission reductions achieved by the technologies. However, in some cases these reductions were overwritten by the addition of infrastructure, consumables and electricity use from the scrubber. In particular, the citric acid was the major contributor to the impact from the wet scrubber.

Below are tables with the absolute results of the impact characterization for the baseline, wet scrubber and dry scrubber in comparison. In greater detail, Tables 8 and 9 report the results for farm A for the scenarios of maximum abatement achievable with the technologies and median abatement respectively, while Tables 10 and 11 report the same results for farm B. When the relative difference between the baseline scenario and the alternative scenarios is negative, and therefore leading to a reduction in impacts, it is highlighted in green in the tables.

As with the baseline scenario above, slightly different values between the two farms for the alternative scenarios can be observed for the alternative scenarios, but the relative differences between the baseline and the alternative scenarios follow the same trends between the two farms.

Focusing on the relative differences, it can be observed that overall the dry scrubber has had more positive environmental performances than the wet scrubber. This is for three reasons:

1. the impact categories positively influenced in the dry scrubber scenario are always more than those in the wet scrubber scenario. In fact, the latter has led to reductions in impact always and only for two categories, namely particulate matter formation potential and terrestrial eutrophication, while the dry scrubber has led to improvements, albeit small, also for other categories including climate change, acidification, marine eutrophication and terrestrial ecotoxicity.
2. For the two categories improved also by the wet scrubber, the dry scrubber has in any case achieved higher mitigations: for PM formation a maximum of -25% and -18% in farms A and B against -14% and -10% in the wet scrubber scenario; and for terrestrial eutrophication a maximum of -24% and 16% in farms A and B versus -18% and -12% in the wet scrubber scenario.

3. Another difference also appears evident: the impact categories not influenced by the emissions abatement given by the machinery. For these, in fact, in the wet scrubber scenario there are non-negligible increases in the impact, which in the worst case (Farm A, maximum emissions reduction scenario) are even greater than 50% for ozone depletion, ionizing radiation, fossil resource use and even greater of 100% for mineral and metal resource use. In the case of the dry scrubber, however, these increases are very limited, always less than 5% across categories, farms and efficiencies scenarios.

Table 8. Farm A in Italy: Environmental impact results for Baseline, Wet Scrubber, Dry Scrubber, and Impact change (%) of both technologies (wet and dry scrubber) with respect to the baseline for the maximum emissions abatement scenario.

Farm A - MAX	**Scenario**	**Baseline**	**Wet Scrubber**	**Dry scrubber**	**WS vs BS**	**DS vs BS**
Impact Category	**Units**	**kg^{-1} LW**	**kg^{-1} LW**	**kg^{-1} LW**	**Change (%)**	**Change (%)**
Climate change	kg CO_2 eq	6.04E+00	6.88E+00	6.03E+00	13.85%	-0.22%
Ozone depletion	kg CFC11 eq	1.50E-07	2.51E-07	1.53E-07	67.16%	1.69%
Ionising radiation	kBq U-235 eq	8.34E-02	1.27E-01	8.60E-02	52.37%	3.06%
Photochemical ozone formation	kg NMVOC eq	1.12E-02	1.40E-02	1.13E-02	25.11%	0.49%
Particulate matter	disease inc.	8.68E-07	7.44E-07	6.49E-07	-14.22%	-25.16%
Human toxicity, non-cancer	CTUh	5.95E-08	9.63E-08	5.97E-08	61.92%	0.41%
Human toxicity, cancer	CTUh	1.83E-09	2.71E-09	1.85E-09	47.72%	1.00%
Acidification	mol H+ eq	2.44E-02	3.06E-02	2.33E-02	25.43%	-4.37%

Eutrophication, freshwater	kg P eq	6.65E-04	9.44E-04	6.72E-04	41.95%	0.98%
Eutrophication, marine	kg N eq	2.25E-02	2.33E-02	2.16E-02	3.41%	-4.05%
Eutrophication, terrestrial	mol N eq	3.55E-01	2.91E-01	2.71E-01	-18.17%	-23.66%
Ecotoxicity, freshwater	CTUe	7.29E+01	1.09E+02	7.19E+01	48.96%	-1.30%
Land use	Pt	2.10E+02	2.19E+02	2.10E+02	4.29%	0.05%
Water use	m^3 depriv.	1.86E+01	1.98E+01	1.86E+01	6.54%	0.07%
Resource use, fossils	MJ	1.42E+01	2.30E+01	1.45E+01	62.35%	2.13%
Resource use, minerals and metals	kg Sb eq	1.29E-05	3.19E-05	1.34E-05	146.48%	3.69%

Table 9. Farm A in Italy: Environmental impact results for Baseline, Wet Scrubber, Dry Scrubber, and Impact change (%) of both technologies (wet and dry scrubber) with respect to the baseline for the median emissions abatement scenario.

Farm A - MEDIAN	**Scenario**	**Baseline**	**Wet Scrubber**	**Dry scrubber**	**WS vs BS**	**DS vs B**
Impact Category	**Units**	**kg^{-1} LW**	**kg^{-1} LW**	**kg^{-1} LW**	**Change (%)**	**Change (%)**
Climate change	kg CO_2 eq	6.04E+00	6.39E+00	6.05E+00	5.77%	0.17%
Ozone depletion	kg CFC11 eq	1.50E-07	1.92E-07	1.53E-07	27.96%	1.69%
Ionising radiation	kBq U-235 eq	8.34E-02	1.04E-01	8.60E-02	24.75%	3.06%
Photochemical ozone formation	kg NMVOC eq	1.12E-02	1.24E-02	1.13E-02	10.18%	0.49%
Particulate matter	disease inc.	8.68E-07	8.24E-07	8.02E-07	-5.03%	-7.53%

Human toxicity, non-cancer	CTUh	5.95E-08	7.38E-08	5.98E-08	24.01%	0.57%
Human toxicity, cancer	CTUh	1.83E-09	2.30E-09	1.85E-09	25.76%	1.00%
Acidification	mol H+ eq	2.44E-02	2.69E-02	2.42E-02	10.26%	-0.92%
Eutrophication, freshwater	kg P eq	6.65E-04	7.80E-04	6.72E-04	17.16%	0.98%
Eutrophication, marine	kg N eq	2.25E-02	2.28E-02	2.22E-02	1.40%	-1.18%
Eutrophication, terrestrial	mol N eq	3.55E-01	3.31E-01	3.30E-01	-6.75%	-7.19%
Ecotoxicity, freshwater	CTUe	7.29E+01	8.68E+01	7.29E+01	19.13%	-0.01%
Land use	Pt	2.10E+02	2.14E+02	2.10E+02	1.70%	0.05%
Water use	m^3 depriv.	1.86E+01	1.91E+01	1.86E+01	2.58%	0.07%
Resource use, fossils	MJ	1.42E+01	1.80E+01	1.45E+01	26.89%	2.13%
Resource use, minerals and metals	kg Sb eq	1.29E-05	2.05E-05	1.34E-05	58.43%	3.69%

Table 10. Farm B in Italy: Environmental impact results for Baseline, Wet Scrubber, Dry Scrubber, and Impact change (%) of both technologies (wet and dry scrubber) with respect to the baseline for the maximum emissions abatement scenario.

Farm B - MAX	**Scenario**	**Baseline**	**Wet Scrubber**	**Dry scrubber**	**WS vs BS**	**DS vs BS**
Impact Category	**Units**	**kg^{-1} LW**	**kg^{-1} LW**	**kg^{-1} LW**	**Change (%)**	**Change (%)**
Climate change	kg CO_2 eq	4.95E+00	5.48E+00	4.95E+00	10.67%	-0.13%
Ozone depletion	kg CFC11 eq	1.70E-07	2.34E-07	1.72E-07	37.64%	1.09%
Ionising radiation	kBq U-235 eq	9.93E-02	1.27E-01	1.01E-01	27.95%	1.89%

Photochemical ozone formation	kg NMVOC eq	1.03E-02	1.21E-02	1.03E-02	17.39%	0.39%
Particulate matter	disease inc.	7.67E-07	6.88E-07	6.28E-07	-10.25%	-18.13%
Human toxicity, non-cancer	CTUh	3.77E-08	6.11E-08	3.79E-08	62.00%	0.51%
Human toxicity, cancer	CTUh	1.30E-09	1.87E-09	1.31E-09	44.29%	1.04%
Acidification	mol H+ eq	3.33E-02	3.73E-02	3.27E-02	11.80%	-1.98%
Eutrophication, freshwater	kg P eq	1.74E-03	1.92E-03	1.75E-03	10.19%	0.27%
Eutrophication, marine	kg N eq	1.38E-02	1.43E-02	1.32E-02	3.53%	-4.18%
Eutrophication, terrestrial	mol N eq	3.39E-01	2.98E-01	2.86E-01	-12.08%	-15.72%
Ecotoxicity, freshwater	CTUe	7.74E+01	1.00E+02	7.69E+01	29.24%	-0.73%
Land use	Pt	1.95E+02	2.01E+02	1.95E+02	2.93%	0.04%
Water use	m^3 depriv.	4.46E+01	4.53E+01	4.46E+01	1.73%	0.02%
Resource use, fossils	MJ	1.37E+01	1.93E+01	1.39E+01	40.89%	1.61%
Resource use, minerals and metals	kg Sb eq	8.98E-06	2.10E-05	9.33E-06	134.06%	3.90%

Table 11. Farm B in Italy: Environmental impact results for Baseline, Wet Scrubber, Dry Scrubber, and Impact change (%) of both technologies (wet and dry scrubber) with respect to the baseline for the median emissions abatement scenario.

Farm B - MEDIAN	**Scenario**	**Baseline**	**Wet Scrubber**	**Dry scrubber**	**WS vs BS**	**DS vs BS**
Impact Category	**Units**	**kg^{-1} LW**	**kg^{-1} LW**	**kg^{-1} LW**	**Change (%)**	**Change (%)**
Climate change	kg CO_2 eq	4.95E+00	5.17E+00	4.96E+00	4.46%	0.18%

Ozone depletion	kg CFC11 eq	1.70E-07	1.97E-07	1.72E-07	15.69%	1.09%
Ionising radiation	kBq U-235 eq	9.93E-02	1.12E-01	1.01E-01	13.25%	1.89%
Photochemical ozone formation	kg NMVOC eq	1.03E-02	1.10E-02	1.03E-02	7.07%	0.39%
Particulate matter	disease inc.	7.67E-07	7.39E-07	7.25E-07	-3.60%	-5.41%
Human toxicity, non-cancer	CTUh	3.77E-08	4.68E-08	3.79E-08	24.08%	0.67%
Human toxicity, cancer	CTUh	1.30E-09	1.62E-09	1.31E-09	24.66%	1.04%
Acidification	mol H+ eq	3.33E-02	3.49E-02	3.32E-02	4.77%	-0.38%
Eutrophication, freshwater	kg P eq	1.74E-03	1.81E-03	1.75E-03	4.18%	0.27%
Eutrophication, marine	kg N eq	1.38E-02	1.40E-02	1.36E-02	1.45%	-1.21%
Eutrophication, terrestrial	mol N eq	3.39E-01	3.24E-01	3.23E-01	-4.48%	-4.77%
Ecotoxicity, freshwater	CTUe	7.74E+01	8.63E+01	7.75E+01	11.45%	0.04%
Land use	Pt	1.95E+02	1.97E+02	1.95E+02	1.16%	0.04%
Water use	m^3 depriv.	4.46E+01	4.49E+01	4.46E+01	0.68%	0.02%
Resource use, fossils	MJ	1.37E+01	1.61E+01	1.39E+01	17.68%	1.61%
Resource use, minerals and metals	kg Sb eq	8.98E-06	1.38E-05	9.33E-06	53.65%	3.90%

As for Spain, wet scrubber was more efficient reducing ammonia emissions compared to the dry scrubber in the Spanish context (as reported in Tables 6 & 7), which was related to an improvement in different impact categories (Tables 12 & 13). Particulate matter and terrestrial eutrophication reduced the impact by 9.66 and 1.80%. Also, marine eutrophication, but to a lesser extent. Ammonia emissions reduction had also an impact on cancer human toxicity, acidification and freshwater ecotoxicity, but this was overwritten by the increase in impact to these categories coming from the consumables used for the wet scrubber operation, and specifically citric acid consumption.

Indeed, while reducing impact for the abovementioned categories, both wet and dry scrubbers add impact over the baseline scenario for all remaining categories. This is because the implementation of these technologies involves extra energy (electricity), infrastructure and, in the case of wet scrubber, also consumables (citric acid and water).

The dry scrubber showed less efficiency in the removal of ammonia, but it also added less impact to the overall results for each indicator (<1% contribution to all indicators).

Table 12. Transition farm in Spain: Environmental impact results for Baseline, Wet Scrubber, Dry Scrubber, and Impact change (%) of both technologies (wet and dry scrubber) with respect to the baseline for the maximum emissions abatement scenario.

TRANSITION FARM - MAX	**Scenario**	**Baseline**	**Wet Scrubber**	**Dry scrubber**	**WS vs BS**	**DS vs BS**
Impact Category	**Units**	**kg^{-1} LW**	**kg^{-1} LW**	**kg^{-1} LW**	**Change (%)**	**Change (%)**
Climate change	kg CO_2 eq	2,18E+00	2,21E+00	2,10E+00	1,42%	-3,68%
Ozone depletion	kg CFC11 eq	1,44E-07	1,48E-07	1,44E-07	2,71%	0,16%
Ionising radiation	kBq U-235 eq	3,00E-01	3,04E-01	3,01E-01	1,37%	0,44%
Photochemical ozone formation	kg NMVOC eq	5,89E-03	6,00E-03	5,87E-03	1,84%	-0,26%
Particulate matter	disease inc.	2,55E-07	2,30E-07	2,35E-07	-9,66%	-7,98%
Human toxicity, non-cancer	CTUh	5,48E-08	5,62E-08	5,48E-08	2,57%	-0,03%
Human toxicity, cancer	CTUh	1,47E-09	1,53E-09	1,47E-09	4,66%	0,39%
Acidification	mol H+ eq	3,07E-02	3,09E-02	3,06E-02	0,70%	-0,03%

Eutrophication, freshwater	kg P eq	4,15E-04	4,26E-04	4,16E-04	2,59%	0,25%
Eutrophication, marine	kg N eq	1,11E-02	1,11E-02	1,11E-02	-0,16%	-0,41%
Eutrophication, terrestrial	mol N eq	1,33E-01	1,31E-01	1,31E-01	-1,80%	-1,29%
Ecotoxicity, freshwater	CTUe	6,75E+01	6,88E+01	6,75E+01	1,95%	0,04%
Land use	Pt	1,43E+02	1,43E+02	1,43E+02	0,24%	0,01%
Water use	m^3 depriv.	1,13E+01	1,14E+01	1,13E+01	0,45%	0,01%
Resource use, fossils	MJ	1,51E+01	1,55E+01	1,52E+01	2,49%	0,30%
Resource use, minerals and metals	kg Sb eq	2,08E-05	2,15E-05	2,09E-05	3,61%	0,71%

Table 13. Transition farm in Spain: Environmental impact results for Baseline, Wet Scrubber, Dry Scrubber, and Impact change (%) of both technologies (wet and dry scrubber) with respect to the baseline for the median emissions abatement scenario.

TRANSITION FARM - MEDIAN	**Scenario**	**Baseline**	**Wet Scrubber**	**Dry scrubber**	**Impact change (%)**	**Impact change (%)**
Impact Category	**Units**	**kg^{-1} LW**	**kg^{-1} LW**	**kg^{-1} LW**	**WS vs BS**	**DS vs BS**
Climate change	kg CO_2 eq	2,18E+00	2,20E+00	2,13E+00	0,80	-2,50
Ozone depletion	kg CFC11 eq	1,44E-07	1,46E-07	1,44E-07	1,47	0,16
Ionising radiation	kBq U-235 eq	3,00E-01	3,02E-01	3,01E-01	0,75	0,44
Photochemical ozone formation	kg NMVOC eq	5,89E-03	5,95E-03	5,88E-03	1,03	-0,13
Particulate matter	disease inc.	2,55E-07	2,37E-07	2,38E-07	-6,86	-6,79
Human toxicity, non-cancer	CTUh	5,48E-08	5,56E-08	5,48E-08	1,41	0,04

Human toxicity, cancer	CTUh	1,47E-09	1,52E-09	1,47E-09	3,88	0,39
Acidification	mol H+ eq	3,07E-02	3,08E-02	3,07E-02	0,39	0,01
Eutrophication, freshwater	kg P eq	4,15E-04	4,21E-04	4,16E-04	1,46	0,25
Eutrophication, marine	kg N eq	1,11E-02	1,11E-02	1,11E-02	-0,07	-0,29
Eutrophication, terrestrial	mol N eq	1,33E-01	1,32E-01	1,32E-01	-0,95	-0,92
Ecotoxicity, freshwater	CTUe	6,75E+01	6,82E+01	6,75E+01	1,08	0,07
Land use	Pt	1,43E+02	1,43E+02	1,43E+02	0,13	0,01
Water use	m^3 depriv.	1,13E+01	1,13E+01	1,13E+01	0,24	0,01
Resource use, fossils	MJ	1,51E+01	1,53E+01	1,52E+01	1,40	0,30
Resource use, minerals and metals	kg Sb eq	2,08E-05	2,12E-05	2,09E-05	2,04	0,71

DISCUSSION

Both tested technologies showed their potential to reduce emissions in the pig housing stage. An holistic environmental assessment allowed us to have a more deep view of the results. Various trade-offs have been observed between the categories that are affected by the emission abatement and those that are instead more linked to energy and resource use due to the addition of processes and infrastructure in the alternative scenarios, necessary for the construction and operation of scrubbers.

The results in Spain and Italy confirmed these trade-offs, albeit to different extents. However, the comparison of these results can only provide general conclusions since they are not directly comparable as the analysis in Spain is limited to a specific phase of the fattening pigs life cycle. In general, however, it is noted that in Italy greater mitigations have been achieved for the categories affected positively by scrubbers, but at the same time greater trade-offs for the others. This is probably related to the fact that the production cycle of transition piglets is much shorter than that of fattening pigs in Italy, and therefore the influence of the scrubber is less marked.

A small possibility of mitigation regarding the alternative scenario, at least for the wet scrubber, is represented by the possibility of recycling the steel that makes up the machinery. In any case, the impact of the infrastructure itself, including its disposal, was secondary to that of the consumables (citric acid and electricity in particular) and therefore the environmental optimization options of this alternative scenario should rather be sought in their more efficient use.

Sensitivity analysis

Overall contribution of emissions to acidification and terrestrial eutrophication seemed smaller than values found in the literature. Regionalization of characterization factors when available was used (Table 14). These choices were made in consistency with the location of the study. Therefore, specific characterization for Spain and Italy factors were used to characterize acidification, and terrestrial eutrophication (Seppälä et al., 2006; Posch et al., 2008). Thus, a sensitivity analysis was performed to test the robustness of the results to changes in the methodological choices. This test was performed for those impact categories and flows where we had the choice to utilize regionalized CFs (thus, ammonia and nitrogen dioxide for acidification and eutrophication, terrestrial).

Table 14. Characterization Factors regionalized vs unspecific for main contributors to acidification and terrestrial eutrophication impact categories.

Impact category	Flow	Unspecified	Regionalized, SP	Regionalized, IT
Acidification (mol H+ eq / kg emitted pollutant)	Ammonia	3.02	0.076	0.12
	Nitrogen dioxide, Nitrogen oxides	0.74	0.052	0.065
Eutrophication terrestrial (mol N eq / kg emitted pollutant)	Ammonia	13.47	3.431	8.363
	Nitrogen dioxide, Nitrogen oxides	4.26	0.877	1.48

Impact results on both acidification and terrestrial eutrophication resulted sensitive to the use of different characterization factors. As can be seen from the table 14, in fact, the non-specific factors are much higher than the regionalized ones. This translates into the fact that in the baseline scenarios the impact categories affected by this change have higher impacts. At the same time, however, these are mitigated to a greater extent in the alternative scenarios. Therefore, in the scenario with maximum emission abatement efficiency, reduction in impact for acidification changes from being close to null (increase in impact of 0.7% and reduction of 0.03% with wet and dry scrubber respectively) to a reduction of 6.74% and 4.49% with wet scrubber and dry scrubber respectively when using unspecific characterization factors in Spain. Same occurs in impact to terrestrial eutrophication where impact reduction assessed by using wet and dry scrubber is 7.55% and 4.77% respectively, compared to the previous 1.80% and 1.29% when impact is assessed with the regionalized characterization factors.

The same trend was observed for the results relating to Italy. Again, in the scenario with maximum emission abatement efficiency, acidification goes from +25% and -4% to -18% and -25% for wet and dry scrubber scenarios, respectively, for farm A; and from +12% and -2% to -13% and -18% for wet and dry scrubber scenarios, respectively, for farm B. In Italy, therefore, the trend was even more accentuated, not much for the difference between regionalized and unspecific characterization factors, greater for Spain, but for the reason already mentioned above, i.e. that scrubbers in the Italian production cycle have had a greater influence (both positive and negative) due to the longer duration of the phases of use.

Thus, potential of wet and dry scrubber to reduce impact depends not only on the emission reduction achieved, but also on the characterization of the impact which is region dependent. Therefore, results of this study cannot be extrapolated to other regions without considering these methodological aspects.

CONCLUSIONS

A comprehensive vision should allow to think of potential effects on the pig's wellbeing that could ultimately result in better pig performance, or on the social aspects linked to a potential reduction of odours for the farm workers. Moreover, regarding the increased energy consumption alternative energy sources could be assessed in the future such as the use of solar panels to feed the scrubber energy needs. This could increase the use of resources but reduce impact on ionising radiation in the Spanish farms, for example. On the other hand, given that farms are located in the Mediterranean basin, where water can be scarce, the reutilization of water from the farm could be assessed to reduce the contribution of its consumption for the wet scrubber functioning. At last, alternative products to citric acid could be assessed, such as residual acids from other industrial systems that could potentially reduce the environmental impact at the farm.

This leads to conclude that a better environmental performance can be difficult to achieve by one unique solution, but that a combination of different technologies at different farm stages is needed for better results. Life cycle assessment showed to be a useful tool to assess the overall balance between the achieved emission reduction and these added impacts from implementing these technologies.

References

De Vries J.W., Melse R.W. (2018). Comparing environmental impact of air scrubbers for ammonia abatement at pig houses: a life cycle assessment. Biosyst. Eng., 161 (2017), pp. 53-61, 10.1016/j.biosystemseng.2017.06.010

Dourmad, J. Y., Ryschawy, J., Trousson, T., Bonneau, M., Gonzàlez, J., Houwers, H. W. J., Hviid, M., Zimmer, C., Nguyen, T. L. T., & Morgensen, L. (2014). Evaluating environmental impacts of contrasting pig farming systems with life cycle assessment. Animal, 8(12), 2027–2037. https://doi.org/10.1017/S1751731114002134

EEA, 2019. EMEP/EEA air pollutant emission inventory Guidebook 2019 – 3.B, Manure management.

FAO. 2018. Environmental performance of pig supply chains: Guidelines for assessment (Version 1). Livestock Environmental Assessment and Performance Partnership. Rome, FAO. 172 pp.

Fazio, S., Castellani, V., Sala, S., Schau, E. M., Secchi, M., Zampori, L., & Diaconu, E. (2018). Supporting information to the characterisation factors of recommended EF Life Cycle Impact Assessment methods, version 2.0. from ILCD to EF 3.0, EUR29600 EN, European Comission, Ispra, 2018. In ISBN 978-92-79-98584-3, doi:10.2760/002447, PUBSY No. JRC114822. https://doi.org/10.2760/671368

IPCC, 2019. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and Other Land Use. Chapter 10: Emissions from Livestock and Manure Management.

ISO-14040. (2006). Environmental management-Life cycle assessment-Principles and framework. International standard 14040. . International Organisation for Standardisation ISO, Geneva.

ISO-14044. (2006). 14044:2006/AMD 2:2020 Environmental management — Life cycle assessment — Requirements and guidelines — Amendment 2.

Poore, J., & Nemecek, T. (2018). Reducing food's environmental impacts through producers and consumers. Science (New York, N.Y.), 360(6392), 987–992. https://doi.org/10.1126/science.aaq0216

Posch, M., Seppälä, J., Hettelingh, J. P., Johansson, M., Margni, M., & Jolliet, O. (2008). The role of atmospheric dispersion models and ecosystem sensitivity in the determination of characterisation factors for acidifying and eutrophying emissions in LCIA. International Journal of Life Cycle Assessment, 13(6), 477–486. https://doi.org/10.1007/s11367-008-0025-9

Seppälä, J., Posch, M., Johansson, M., & Hettelingh, J. P. (2006). Country-dependent characterisation factors for acidification and terrestrial eutrophication based on accumulated exceedance as an impact category indicator. International Journal of Life Cycle Assessment, 11(6), 403–416. https://doi.org/10.1065/lca2005.06.215

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., and Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. The International Journal of Life Cycle Assessment, 21(9), pp.1218–1230.

Zampori, L., & Pant, R. (2019). Suggestions for updating the Product Environmental Footprint (PEF) method. In Publications Office of the European Union. https://doi.org/10.2760/424613

3.3. Improvement of human health and environmental costs in the European Union by air scrubbers in intensive pig farming

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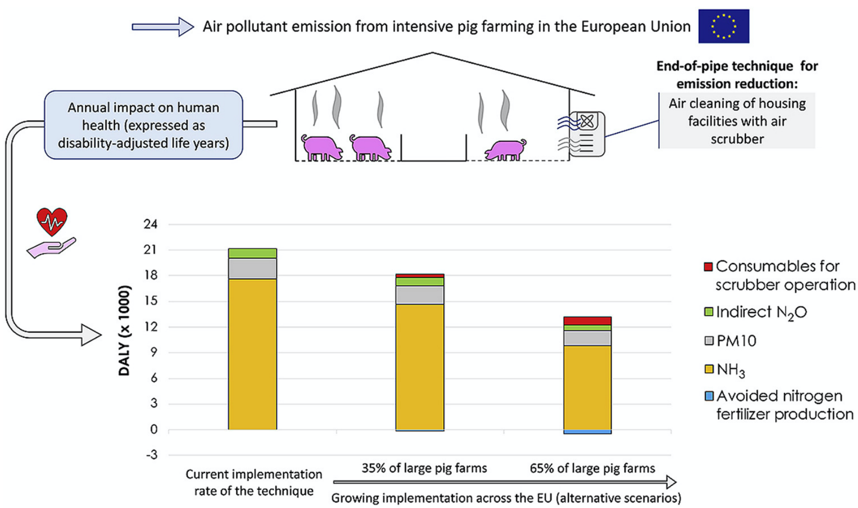
Graphical abstract



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Improvement of human health and environmental costs in the European Union by air scrubbers in intensive pig farming

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A B S T R A C T

Intensive pig farming is responsible for significant air pollutant emissions. This study explores the effect that the large-scale implementation of air cleaning technologies (wet acid scrubbers) for pig housing facilities could have in the European Union. Emissions related to the housing stage of NH_3, PM10, NMVOC and indirect N_2O from large pig farms (>1000 heads of sows or fattening pigs) are first estimated in the actual situation (current scenario - CS), considering implementation rates and removal efficiencies of the different emission abatement techniques available. Subsequently, alternative scenarios (AS1 and AS2) are simulated with a growing implementation rate of the wet acid scrubber (35% and 65% of the concerned pig farms in all Member States). A comparison between the scenarios was carried out, taking into account emissions reduction, consumables for scrubber operation and environmental credit given by the avoidance of synthetic nitrogen fertilizer production. The annual impact on human health of 21,212 disability-adjusted life years (DALY) in CS was significantly reduced in AS1 (-15%) and in AS2 (-40%), showing that the environmental trade-off given by the consumables is largely overwhelmed by emission abatement. At the same time, the current environmental cost to society of the concerned emissions was estimated at 4154 million € per year (of which 89% due to NH_3), which also was reduced in alternative scenarios (−668 and −1765 million € for AS1 and AS2). The abatement of NH_3, on which the wet acid scrubber expresses the greatest removal efficiency, was fundamental in both reducing the human health impact and environmental costs, demonstrating the key environmental role of this pollutant and the growing need to find solutions for its containment in the EU.

1. Introduction

The agricultural sector is the major responsible of ammonia (NH_3) emissions in the EU, accounting for 92% of them in 2017 (EEA, 2019a). In particular, the livestock sector contributes to about 80% of the agricultural share due to NH_3 emissions from effluents, occurring during permanence in housing facilities, storage and field application. The deal of livestock effluents management is that even when it is possible to conserve ammoniacal nitrogen at a certain stage (e.i. during the permanence in animal housing), this still remains available to volatilize for subsequent ones (handling, storage, field spreading) (Reis et al., 2015). Agriculture also contributes to PM pollution, by means of direct emissions from livestock (Cambra-Lopez et al., 2010) and mechanization (Lovarelli and Bacenetti, 2019) and, indirectly, by means of NH_3. In fact, the latter may react with sulphur dioxide (SO_2) and nitrogen oxides (NO_x) while in the atmosphere, leading to the formation of secondary sulphate and nitrate particles, major components of fine particular matter (PM2.5) (Lovarelli et al., 2020b). Indeed, Backes et al. (2016) for Europe and Zhao et al. (2017) for China have shown that the reduction of NH_3 emissions of agricultural origin can contribute contain PM2.5 pollution.

Efforts made by the European Commission (EC) and Member States under the Convention on Long-range Transboundary Air Pollution (UN, 2020) and the protocols that extend it have already led to significant improvements in NH_3 emissions, achieving a 24% decrease from 1990 to 2017 (EEA, 2019a). For the livestock sector the reduction has primarily been due to a decrease in livestock numbers (especially cattle), changes in the handling and management of effluents and improved feeding techniques (Jacobsen et al., 2019). In recent years, however, the downward NH_3 emission trend

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Acronyms and abbreviations

AS	Alternative scenario
BAT	Best available techniques
CS	Current scenario
DALY	Disability-adjusted life years
EC	European Commission
EU	European Union
IED	Industrial Emissions Directive
N_2O	Dinitrogen monoxide
NH_3	Ammonia
NMVOC	Non-methane volatile organic compounds
NO_x	Nitrogen oxides
PM	Particulate matter
SO_2	Sulphur dioxide
TAN	Total ammoniacal nitrogen
UN	United Nations
VOC	Volatile organic compounds
WAS	Wet acid scrubber
WHO	World Health Organization
YLD	Years lost due to disability
YLL	Year of life lost

has slowed down and since 2014 it was even found to be positive (+2.3% from 2014 to 2017) (EEA, 2019a). Moreover, international policies adopted in recent decades for the abatement of anthropogenic emissions of SO_2 and NO_x, also involved in PM2.5 formation, have led to greater reductions in relative terms than those of NH_3 (Reis et al., 2015), which favors greater focus on the latter.

The application of engineering principles and precision techniques for monitoring and manage livestock production processes basically allows to improve animal welfare and health (Berckmans, 2014; Lovarelli et al., 2020a). Especially thanks to the consequent superior productive and reproductive animal performances, this is also accepted as a way to make livestock systems more environmentally sustainable (Tullo et al., 2019). Hence, the use of specific technologies can play a role in solving environmental challenges, if these present a positive balance in conserving the natural environment and contrasting the negative impact of human activities (Aarras et al., 2014).

Air scrubbers are air cleaning devices used to control and remove pollutants from exhaust air, commonly adopted for industrial streams, but which can also be used in pig and poultry housing facilities (Van der Heyden, 2015). For the latter sector air scrubbers are normally installed with ammonia as the main target substance for which to reduce emissions, but also involve, to a lesser extent, abatements of other pollutants such as VOC and PM, since these are partially captured by washing water (Van der Heyden, 2015).

Air scrubbers represent an end-of-pipe technique, i.e. a technique that reduces final emissions by some additional process but does not change the fundamental operation of the core process (Santonja et al., 2017). In the Best Available Techniques[1] (BAT) Reference Document for the Intensive Rearing of Poultry or Pigs the air cleaning systems are listed in the techniques to be considered BAT (EC, 2017).

As regards the pig sector, the wet acid scrubber (WAS), which involves the capture of NH_3 by means of an acid solution, is currently the most widely applied air cleaning technology (Table 1). It entails greater removal efficiency of NH_3 (normally in a range between 70% and 99% of the background air concentration) and lower water consumption (and consequently also less output stream, which translates into lower management costs) compared to bioscrubber (or biotrickling filter), the main alternative technology currently available.

This technology is increasingly promising in environmental terms and could play an important role in the near future for air pollutants control from the agricultural sector in the EU. This could contribute to fall within the PM2.5 concentration thresholds set by the Air Quality Directive, as well as within the National Emission Ceilings of air pollutants, set by Directive (2016)/2284/EU, to be achieved by all Member States by 2030. Moreover, looking for environmentally-friendly food systems falls within the objectives of the European Green Deal (EC, 2019), and in particular of the Farm to Fork Strategy (EC, 2020).

In this study, large-scale implementations of the WAS in EU pig housing facilities are simulated and potential benefits on human health are assessed. In addition, economic considerations are made related to saving the society damage costs given by air pollutants from pig housing thanks to their containment. To the best of the authors' knowledge, this is the first study that: focuses on scenarios of a large-scale implementation in the EU of an air cleaning technology in pig housing, estimating the consequent emissions abatement obtainable; in this context, carries out an environmental assessment in an endpoint perspective, focusing on the impact on human health, and makes economic considerations beyond operating costs by coupling emissions with environmental costs.

2. Methods

In order to explore the consequences that the large-scale implementation in the EU pig housing facilities of the WAS could have, methodology has been structured as follows: section 2.1 defines the analysis reference framework and describes how the starting emission inventory was built; in section 2.2 different scenarios are modeled, in order to be able to compare the current situation with hypothetical alternatives in which the WAS technology is widely adopted in pig farming; finally, sections 2.3 and 2.4 deal with the methods used to quantify human health impact and environmental costs, respectively. A schematic overview of the methodologies is illustrated in Fig. 1.

2.1. Definition of the reference framework and emission inventory

Quantifying the magnitude of emissions is a key component for the development of control policies for atmospheric pollutants (Rebolledo et al., 2013). Therefore, an inventory of NH_3, PM10 and NMVOC emissions related to pig production was first built. These have been selected as they are among the pollutants that cause the greatest public concerns related to pig farming activities and, at the same time, their emissions are the most affected by the implementation of the technology addressed in this study (i.e. the WAS). Only emissions that occur at the housing stage were computed, being the stage affected by the WAS. On the other hand, emissions from handling, storage and distribution of effluents were not

[1] According to the Directive 2010/75/EU on Industrial Emissions (IED), 'techniques' refers to the technology used to prevent and/or reduce emissions and the way in which the installation is designed, built, maintained, operated and decommissioned; 'available techniques' means those developed on a scale which allows implementation in the relevant industrial sector, under economically and technically viable conditions, taking into consideration the costs and advantages, whether or not the techniques are used or produced inside the Member State in question, as long as they are reasonably accessible to the operator; 'best' means most effective in achieving a high general level of protection of the environment as a whole.

Table 1
Main considerations to be taken into account regarding the installation of the wet acid scrubber in pig housing facilities.

Pros	Cons
- Very effective for ammonia emission abatement (with fluctuations given by ventilation rate, pollutant load, relative humidity and temperature of incoming air, etc.) (Van der Heyden et al., 2015); - Effective for VOC and PM emission abatement (Van der Heyden et al., 2015); - Could also have relevant capture effects for CH_4 and N_2O (Mostafa et al., 2020); - The water discharged contains high nitrogen concentration (3–9% according to Sigurnjak et al., 2019) and can be used as fertilizer with good agronomic performances (Martin et al., 2018); - Currently represents the most suitable air cleaning technology in economic (Santonja et al., 2017) and environmental (De Vries and Melse, 2017) terms. Confirming the latter, the WAS does not promote N_2O formation, which instead occurs for bioscrubbers as side effect of the NH_3 abatement reaction, causing an environmental trade-off with climate change (Dumont, 2018); - Can be designed for specific target substances according to the needs; can be combined with other technologies to form multi-stage scrubbers (Van der Heyden et al., 2015).	- Requires significant investment and operating costs. Melse et al. (2008) reported the former at 32.8 €/animal place while the latter at 8.2 €/animal place/year. Hence, considering the depreciation, the WAS would cost 10.3 €/animal place/year in total; - Involves a considerable water consumption and water input and discharge flows suffer from some uncertainty (Santonja et al., 2017). In any case, it requires efforts to manage an effluent stream; - Safety measures are required for the storage and handling of chemicals, specific staff training may be needed (Santonja et al., 2017); - If used with other acids other that sulfuric acid, the effluent solution may need to be disposed (Santonja et al., 2017); - It may not be suitable for facilities without centralized ventilation systems (Santonja et al., 2017).

considered. The reference pig population of each Member State was taken from the Eurostat database for the year 2018 (Eurostat, 2020a). The calculation only concerned animals raised in large farms (i.e. sows and fattening pigs bred in farms with more than 1000 heads of the same category), as these reflect intensive rearing practices and may be realistically involved in the installation of the WAS. The pig population housed in these farms actually represents the majority of the EU pig population, accounting for 78% and 75% of total sows and fattening pigs, respectively (elab. on Eurostat, 2020a). More details on the concerned pig population on which the emission inventory was built can be found in Tables S1 and S2 (supplementary materials). NH_3, PM and NMVOC emission factors (kg of pollutant$\cdot$ head^{-1} $\cdot$ year^{-1}) were derived using sources from official EU publications and databases (Table 2). Regarding pig nitrogen excretion, despite the availability of national emission factors, it was preferred to use a single European average reference (EEA, 2019b) due to the poor harmonization encountered across country-specific inventories, an issue already highlighted by Velthof et al. (2015).

NH_3 emitted by livestock systems may determine, after re-deposition, the formation of dinitrogen monoxide (N_2O) through nitrification and incomplete denitrification processes. These N_2O emissions, referred to as indirect, have been included in the assessment, being directly connected to NH_3, and computed considering the emission rate of 0.01 kg N_2O–N $\cdot$ kg NH_3–N emitted^{-1} (IPCC, 2019). The combination of the emission factors with the concerned pig population completed the starting emission inventory.

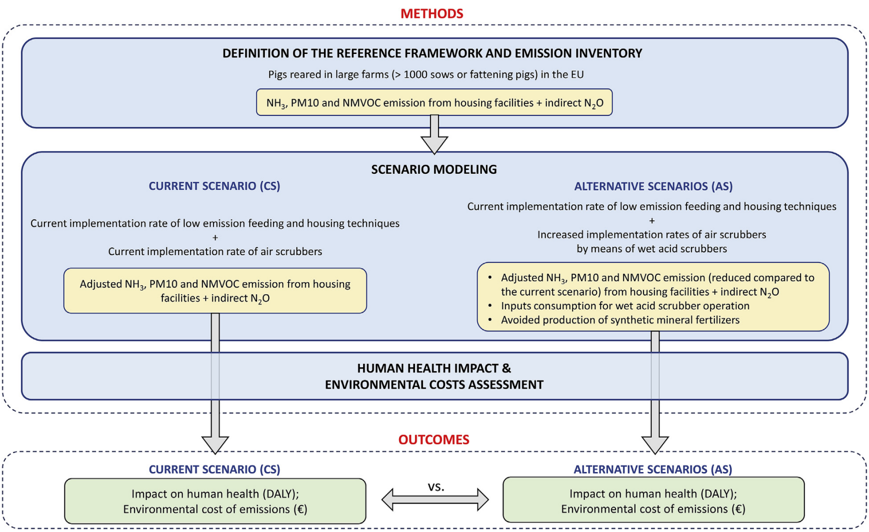

Fig. 1. Conceptual framework of the assessment.

Table 2
Parameters and emission factors used to build the emission inventory for NH_3, PM10 and NMVOC.

Item	Unit of measure	Category		Source
		Sows (and piglets)	Fattening pigs	
N excretion	kg N · head^{-1} · year^{-1}	34.5	12.1	EEA (2019b)
Proportion of N excreted as TAN	kg TAN · kg N^{-1}	0.7	0.7	EEA (2019b)
Proportion of excreta handled as slurry	Dimensionless	0.91	0.91	Eurostat (2020b)
NH_3–N emissions from TAN of slurry (during housing)	kg NH_3–N · kg slurry TAN^{-1}	0.35	0.27	EEA (2019b)
NH_3–N emissions from TAN of manure (during housing)	kg NH_3–N · kg manure TAN^{-1}	0.24	0.23	EEA (2019b)
PM10 emission factor (from animal husbandry)	kg PM10 · head^{-1} · year^{-1}	0.17	0.14	EEA (2019b)
Default values for Live Weights	kg	190 (WE)[a]; 204 (EE)[a]	61 (WE)[a]; 59 (EE)[a]	IPCC (2019b)
Default values for volatile solid excretion	kg · 1000 kg live weight^{-1} · day^{-1}	2.4 (WE)[a]; 2.0 (EE)[a]	4.9 (WE)[a]; 5.3 (EE)[a]	IPCC (2019b)
NMVOC emission factor (during housing)	kg NMVOC · kg VS excreted^{-1}	0.007042	0.001703	EEA (2019b)

[a] WE: Western Europe, including AT, BE, CZ, DE, DK, ES, FI, FR, GB, GR, IE, IT, LU, MT, NL, PT, SE, SI; EE: Eastern Europe, including BG, CY, EE, HR, HU, LT, LV, PL, RO, SK.

2.2. Scenario modeling

The emission inventory in its starting condition defines emissions of the main air pollutants from pig housing facilities in a condition of absolute lack of control measures, which does not actually correspond to the current condition as different emission reduction techniques are already widespread in pig farms. Therefore, first the current scenario (CS) was defined, i.e. a scenario in which the reduction techniques are applied at the housing stage to the current diffusion. Two alternative scenarios (AS1 and AS2) were subsequently modeled in which, compared to CS, the air cleaning technique is implemented at increasing rates, and considering that this occurs exclusively through the WAS technology.

Emission reduction techniques were divided into two categories: *air cleaning* and *feeding and housing management*. The first refers to the air scrubbing technique, the latter include all the other measures adopted at the pig housing stage with the aim of reducing NH_3 emissions. These are mainly represented by: precision feeding strategies, presence of deep pit (in case of a partly slatted floor), frequent slurry removal (by means of vacuum systems or flushing) and slurry cooling systems (Pexas et al., 2020a). The removal efficiencies considered for the two categories of techniques are shown in Table 3.

The removal efficiencies remain fixed across scenarios, which instead are differentiated by the diffusion (implementation rate) of the techniques themselves. The implementation rates considered for the three scenarios are shown in Table 4. Since no official nor detailed data on the diffusion across the EU of *feeding and housing management* techniques have been found, they were assumed to affected 50% of the total concerned pig population. This was considered fixed for the three scenarios, as the analysis was focused on the variability given by different implementation rates of *air cleaning* techniques. The assumption was made considering that the pig farms addressed in this study largely coincides with those subjects to the IED for operating permits.[2] These are officially required to monitor and report their environmental performances, demonstrating to apply one or more of the techniques listed in the BAT conclusions document, which also includes those of feeding and housing management. However, to check how this methodological choice affected the results, a sensitivity analysis was carried out in this regard.

With regard to air cleaning, the actual diffusion of this technique is also currently unknown. Both Van der Heyden et al. (2015) and Santonja et al. (2017) state that air scrubbers are fairly widespread in Belgium, Denmark, Germany and the Netherlands (hereinafter, north-continental countries), while in the other Member States this technology is uncommon. According to Blonk Consultants (2019), in The Netherlands about 35% of pig farms practice *air cleaning* techniques. This share was extended to the other north-continental countries, assuming the same implementation rate between them for CS. Implementation rate of *air cleaning* techniques was assumed at 5% for all other Member States. AS1 simulates a situation in which all Member States where the use of *air cleaning* techniques is currently uncommon reach the implementation rate of the north-continental ones. AS2 instead simulates a situation in which all Member States increase their implementation rate up to 65%, which corresponds to the current European average implementation limit (maximum feasible applicability) of this technology (elab. on Klimont and Winiwarter, 2011). In particular, the gap in the implementation rate of *air cleaning* between CS and the alternative scenarios has been assumed to be bridged exclusively by the adoption of the WAS.

In all three scenarios, the implementation rates of the two emission reduction techniques were considered to be uncorrelated, i.e. independent events. This leads to the possible occurrence of four cases in the simulation: application of *feeding and housing management*; application of *air cleaning*; application of both *feeding and housing management* and *air cleaning*; neither of the two techniques applied. These were determined with the following equations:

$$PA_{1,s,ms} = IR_{F\&H} - (PA_{3,s,ms}) \tag{1.1}$$

$$PA_{2,s,ms} = IR_{AC,s,ms} - (PA_{3,s,ms}) \tag{1.2}$$

$$PA_{3,s,ms} = (IR_{F\&H} IR_{AC,s,ms}) \tag{1.3}$$

Table 3
Emission reduction techniques considered and their assumed removal efficiencies. Indirect N_2O depends on emitted NH_3, therefore it is not individually influenced by the different reduction techniques.

Concerned air pollutant	Feeding and housing management	Air cleaning
NH_3	30 %[a b]	80 %[d]
PM10	25 %[a]	50 %[d]
NMVOC	20 %[c]	35 %[d]

[a] Blonk Consultants (2019).
[b] Consistent with the reference removal efficiencies of these techniques reported by the NEC Directive (2016/2284/EU) and the Ammonia Guidance Document (ECE/EB.AIR/120).
[c] Assumed considering information reported by Ni et al. (2012).
[d] Average removal efficiencies of the ranges reported by Van der Heyden et al. (2015).

[2] Farms with more than 2000 places for production of pigs (over 30 kg) or with more than 750 places for sows, as specified in Section 6.6 of Annex I to Directive 2010/75/EU on Industrial Emissions (IED).

Table 4
Implementation rates of emission reduction techniques for the three scenarios. Percentages express the share of the concerned pig population that is affected by emission reduction techniques across the specified countries.

		Scenario		
Emission reduction technique	Countries	CS	AS1	AS2
Feeding and housing management	All Member States	50%	50%	50%
Air cleaning	North-continental countries (BE, DE, DK, NL)	35%	35%	65%
	All others Member States	5%	35%	65%

$$PA_{4,s,ms} = 1 - (PA_{1,s,ms} + PA_{2,s,ms} + PA_{3,s,ms}) \quad (1.4)$$

where:

- $PA_{1,2,3,4}$: probability that a case of emission reduction technique application occurs (four cases: *feeding and housing management* [1], *air cleaning* [2], *feeding and housing management* and *air cleaning* [3], neither of the two techniques applied [4]), {%};
- *s, ms*: scenario (CS; AS1; AS2), Member State;
- $IR_{F\&H}$: implementation rate of *feeding and housing management* emission reduction techniques, {%};
- IR_{AC}: implementation rate of *air cleaning* emission reduction techniques, {%};

The whole process of estimating the emission concerned and adjusting to the different scenarios can be mathematically resumed as follows [Eq. (2)]:

$$E_{p,s} = \sum_{ms,c} PP_{ms,\,c}\, LF_{ms,c}\, EF_{p,c}\ \left[PA_{1,s,ms}\left(1 - Re_{F\&H,\,p}\right) + PA_{2,s,ms} \left(1 - Re_{AC,p}\right) + PA_{3,s,ms}\left(1 - Re_{F\&H,p}\right)\left(1 - Re_{AC,p}\right) + PA_{4,\ s,ms}\right]$$

where:

- *E*: total emission from EU large pig farms at the housing stage, $\{\text{Gg} \cdot \text{year}^{-1}\}$;
- *p, s, ms, c*: pollutant (NH_3; PM10; NMVOC), scenario (CS; AS1; AS2), Member State, pig category (sows; fattening pigs);
- *PP*: pig population, reference year: 2018, {heads};
- *LF*: share of population hosted in large farms (>1000 heads per pig category), {%};
- *EF*: emission factor at the housing stage, $\{\text{kg} \cdot \text{head}^{-1} \cdot \text{year}^{-1}\}$;
- $PA_{1,\ 2,\ 3,\ 4}$: probability that a case of emission reduction technique application occurs, Eq. (1.1), Eq. (1.2) Eq. (1.3), Eq. (1.4), {%};
- $Re_{F\&H,\ AC}$: removal efficiency of different techniques (two techniques: *feeding and housing management* [F&H], *air cleaning* [AC]), {%}.

2.3. Human health impact assessment

Human health represents an endpoint environmental impact indicator.[3] In fact, midpoint impact indicators can be useful for identifying reduction targets and measures for specific environmental concerns, but often they cannot be easily understood or even show contradictory trends across different categories. For this reason, endpoint results represent a more direct and clearer tool for decision making, if supported by relevant and transparent information (Kägi et al., 2016).

The disability-adjusted life years (DALY) concept was adopted to quantify the human health impact. This metric is used by the WHO to account the overall burden associated with health problems. One DALY represent the loss of one healthy year and is calculated as the sum of the years of life lost (YLL) due to premature mortality and the years lost due to disability (YLD) (WHO, 2008). In the context of the present study, the DALY indicator is meant to be a measurement of the gap between the current health status (in CS) related to emissions from the housing stage of intensive pig farming and an improved health situation achievable with the large-scale implementation of the WAS technology (in AS1 and AS2). It is necessary to consider that the level of detail remains approximate in spatial terms, given the complexity of accounting for human health and some variability depending on the location, the pollution source and the target population involved, together with numerous other factors. However, this study aims to quantify the extent of the overall impact that large-scale adoption of WAS technology could have, rather than measuring the variation of human health precisely in geographical terms within the EU.

The inventory data to carry out the assessment included emission from housing facilities of the concerned pig population, adjusted for each scenario according to *Eq. (2)*. In AS1 and AS2, the consumable inputs necessary for the WAS operation were considered. As for electricity, a consumption of 10.3 kWh $\cdot$ kg of treated $NH_3{-}N^{-1}$ was considered, according to De Vries and Melse (2017). Other inputs considered were water (250 L kg of treated $NH_3{-}N^{-1}$, according to De Vries and Melse, 2017) and acid chemicals (1.5 L of sulfuric acid (H_2SO_4) $\cdot$ kg of removed NH_3^{-1}, according to Sigurnjak et al., 2019). Impacts related to capital goods (production, use, depreciation and final disposal of materials that make up the WAS machine) were excluded due to lack of information, however considering their human health impact negligible compared to that of operational consumable inputs over multiple years lifespan (Li et al., 2019). The discharge water produced by the WAS operation can be viewed as an effluent to be valorized through the agronomic exploitation of its nutrients. This could lead to the replacement of considerable amounts of nitrogen fertilizer. In AS1 and AS2 the environmental credit given by the avoidance of synthetic fertilizer production has therefore been included, assuming to replace a nitrogen dose equal to the ammoniacal nitrogen captured by means of WAS operation. Urea has been used as a replaced fertilizer, given its widespread use on a European scale. All the outputs (emissions) and inputs (both consumed and avoided) have been considered for each scenario and the overall human health impact was derived from their combination. Background data relating to all inputs were taken from the Ecoinvent® database v3.5 (Weidema et al., 2013). Table S3 reports the list of different Ecoinvent® processes used.

The characterization factors of environmental impacts (i.e. correlations between emitted/avoided pollutants and DALY) were

[3] Midpoint environmental impact categories are indicators (e.i. climate change, particulate matter formation, ozone depletion, etc.) that convert the emission of substances to the environment and/or the resource scarcity into a series of potential impacts in the middle of environmental cause-effect chain, rather than expressing the actual damage level. Endpoint indicators, on the other hand, reflect the midpoint impact categories at a further level of the cause-effect chain, associating them with different stressors and pathways into three areas of protection (human health, ecosystem quality and resource scarcity) which represent the main environmental concerns at the human society level.

obtained from the established ReCiPe method (v 1.13/Europe, H/A) (Goedkoop et al., 2009). The assessment was performed by using SimaPro software v.8.5 (Pré-Sustainability, 2018).

2.4. Environmental costs assessment

Environmental costs, even referred to as *external, shadow* or *damage costs*, arise when the production or consumption of a good or service imposes, due to additional amounts of pollutants emitted to the environment, one or more negative effects on a third party (Allacker and de Nocker, 2012). Environmental prices proposed by the CE Delft EU-28 Environmental Prices Handbook (De Bruyn et al., 2018a) were used in this study. These are indicators of the social marginal value of preventing emissions, coming express in € per kilogram pollutant (De Bruyn et al., 2018a). The Handbook reports monetary values referring to 2015 for the loss of welfare in EU-28 due to environmental pollution: relationship between emissions and endpoint impacts are built, for each pollutant, on concentration-response functions for human health, ecosystem services, damage to buildings/materials, resource availability and nuisance (De Bruyn et al., 2018a). The environmental prices for pollutants concerned in this study are shown in Table 5. These were applied to the emission inventory adjusted for each scenario, according to *Eq (2)*. It should be noted that these values refer to 2015, therefore they may have undergone some changes over the years due to inflation, variations in emissions trends and/or in the value attributed by people to environmental goods or ecoservices (since some prices are determined by contingent valuation methods). However, in the present study the conservative approach of assuming that these prices remain constant over time was adopted, as suggested by the Handbook.

3. Results

In the current scenario (CS), aimed at representing the real situation in the EU for 2018, emissions of the concerned pollutants from intensive pig housing facilities (farms with more than 1000 heads of sows or fattening pigs) account for 212.2 Gg of NH_3, 9.3 Gg of PM10 and 132.8 Gg of NMVOC (Table 6). These values respectively represent 6%, 11% and 15% of total agricultural emissions of the relative pollutants reported by the EEA for 2017 (EEA, 2019a). Still considering the EEA reference, NH_3 emissions in CS represent 43% of the total from the swine sector manure management in the EU. Indirect N_2O emissions account instead for 2.75 Gg, equal to 728.8 Gg of CO_2 eq, according to the characterization factor proposed by the Fifth Assessment Report of IPCC (IPCC, 2013).

In the alternative scenarios (AS1 and AS2), great emission reductions compared to CS are obtained. NH_3, being the pollutant on which the WAS expresses the highest removal efficiency, is the one that faces the most significant reductions, of 17% and 45% respectively for AS1 and AS2. The capture of a large quantity of ammonia also leads to the avoidance of the production of significant amounts of synthetic nitrogen fertilizer (64.0 and 169.2 Gg of urea per year, respectively for AS1 and AS2) that would be necessary to provide for the same nitrogen dose. On the other hand, the consumption of inputs necessary for the WAS operation is considerable.

Table 5
Environmental prices for atmospheric pollutants considered for the assessment, expressed in $€_{2015}$/kg. Source: De Bruyn et al. (2018a).

Pollutant	Lower value[b]	Central value[b]	Upper value[b]
Ammonia[a]	10.0	17.5	25.2
Particulates, < 10 µm	19.0	26.6	41.0
NMVOC	0.84	1.15	1.84
Dinitrogen monoxide	5.78	15.0	25.0

[a] Consistent with the values previously reported by Brink and van Grinvsen (2011), which identified an average price of 14 € (but in a wider range of 4–30 €) per kg NH_3–N emitted to the environment in the EU.
[b] Central value is calculated according to standard economic principles and is the one recommended for most applications. However, lower and upper values express thresholds given by the uncertainties in people's assessment of environmental quality and have been reported to reflect the intrinsic variability of environmental prices.

3.1. Human health impact

The estimated emissions for CS translate into an annual human health impact equal to 21,212 DALY. These are mostly (95%) a consequence of particulate matter formation, which in turn is primarily due to ammonia emissions (88%) and, to a lesser extent, to PM10 direct emissions (12%). The remaining DALY portion (5%) is instead a consequence of climate change through indirect N_2O emissions. NMVOC emissions are not included in the DALY evaluation neither in CS nor in the alternative scenarios due to data limitation, as the ReCiPe LCIA method (Goedkoop et al., 2009) provides characterization factors for individual compounds but not for unspecified NMVOC. The total DALY in the alternative scenarios are gradually reduced with the increase in the WAS implementation rate. In particular, the human health impact is reduced to 18,007 DALY (−15%) for AS1 and 12,730 DALY (−40%) for AS2. Fig. 2 shows the DALY variation for AS1 and AS2 compared to CS, divided by different contributors. The increase in the WAS implementation in the alternative scenarios leads to a growing consumption of inputs necessary for their operation, which implies a positive DALY variation (+386 for AS1 and +1021 for AS2). In particular, the positive variation due to the consumables in both AS1 and AS2 is given mainly by electricity consumption (81%), followed by acid chemicals (17%) and water (2%). However, the trade-off due to consumables is largely overwhelmed by the DALY values negative variations given by emission reduction. The reduction of ammonia emission is the one that most contributes to mitigation, representing alone 87% of the DALY negative variation given by overall emissions reduction in both AS1 and AS2. The results also show that avoiding the production of synthetic fertilizers contributes to further reducing the DALY in the alternative scenarios, as a consequence of their production being highly energy consuming.

3.2. Environmental costs results

The overall annual environmental cost given by the sum of the individual emissions of the current scenario (CS) turns out to be 4154 million $€_{2015}$ (range of 2426–6041). Considering the EU population for the same year (i.e. 512 million inhabitants, Eurostat, 2020a), these environmental costs lead to an average annual social weight of about 8 $€_{2015}$ per capita. NH_3 is the primary cause of this, accounting for 3714 million $€_{2015}$ (range of 2122–5347), or about 89% of the total. This result depends both on the large amount of NH_3 emitted, compared to PM10 and indirect N_2O, and on its relatively high environmental price per kg, compared to NMVOC, mostly as a result of increased morbidity and mortality associated with increasing PM2.5 formation (De Bruyn et al., 2018a). Fig. 3 shows the environmental costs save for AS1 and AS2 compared to the current scenario as a result of reduced emissions of NH_3 (Fig. 3a) and PM10, NMVOC and indirect N_2O (Fig. 3b) by means of WAS implementation.

In AS1 and AS2 can be saved respectively 668 million $€_{2015}$ (range of 386–968) and 1765 million $€_{2015}$ (range of 1019–2557) per year related to the effects of the overall emissions. Despite a wide variability given by the uncertainty of environmental prices of pollutants, these reductions in environmental costs are still

Table 6
Resulting air pollutants emission from intensive EU pig housing facilities (farms with more than 1000 heads of sows or fattening pigs) in the three scenarios; consumable inputs necessary for the WAS implementation (AS1 and AS2); amount of fertilizer avoided by recovering and valorizing the discharge water as nitrogen fertilizer (AS1 and AS2).

Item		Unit of measure	CS	AS1	AS2
Pollutant	Ammonia (NH_3)	Gg · year^{-1}	212.2	176.5	117.7
	Particulate matter (PM10)	Gg · year^{-1}	9.27	8.34	6.82
	Non-methane volatile organic compounds (NMVOC)	Gg · year^{-1}	132.8	123.6	108.8
	Dinitrogen monoxide (N_2O) − indirect, from NH_3	Gg · year^{-1}	2.75	2.29	1.53
Consumables for WAS operation	Electricity	GWh· year^{-1}	–	379.1	1002.8
	Water	dam^3· year^{-1}	–	9197.4	24,326.8
	Acid chemicals	dam^3 · year^{-1}	–	53.6	141.8
Avoided synthetic nitrogen fertilizer production	Urea	Gg · year^{-1}	–	64.0	169.2

significant quantities, which can contribute to improving the influence of livestock farming on EU social well-being. Both in AS1 and AS2, NH_3 emission reduction is responsible for 94% of the cost reduction compared to the CS, which again highlights the role of primary importance of this pollutant, and consequently the need to constantly improve the control of its emission.

3.3. Sensitivity analysis

To test the robustness of the results, a sensitivity analysis was carried out by changing key variables of the scenario modeling. The first change was made to the implementation rate of *feeding and housing management* techniques, which had been assumed to be 50%, fixed for the three scenarios. Results variation was arbitrary explored for 25% (low) and 75% (high) implementation rates of these techniques. The second change regarded the removal efficiency of the *air cleaning* technique, that have been tested for removal variations in different performance conditions. The achievable reductions were therefore varied considering 70% for NH_3, 40% for PM10 and 30% for NMVOC in low performance conditions and 90% for NH_3, 60% for PM10 and 40% for NMVOC in high performance conditions. In each analysis performed, indirect N_2O emissions, inputs consumed for WAS operation and avoided nitrogen fertilizer production were modified accordingly. The setting of the analysis has been reported in detail in Table S4, while the results are shown in Tables S5 and S6.

Despite the wide variability tested (±50% of the baseline value) for the implementation rate of *feeding and housing management* techniques, the absolute values (both in terms of DALY and environmental costs) undergo a limited change (constant across scenarios) of ±8.6% compared to the values of the baseline scenarios for CS, AS1 and AS2. Even regards the removal efficiency of the scrubber, there is a reduced variation in the results under the different tested performance conditions. In this case, however, the variability compared to the baseline scenarios gradually widens as the implementation rate of the *air cleaning* technique increases, going from ±1.9% for CS to ±4.8% for AS1, finally reaching ±12.7% for AS2.

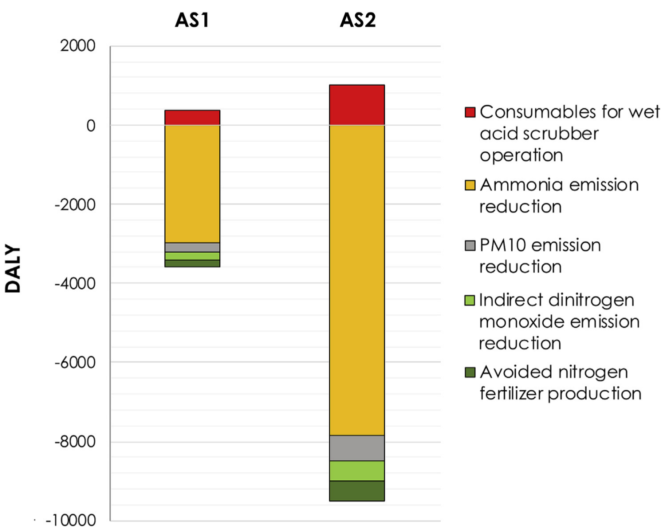

Fig. 2. Variation for AS1 and AS2 in the human health endpoint impact, expressed as disability-adjusted life year (DALY), compared to CS, divided by contributors. The consumables show positive values because compared to CS their increased consumption represents an additional environmental burden, while the emissions reduction and the avoidance of fertilizer production are negative because they involve environmental credits compared to CS.

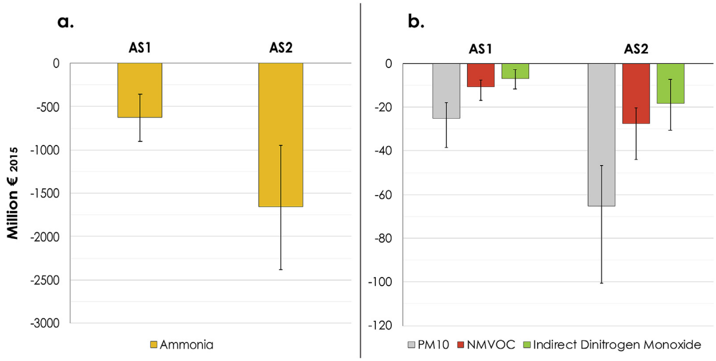

Fig. 3. Environmental costs save for AS1 and AS2, expressed as million €$_{2015}$ reduction with respect to CS. The graph has been split because of the different order of magnitude of the environmental cost saving between ammonia and the other pollutants. The error bars refer to the variability given by the upper and lower thresholds of environmental prices as shown in Table 5.

4. Discussion

The consequences of the large-scale implementation in the EU pig sector of the WAS go far beyond the farms' boundaries, leading to net positive environmental and economic endpoint effects on the impact of intensive pig farming.

In AS1 a great reduction on human health impact and environmental costs is achieved with the increased WAS implementation rate of all Member States at the current level of implementation of the north-continental countries. WAS implementation should therefore be a target primarily for those countries where it is currently under-used. In AS2 there is an even more significant reduction, more than double compared to AS1. The north-continental countries are in fact major players in the EU swine market and host alone 43% of sows and 37% of fattening pigs on the respective total EU populations hosted in large farms (>1000 heads of the same category) (elab. on Eurostat, 2020a), therefore an increased implementation rate even in these countries boosts the reduction of the human health impact and environmental costs.

Reductions obtained in this study could be further accentuated by means of future improvements in the WAS operation, so as to increase its removal efficiencies and minimize the consumption of inputs. For instance, the coupling of this machine with microclimatic smart tools that activate its operation only when the air pollutants concentration exceed fixed thresholds can be a way of reducing electricity consumption, which emerged as the main contributor to the trade-off consumables impact in the alternative scenarios. Electricity itself in a long-term perspective is destined to weigh less and less from an environmental point of view on the performance of the WAS, since the EU aims to constantly increase the energy mix share deriving from renewable sources (Ingrao et al., 2018).

As shown in Table 1, the relatively high implementation and running costs currently represent the main obstacle to the widespread application of WAS technology in pig farms in the EU. However, its diffusion in north-continental countries proves that this technique is actually economically viable in intensive livestock systems (Melse et al., 2009). Pexas et al. (2020b) recently performed a comparative cost-effectiveness analysis of several abatement measures to mitigate, among others, ammonia emissions from pig housing, but did not include any air cleaning technology. Future studies will have to deepen the costs of air scrubbing to identify ways of making its performance fully sustainable even from an economic point of view.

Furthermore, the relationship between a better environmental condition inside the pig facilities given by air scrubbing and a possible improvement in animal productive efficiency (e.g., better feed conversion rate) and welfare have never been considered in literature. An improvement in animal welfare could enhance the fattening and reproductive performances and the slaughtering yield, thus bringing direct economic benefits to farmers. The working and health conditions of agricultural operators directly involved in pig farming are also likely to be improved thanks to a better environment inside the animal's housing facilities. All these factors could be determinant for the decision-making of farmers towards the implementation of air scrubbers and need further future study.

The reuse of discharge solution from WAS as fertilizer is also a factor that can influence the farmers decision towards the implementation of this technology, allowing to reduce synthetic fertilizers costs for European mixed crop-livestock systems. However, discharge water from air cleaning technologies is still defined as 'livestock manure' in the EU legislation in force (EC, 1991). Therefore, this product falls within the application limits at a maximum rate of 170 kg N $\cdot$ ha^{-1} in Nitrate Vulnerable Zones, leading it to compete with "real" manure, thus limiting the adoption of the WAS technology in these areas due to the lack of benefit for farmers from this point of view. This contributes to the paradox that nutrient surplus regions are also among the largest consumers of synthetic fertilizers for meeting crop requirements (Sigurnjak et al., 2019). Currently, research on behalf of the EC is being carried out to promote the sustainable recovery of nutrients from manure which could possibly solve this issue (Huygens et al., 2019), favoring a growing implementation of WAS technology in the near future.

Pig meat production in EU-28 accounted for 23.8 Mt of carcass weight in 2018, or 49.8% of the total meat production. In the same year, the output value at basic prices of the pig sector was an estimated 36,300 million €, or approximately 21% of the output from all animal products and 8.3% of the total agricultural output (Eurostat, 2020a). Hence, this sector plays an important role in the agricultural economy of the EU, but it is necessary to look for an increasingly sustainable production that also contains the environmental costs associated with it. As for the pig housing phase, this study has shown that the WAS large-scale implementation appears to be a viable option for significantly alleviate the huge environmental costs of air pollutant emissions. The question remains on how to internalize these costs in the production chain. Environmental management strategies (in this case, the installation of WAS technology) entail costs and farmers may generally find it difficult to bear their full economic weight by aggravating existing production costs. On the other hand, Nguyen et al. (2012) estimated that the load of environmental costs on the market price of pork would lead it to at least double its value. De Bruyn et al. (2018b) instead made a smaller estimate according to which the pork market price would increase by about 50%. In any case, the hypothesis of fully charging the environmental shadow cost of pork production to the final consumer is unlikely to happen as the price is a key factor in the food choice and the attitude of most consumers already undergoes substantial variations for food taxes or subsidies ranging from 10% to 20% (Thow et al., 2014). Nevertheless, consumers have a primary role in making food chains more sustainable (Grunert, 2011). While a recent study has shown that EU consumers are not willing to pay for improving pig welfare beyond the medium level (Denver et al., 2017), they have recently been increasingly interested in promoting environmental sustainability in the agri-food sector (EC, 2018). At present there is still a gap between the positive attitude towards this concept and the market everyday behavior (Rejman et al., 2019). However, if appropriately encouraged by a targeted product positioning strategy, EU consumers may have a greater propensity to purchase environmentally sustainable pork, knowing that this would bring benefits for society as a whole, in terms of human health. For this reason, future studies could explore the willingness to pay of European consumers for this type of product to verify whether, at least partially, the environmental costs can be met by consumers.

This study was focused on the intensive pig farming, but the same method and considerations could be extended to poultry housing facilities. In fact, the WAS technology has been proven to be applicable even for the poultry sector with good performances (Van der Heyden et al., 2015).

Finally, carrying out this analysis highlighted the current lack of detailed data that cover livestock systems in the EU by type of feeding, housing and manure management. There is a future need for improved information in these areas, because they are increasingly crucial for an accurate estimate of emissions, which in turn can influence mitigation strategies and policies.

5. Conclusions

Large pig farms (>1000 heads of sows or fattening pigs) host the majority of pig population in the EU and are responsible for significant air pollutant emissions, a considerable part of which occurs at the housing stage. End-of-pipe air cleaning techniques are among the possible measures to control and reduce these emissions. However, they are currently little adopted on a European scale, despite their removal efficiency have already been proven be great, in particular with regard to ammonia.

This study explored the emission reduction achievable with increased implementation rates of the wet acid scrubber technology in intensive pig farms across the EU, demonstrating that it would bring a largely positive endpoint effect on human health, and also lead to significant alleviation of current environmental costs on society of air pollution related to intensive pig farming.

Further assessments are to be done to better investigate various issues regarding the wet acid scrubber, including cost-effectiveness, influence on animal welfare and production performance, impact on working conditions of agricultural operators and discharge water management. Consumer behavior towards a more sustainable pig production is also a study field to be deepened in the future. Nonetheless, what emerged clearly is that there is vast room for improve the environmental sustainability of intensive pig farming at the housing stage and the use of the wet acid scrubber can push strongly in this direction. Therefore, in our vision, its implementation should be increasingly encouraged by EU and/or national policies, especially in countries other than north-continental ones, where its use is currently uncommon.

CRediT authorship contribution statement

Michele Costantini: Conceptualization, Conception and design of study, Funding acquisition, acquisition of data, Formal analysis, analysis and/or interpretation of data, Writing - original draft, Drafting the manuscript, On behalf of. **Jacopo Bacenetti:** Conceptualization, Conception and design of study, Formal analysis, analysis and/or interpretation of data, Writing - original draft, Drafting the manuscript, On behalf of. **Giuseppe Coppola:** Funding acquisition, acquisition of data, Formal analysis, analysis and/or interpretation of data, On behalf of. **Luigi Orsi:** Writing - original draft, Drafting the manuscript, Writing - review & editing, revising the manuscript critically for important intellectual content: On behalf of. **Andrea Ganzaroli:** Writing - review & editing, revising the manuscript critically for important intellectual content: On behalf of. **Marcella Guarino:** Conceptualization, Conception and design of study, Writing - review & editing, revising the manuscript critically for important intellectual content: On behalf of.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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References

Aarras, N., Rönkä, M., Kamppinen, M., Tolvanen, H., Vihervaara, P., 2014. Environmental technology and regional sustainability – the role of life-based design. Technol. Soc. 36, 52–59. https://doi.org/10.1016/j.techsoc.2013.12.003.

Allacker, K., de Nocker, L., 2012. Approach for calculating the environmental external costs of the Belgian building sector. J. Ind. Ecol. 16 (Issue 5), 710–721. https://doi.org/10.1111/j.1530-9290.2011.00456.x.

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. Atmos. Environ. 126, 153–161. https://doi.org/10.1016/j.atmosenv.2015.11.039.

Berckmans, D., 2014. Precision livestock farming technologies for welfare management in intensive livestock systems. Rev Sci Tech 33, 189–196. https://doi.org/10.20506/rst.33.1.2273.

Blonk Consultants, 2019. Agri-footprint 5.0 - methodology report – Part 2: description of data (available online at: https://www.agri-footprint.com/wp-content/uploads/2019/11/Agri-Footprint-5.0-Part-2-Description-of-data-17-7-2019-for-web.pdf. accessed on April 2020).

Brink, C., van Grinsven, H., et al., 2011. Costs and benefits of nitrogen in the environment, Chapter 22 (513-540. In: Sutton, M.A., et al. (Eds.), The European Nitrogen Assessment. Cambridge University Press.

Cambra-Lopez, M., Aarnink, A.J.A., Zhao, Y., Calvet, S., 2010. Airborne particulate matter from livestock production systems: a review of an air pollution problem. Environ. Pollut. 158 (Issue 1), 1–17. https://doi.org/10.1016/j.envpol.2009.07.011.

De Bruyn, S., Bijleveld, M., de Graaff, L., Schep, E., Schroten, A., Vergeer, R., Ahdour, S., 2018a. Environmental Prices Handbook – EU28 Version. Publication Code: 18.7N54.125. CE Delft, Delft.

De Bruyn, S., Warringa, G., Odegard, I., 2018b. *The True Price of Meat* (In Dutch). Publication Code: 18.7N81.009. CE Delft, Delft.

De Vries, J.W., Melse, R.W., 2017. Comparing environmental impact of air scrubbers for ammonia abatement at pig houses: a life cycle assessment. Biosystem engineering 161, 53–61. https://doi.org/10.1016/j.biosystemseng.2017.06.010.

Denver, S., Sandøe, P., Christensen, T., 2017. Consumer preferences for pig welfare – can the market accommodate more than one level of welfare pork? Meat Sci. 129, 140–146. https://doi.org/10.1016/j.meatsci.2017.02.018.

Dumont, É., 2018. Impact of the treatment of NH_3 emissions from pig farms on greenhouse gas emissions. Quantitative assessment from the literature data. New BIOTECHNOLOGY 46, 31–37. https://doi.org/10.1016/j.nbt.2018.06.001.

Ec (European Commission), 1991. Directive of the council of 12 December 1991 concerning the protection of waters against pollution cause by nitrates from agricultural sources (91/676/EC). Official Journal L304, 1–194.

Ec (European Commission), 2017. Corrigendum to Commission Implementing Decision (EU) 2017/302 of 15 February 2017 establishing best available techniques (BAT) conclusions, under Directive 2010/75/EU of the European Parliament and of the Council, for the intensive rearing of poultry or pigs (OJ L43, 21.2.2017). Official Journal L105, 21–21.

Ec (European Commission), 2018. Europeans, Agriculture and the CAP - Summary. Special Eurobarometer 473 – Wave EB88.4 – TNS Opinion & Social.

Ec (European Commission), 2019. The European Green Deal. Communication from the Commission to the European Parliament. the European Council, the Council, the European economic and social Committee and the Committee of the Regions. COM/2019/640 final.

Ec (European Commission), 2020. Farm to Fork Strategy – for a fair, healthy and environmentally-friendly food system (available online at: https://ec.europa.eu/food/farm2fork_en. accessed on May 2020).

EEA (European Environmental Agency), 2019a. European Union Emission Inventory Report 1990-2017, under the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP). EEA Report | No 08/2019.

EEA (European Environmental Agency), 2019b. EMEP/EEA Air Pollutant Emission Inventory Guidebook 2019, 3. B - Manure management.

Eurostat, 2020a. European Commission online statistical database (available online at: https://ec.europa.eu/eurostat/data/database. accessed on March 2020).

Eurostat, 2020b. Eurostat statistics explained. Archive: agri-environmental indicator – animal housing (available online at: https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Archive:Agri-environmental_indicator_-_animal_housing. accessed on April 2020).

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. Report I: Characterisation, first ed.

Grunert, K.G., 2011. Sustainability in the food sector: a consumer behaviour perspective. Int. J. Food Syst. Dynam. 2 (3), 207–218. https://doi.org/10.18461/ijfsd.v2i3.232.

Huygens, D., Orveillon, G., Lugato, E., Tavazzi, S., Comero, S., Jones, A., Gawlik, B., Saveyn, H., 2019. SAFEMANURE – developing criteria for safe use of processed manure in nitrates vulnerable Zones above the threshold established by the nitrates directive – interim report. European Commission, DG Joint Research Centre (JRC). https://phosphorusplatform.eu/images/download/JRC%20Safemanure%20DRAFT%20Interim%20Report%20Sept%202019.pdf.

Ingrao, C., Bacenetti, J., Bezama, A., Blok, V., Goglio, G., Koukios, E.G., Lindner, M., Nemecek, T., Siracusa, V., Zabaniotou, A., Huisingh, D., 2018. The potential roles of bio-economy in the transition to equitable, sustainable, post fossil-carbon societies: findings from this virtual special issue. J. Clean. Prod. 204, 471–488. https://doi.org/10.1016/j.jclepro.2018.09.068.

IPCC (Intergovernmental Panel on Climate Change), 2013. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), Climate Change 2013: the Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, p. 1535.

IPCC (Intergovernmental Panel on Climate Change), 2019. 2019 refinement to the 2006 guidelines for national greenhouse gas inventories. Agriculture, Forestry and other land use. Chapter 10: Emissions from livestock and manure management 4.

Jacobsen, B.H., Latacz-Lohmann, U., Luesink, H., Michels, R., Ståhl, L., 2019. Cost of regulating ammonia emissions from livestock farms near Natura 2000 area – analysis of case farms from Germany, Netherlands and Denmark. J. Environ. Manag. 246, 897–908. https://doi.org/10.1016/j.jenvman.2019.05.106.

Kägi, T., Dinkel, F., Frischknecht, R., Humbert, S., Lindberg, J., De Mester, S., Ponsionen, T., Sala, S., Schenker, U.W., 2016. Session "midpoint, endpoint or single score for decision-making?"-SETAC Europe 25th annual meeting, may 5th, 2015. Int. J. Life Cycle Assess. 21, 129–132. https://doi.org/10.1007/s11367-015-0998-0.

Klimont, Z., Winiwarter, W., 2011. Integrated Ammonia Abatement – Modelling of Emission Control Potentials and Costs in GAINS. IIASA Interim Report IR-11-027. Laxenburg, Austria.

Li, S., Subbiah, J., Dvorak, B., 2019. Environmental and occupational impacts from U.S. beef slaughtering are of same magnitude of beef foodborne illnesses on human health. Environ. Int. 129, 507–516. https://doi.org/10.1016/j.envint.2019.05.051.

Lovarelli, D., Bacenetti, J., 2019. Exhaust gases emissions from agricultural tractors: state of the art and future perspectives for machinery operators. Biosyst. Eng. 186, 204–213. https://doi.org/10.1016/j.biosystemseng.2019.07.011.

Lovarelli, D., Bacenetti, J., Guarino, M., 2020a. A review on dairy cattle farming: is precision livestock farming the compromise for an environmental, economic and social sustainable production? J. Clean. Prod. 262, 121409. https://doi.org/10.1016/j.jclepro.2020.121409.

Lovarelli, D., Conti, C., Finzi, A., Bacenetti, J., Guarino, M., 2020b. Describing the trend of ammonia, particulate matter and nitrogen oxides: the role of livestock activities in northern Italy during Covid-19 quarantine. Environ. Res. 191, 110048.

Martin, J.W., Moore, P.A., Li, H., Ashworth, A.J., Miles, D.M., 2018. Effects of land-applied ammonia scrubber solutions on yield, nitrogen uptake, soil test phosphorus, and phosphorus runoff. J. Environ. Qual. 47, 263–269. https://doi.org/10.2134/jeq2017.09.0383.

Melse, R.W., Ogink, N.W.M., Rulkens, W.H., 2008. Multi-pollutant scrubbers for removal of ammonia, odor, and particulate matter from animal house exhaust air. In: Mitigating Air Emissions from Animal Feeding Operations - Conference Proceedings, May 19-21, 2008. Des Moines, Iowa, pp. 162–168.

Melse, R.W., Ogink, N.W.M., Rulkens, W.H., 2009. Overview of European and Netherlands' regulations on airborne emissions from intensive livestock production with a focus on the application of air scrubbers. Biosyst. Eng. 104, 289–298. https://doi.org/10.1016/j.biosystemseng.2009.07.009.

Mostafa, E., Selders, A., Gates, R.S., Buescher, W., 2020. Pig barns ammonia and greenhouse gas emission mitigation by slurry aeration and acid scrubber. Environ. Sci. Pollut. Control Ser. 27, 9444–9453. https://doi.org/10.1007/s11356-020-07613-x.

Nguyen, T.L.T., Hermansen, J.E., Mogensen, L., 2012. Environmental costs of meat production: the case of typical EU pork production. J. Clean. Prod. 28, 168–176. https://doi.org/10.1016/j.jclepro.2011.08.018.

Ni, J.-Q., Robarge, W.P., Xiao, C., Heber, A.J., 2012. Volatile organic compounds at swine facilities: a critical review. Chemosphere 89, 769–788. https://doi.org/10.1016/j.chemosphere.2012.04.061.

Pexas, G., Mackenzie, S.G., Wallace, M., Kyriazakis, I., 2020a. Environmental impacts of housing conditions and manure management in European pig production systems through a life cycle perspective: a case study in Denmark. J. Clean. Prod. 253, 120005. https://doi.org/10.1016/j.jclepro.2020.120005.

Pexas, G., Mackenzie, S.G., Wallace, M., Kyriazakis, I., 2020b. Cost-effectiveness of environmental impact abatement measures in a European pig production system. Agric. Syst. 182, 102843. https://doi.org/10.1016/j.agsy.2020.102843.

Pré-Sustainability, 2018. SimaPro LCA software v. 8.5. Accessed May 2020. https://simapro.com/.

Rebolledo, B., Gil, A., Pallarés, J., 2013. A spatial ammonia emission inventory for pig farming. Atmos. Environ. 64, 125–131. https://doi.org/10.1016/j.atmosenv.2012.10.005.

Reis, S., Sutton, M.A., Howard, C., 2015. Overview, aims and scope. In: Reis, S., et al. (Eds.), Cost of Ammonia Abatement and the Climate Co-benefits. Springer Science+Business Media Dordrecht, pp. 1–6. https://doi.org/10.1007/978-94-017-9722-1_1.

Rejman, K., Kaczorowska, J., Halicka, E., Laskowski, W., 2019. Do Europeans consider sustainability when making food choices? A survey of Polish city-dwellers. Publ. Health Nutr. 22 (7), 1330–1339. https://doi.org/10.1017/S1368980019000326.

Santonja, G.G., Georgitzikis, K., Scalet, B.M., Montobbio, P., Roudier, S., Sancho, L.D., 2017. Best available techniques (BAT) reference document for the intensive rearing of poultry or pigs. EUR 28674 EN. https://doi.org/10.2760/020485.

Sigurnjak, I., Brienza, C., Snauwaert, E., De Dobbelaere, A., De Mey, J., Vaneeckhaute, C., Michels, E., Schoumans, O., Adani, F., Meers, E., 2019. Production and performance of bio-based mineral fertilizers from agricultural waste using ammonia (stripping-)scrubbing technology. Waste Manag. 89, 265–274. https://doi.org/10.1016/j.wasman.2019.03.043.

Thow, A.M., Downs, S., Jan, S., 2014. A systematic review of the effectiveness of food taxes and subsidies to improve diets: understanding the recent evidence. Nutr. Rev. 72 (9), 551–565. https://doi.org/10.1111/nure.12123.

Tullo, E., Finzi, A., Guarino, M., 2019. Review: environmental impact of livestock farming and Precision Livestock Farming as a mitigation strategy. Sci. Total Environ. 650, 2751–2760. https://doi.org/10.1016/j.scitotenv.2018.10.018.

UN (United Nations), 2020. Status of Treaties. Website of United Nations Treaty Collection. Chapter XXVII (Environment), Sections 1., 1.h., 1.K (available at: https://treaties.un.org/. accessed on April 2020).

Van der Heyden, C., Demeyer, P., Volke, E.I.P., 2015. Mitigating emissions from pig and poultry housing facilities through air scrubbers and biofilters: state-of-the-art and perspectives. Biosyst. Eng. 134, 74–93. https://doi.org/10.1016/j.biosystemseng.2015.04.002.

Velthof, G.L., Hou, Y., Oenema, O., 2015. Nitrogen excretion factors of livestock in the European Union: a review. J. Sci. Food Agric. 95, 15. https://doi.org/10.1002/jsfa.7248.

Weidema, B.P., Bauer, C., Hischier, R., Mutel, C., Nemecek, T., Reinhard, J., Vadenbo, C.O., Wernet, G., 2013. Overview and Methodology. Data Quality

Guideline for the Ecoinvent Database Version 3. Ecoinvent Report 1(v3). The Ecoinvent Centre, St. Gallen.
WHO (World Health Organization), 2008. The Global Burden of Disease: 2004 Update.
Zhao, Z.Q., Bai, Z.Q., Winiwarter, W., Kiesewetter, G., Heyes, C., Ma, L., 2017. Mitigating ammonia emission from agriculture reduces PM2.5 pollution in the Hai River Basin in China. Sci. Total Environ. 609, 1152–1160. https://doi.org/10.1016/j.scitotenv.2017.07.240.

Supplementary materials

Table S1. Breeding pig (made up of breeding sows plus boars) population from which the emissions inventory was built.

Country	Total population (Thousands heads) [a]	Reared in holdings with > 1000 sows (%) [b]
Belgium	406.35	89 %
Bulgaria	71.11	79 %
Czechia	135.11	90 %
Denmark	1255.00	99 %
Germany	1854.90	79 %
Estonia	24.90	96 %
Ireland	142.99	98 %
Greece	96.20	58 %
Spain	2529.42	80 %
France	1026.00	86 %
Croatia	124.20	23 %
Italy	579.91	81 %
Cyprus	33.86	98 %
Latvia	33.06	79 %
Lithuania	45.60	92 %
Luxembourg	5.48	76 %
Hungary	261.00	78 %
Malta	3.85	39 %
Netherlands	974.00	98 %
Austria	232.71	20 %
Poland	758.30	30 %
Portugal	241.12	72 %
Romania	316.60	40 %
Slovenia	19.35	24 %
Slovakia	54.08	82 %
Finland	96.70	71 %
Sweden	131.83	86 %

United Kingdom[c]	502.00	80 %

Table S2. Fattening pig (> 30 kg) population from which the emissions inventory was built.

Country	Total population (Thousands heads) [a]	Reared in holdings with > 1000 fattening pigs (%) [b]
Belgium	4206.26	78 %
Bulgaria	430.97	85 %
Czechia	906.77	89 %
Denmark	6845.00	97 %
Germany	16918.90	69 %
Estonia	160.80	97 %
Ireland	992.30	97 %
Greece	420.90	81 %
Spain	19723.95	84 %
France	9337.00	77 %
Croatia	651.00	42 %
Italy	6505.10	90 %
Cyprus	199.87	99 %
Latvia	214.27	83 %
Lithuania	413.00	75 %
Luxembourg	59.90	90 %
Hungary	1923.00	76 %
Malta	23.86	42 %
Netherlands	5648.00	89 %
Austria	1891.12	16 %
Poland	7990.50	38 %
Portugal	1200.93	85 %
Romania	2910.30	47 %
Slovenia	181.77	18 %
Slovakia	403.08	85 %
Finland	671.20	63 %

Sweden	900.86	87 %
United Kingdom[c]	2930.00	79 %

[a] Source: Eurostat, 2020. Reference year: 2018.

[b] Source: elaboration on Eurostat, 2020. Reference year: 2016.

[c] Having been used as a reference year 2018, the United Kingdom was considered as a Member State.

Table S3. List of processes retrieved from the Ecoinvent database v. 3.5.

Ecoinvent® 3.5 process	**Used for**
Tap water {Europe without Switzerland}|market for| APOS, U	Water consumption for wet acid scrubber operation
Electricity, medium voltage {Europe without Switzerland}| market group for| APOS, U	Electricity consumption for wet acid scrubber operation
Sulfuric acid {GLO}| market for| APOS, U	Chemical acid consumption for wet acid scrubber operation
Urea, as N {RER}| production | APOS, U	Avoided production of mineral nitrogen fertilizer

Table S4. Results of the sensitivity analysis for human health impact.

Scenario	**Unit**	**Baseline**	**Sensitivity on implementation rate of feeding & housing techniques**		**Sensitivity on removal efficiency of wet acid scrubbers**	
			HIGH	LOW	HIGH	LOW
CS	DALY	21212	19380	23042	20807	21618
AS1	DALY	18007	16454	19559	17135	18869
AS2	DALY	12730	11636	13822	11103	14329

Table S5. Results of the sensitivity analysis for environmental costs. The analysis was carried out taking into consideration the central values of the environmental prices of the pollutants.

Scenario	**Unit**	**Baseline**	**Sensitivity on implementation rate of feeding & housing techniques**		**Sensitivity on removal efficiency of wet acid scrubbers**	
			HIGH	LOW	HIGH	LOW
CS	million €	4154.2	3796.7	4511.6	4075.5	4232.8
AS1	million €	3486.4	3187.8	3786.6	3322.1	3650.5
AS2	million €	2388.8	2185.1	2592.5	2084.1	2693.5

CHAPTER 4 – Livestock, climate change and agro-energy

An overview of the link between livestock productions and greenhouse gas emissions has already been presented in Chapter 1, with a focus on the Italian context.

The emission reduction measures examined in the previous chapter focus on interventions directly at the level of animal housing, where there is ample room for improvement for the pig sector. This type of intervention, on the other hand, it is more difficult to implement for intensive cattle stables, due to their constitution, therefore mitigation actions efforts to date are mostly directed to the various operations of manure and slurry collection, storage, processing and field distribution.

Historically, estimating livestock GHG emissions has been particularly challenging. First, because livestock emissions come from a variety of sources, including enteric fermentation (methane), manure management (methane and nitrous oxide), and energy and fuel use on livestock farms. The latter is the least complicated to consider, but it is also the one that typically has the least influence on the overall GHG of these supply chains. Instead, emissions from enteric fermentation and manure management suffer from very high variability due to animal-related factors such as diet, genetics, age, health, and environmental factors such as geographic location, climate, and season. In addition, direct measurement of livestock emissions requires specialized equipment and skills and can be time consuming and costly. As a result, GHG emissions from livestock are often estimated using inventories based on statistical data and emission factors. While this approach is practical and provides general indications, it may not capture all sources of emissions and may not reflect the actual emissions of individual farms due to the variability described above, as well as the difficulty often encountered in collecting data on livestock and their management practices. This variability makes it difficult to develop standardized emission factors, and inventories may not always be comprehensive and/or consistent in realistically representing livestock production scenarios in detail.

This chapter presents two studies focused on the research of mitigation techniques for the environmental impact of cattle farming in the Po Valley, one on the dairy and one on the beef sector. Both are focused on, but not limited to, the analysis of potential strategies for reducing greenhouse gas emissions resulting from the management of livestock effluents.

However, two different approaches are adopted among those described above: for the dairy sector, in fact, a direct measurement approach was adopted, through a campaign that included an in-farm trial lasting several months, adopting slurry treatments and specific technologies for on-site measurement of emitted gases.

For the beef cattle sector, on the other hand, to investigate the environmental impact of integrating on-farm renewable energy production facilities, a case study was developed to analyze a beef cattle farm by combining life cycle assessment and emissions modeling. This provides a broader view of the technical, production and environmental facets and implications of the mitigation techniques analyzed. In particular, the on-farm implementation of an anaerobic digestion plant from agricultural biomass and livestock waste to produce biogas and a photovoltaic plant for solar energy production on the roof of the barn were considered.

This last study is the only one among those presented within this thesis that is based on a widely studied and commercially widespread mitigation strategies. However, to the authors' knowledge, these technologies has never been explored in the context of the beef sector under an LCA study involving the whole farm. Therefore, it was decided to fill this gap in the literature through a case study that was precisely intended to quantify the environmental benefit possibly derived from such integrated management of livestock and energy production.

4.1. Real-scale study on methane and carbon dioxide emission reduction from dairy liquid manure with the commercial additive SOP LAGOON

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Presentation and context of the study

The study whose publication is reported below is the result of a project financed by the Lombardy Region within the framework of a call for research aimed at promoting collaboration between research institutions and small and medium-sized enterprises in the agri-food sector, with a particular focus on the sustainability of bio-resources.

The aim of the project was to evaluate the effectiveness of the SOP Lagoon Additive, manufactured by SOP, for the reduction of greenhouse gas emissions and ammonia associated with the storage of cattle slurry. This was done thanks to direct measurement tests of the emission flows coming from two tanks for the storage of cattle slurry (one treated and one control), compared during several days of measurements carried out approximately 30 days apart during an entire spring-summer season.

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Real-Scale Study on Methane and Carbon Dioxide Emission Reduction from Dairy Liquid Manure with the Commercial Additive SOP LAGOON

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Abstract: Reducing methane (CH_4) is a key objective to address climate change quickly. Manure management and storage play a significant role. In this context, a real-scale trial was performed to measure the ability of the commercial additive SOP LAGOON to reduce carbon-based greenhouse gas (GHG) emissions from liquid manure over approximately 4 months. Gas emissions were measured at a commercial dairy farm from two slurry tanks, one treated with the abovementioned product (SL) and the other used as the untreated control (UNT). After 3 and 4 months from the first additive applications, the SL storage tank showed lower and statistically significantly different emissions concerning the UNT (up to −80% for CH_4 and −75% for CO_2, $p < 0.001$), confirming and showing improved results from those reported in the previous small-scale works. The pH of the UNT tank was lower than that of the SL on two dates, while the other chemical characteristics of the slurry were not affected. In this work, SOP LAGOON proved to be an effective additive to help the farmers mitigate the contribution of stored liquid manure to global CH_4 emissions, potentially improving the overall sustainability of the dairy industry.

Keywords: methane; CO_2; climate change; manure; slurry; dairy; sustainability; mitigation; emissions

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1. Introduction

The Sixth Assessment Report of the Intergovernmental Panel on Climate Change [1] urges immediate action to slow warming in the near term. The most recent projections on climate foresee a 50:50 chance of a temperature rise over the threshold of 1.5 °C above pre-industrial levels within the next 5 years [2], especially if the emissions from the food system are not addressed [3]. In this light, the scientific community and the industry alike are focusing with increasing attention on the so-called short-lived climate pollutants (SLCP), such as methane (CH_4), as a key mitigation strategy [4] to limit the climate impact of human activities and avoid up to 0.6 °C of warming by 2050 [5]. The global warming potential (GWP) of CH_4 on a 100-year timescale is considered 28 times greater than carbon dioxide (CO_2) [6]. More recently, the new Global Warming Potential Stars (GWP*) was proposed, which recalculates the impact of CH_4, taking into account the shorter lifespan, a 20-year timescale, of this gas in the atmosphere [7] before it is converted to CO_2 [8] by a hydroxyl oxidation reaction. The proposed value for CH_4 in the GWP* model is 84 times that of CO_2, implying that these emissions have a more significant impact on the climate than previously estimated. Reducing the CH_4 emissions associated with human activity by 50% over the next 30 years could mitigate a global temperature change of 0.2 °C by 2050, a significant step towards keeping the temperature increase below 2 °C [6]. In this light, the European Commission and the US government launched a climate-related initiative, the Global Methane Pledge [9], at COP26 in Glasgow, inviting the joining countries to set national goals to reduce CH_4. The initiative now has 150 signatory countries after COP27, 50 more than when the initiative was launched [10].

Strategies to mitigate SLCP are therefore put in place to give a more decisive contribution to the national climate goals. For example, the California Senate Bill 1383 [11] has required the implementation of the SLCP strategy by 1 January 2018. The strategy includes a 40% CH_4 emission reduction from 2013 levels by 2030. The EU aims to become carbon neutral by 2050, with a 2030 milestone of reducing at least 55% of CH_4 emissions from the 1990 levels [12], with binding national emission reduction targets under the Effort Sharing Regulation (ESR). In December 2021, an amendment to this regulation was proposed [13] as part of implementing the increased emissions reductions target for 2030. The methane strategy identifies actions to accelerate the CH_4 emissions reduction in line with that ambition [14].

Agricultural activities contribute to global production and are estimated to account for about 12% of the total anthropogenic GHG emissions [15], 10% to 12% of CO_2 and 40% of CH_4 [16], globally. The most significant sources of CH_4 from agriculture are manure management (4%), rice cultivation (10%) and enteric fermentation in ruminants (29%) [17].

As reported, the dairy sector is a contributor to these emissions: while globally over 90% of CH_4 emissions in the dairy sector are connected to enteric fermentations [18], in concentrated animal feed operations (CAFOs), common in most of the developed countries, liquid manure plays an important role.

Amon et al. [19] reported that more than 90% of GHG emissions from slurry originate from CH_4 emissions during the storage phase. In Italy, in 2020, CH_4 emissions from the manure management from dairy cows were 920 kton of CO2eq [20], representing 14.2% of CH_4 emissions for the sector, while a recent study from the California Air Resources Board (CARB) indicates that 57% of CH_4 emissions from the dairy sector in California are attributed to manure management and 43% to enteric fermentation [21].

An increasing number of studies have investigated the ability of feed additives to reduce enteric emissions [22–25], although the timing for their broad application remains to be determined.

Different techniques have been developed for CH_4 emission abatement from liquid manure, such as solid–liquid separation, anaerobic digestion, slurry acidification, storage cover and slurry additives. Mosquera et al. [26] reported that liquid separation could reduce CH_4 emissions by up to 42% while [27] reported that anaerobic digestion reduced the emissions by up to 35% compared to raw manure. Misselbrook et al. [28] found that acidification reduced CH_4 emissions by 61% while Amon et al. [29], with storage cover, reported an abatement ability of up to 70%.

Besides CH_4, CO_2 emissions can be of interest in reducing the impact of slurry storage. Unfortunately, information about this gas and its reduction remains sparse.

In efforts to promote economic and environmental sustainability for dairy farms, slurry additives are considered with increasing interest, as they might represent a simple and economic way to address the GHG emissions from liquid manure.

The commercial additive for the liquid manure SOP LAGOON proved to be effective in reducing CH_4 (and CO_2) emissions from slurry in two lab-scale tests [30,31]. This work aims to investigate this product's ability to reduce carbon-based emissions (namely CH_4 and CO_2) on commercial-scale farms and to investigate other potential benefits on manure management.

2. Materials and Methods

2.1. Site and Manure Management Description

The trials campaign was carried out in 2021 at a dairy farm in the Po Valley, Northern Italy, characterized by humid continental to subtropical climates (Cfa following Köppen classification).

The farm operates a total herd of about 520 heads, half of which are lactating cows, and is representative of typical housing and farming practices found in concentrated animal feeding operations (CAFOs): the animals are housed in a free stall system with straw as topping for the bedding. Animal waste is mainly handled as slurry and is conveyed through scrapers and pumping systems into two adjacent, separate, concrete storage tanks.

During the experiment, the tanks were filled on alternate days with the same type of slurry. The manure was collected for distribution on the fields uniformly from both tanks, aiming at keeping the depth of the slurry in both tanks equal for the duration of the tests.

Both tanks were mixed the day before the measurement using a propeller mixer coupled to a tractor. This was done to break up the possible crust that could form on top of the tanks, which could prevent the release of gaseous emissions. Furthermore, these activities simulate how the farmer manages the tanks before emptying them. The filling level of the tanks was also measured on the day of the gas measurements. This allowed us to confirm that the ratio of emitting surface per volume present was similar between the two storage tanks on each testing day. On the first day of measurement, the surface per volume ratio was approximately 0.35 m^2/m^3 in both tanks; the following measurements registered higher ratios of between 0.4 and 0.5 m^2/m^3, as the tanks were partially, and always uniformly, emptied for field distribution.

SOP LAGOON, SQE034 + SQE610 (www.sopfarm.com, accessed on 1 December 2022), the additive under test, is made up of 100% calcium sulfate dihydrate (gypsum) processed with proprietary technology.

The product was added to one of the two tanks (SOP LAGOON: SL), while the other tank was left as the control (untreated: UNT).

The additive was administered according to the manufacturer's specifications provided in the technical data sheet of the product: the recommended application method consists of weekly applications of 2 g per animal contributing the slurry to the tank, with an additional dose in the first month for the activation period of 4 g/m^3 of slurry already stored in the tank at the time of the first addition. In these test conditions, a total of 40 kg of the product was added over the first 4 weeks, and a total of 11 kg was added from week 5 to the end of the experiment.

The first application of SOP LAGOON was done on 27 May.

2.2. Slurry Analysis

Slurry samples from different positions in the tanks were collected during each measurement day. The chemical composition of the slurry was analyzed to characterize the matrix and to verify any effects of the treatment. Samples were stored in small air-tight containers refrigerated at 4 °C prior to laboratory analysis.

Analyses of the samples for total solids (TS), volatile solids (VS), total Kjeldahl nitrogen (TKN), total ammoniacal nitrogen (TAN), pH and total organic carbon were performed according to standard methods for the examination of water and wastewater [32].

The results will be reported in Table 1 in the Section 3.

2.3. Emission and Fluxes Determination

The emissions of CH_4 and CO_2 originating from the two tanks were assessed throughout four surveys to analyze the entire duration of the slurry storage. The surveys were carried out approximately one month apart (14 days in the case of the last measurement), starting after the first month of activation, from June until September.

Following the reference VERA protocol [33] (Test Protocol Covers and other Mitigation Technologies for Stored Manure—Version 3: 2018-07), and considering the surface of the tanks, five measurement points were installed for each one tank. A floating funnel was positioned at each measuring point, from which the air was sampled.

The measurement system was conceived referring to previous studies on livestock waste emissions, in particular [34,35]. The whole system, shown schematically in Figure 1, consisted of:

- Floating PVC funnels, positioned on the slurry surface. The funnels have a diameter of 42 cm. Each funnel covers a surface equal to approximately 0.14 m^2, for a total covered area for each pit of approximately 0.7 m^2, thus greater than the 0.5 m^2 suggested by the aforementioned VERA protocol;
- PTFE tubes that connected each floating funnel with a corresponding "lung flask", hermetically sealed with rubber stoppers. Utilizing a vacuum generated by a pump positioned downstream of the system, the air was sucked by the multi-component gas analyzer from each lung flask, passing through a multipoint gas sampler. The use of external pumps was necessary to support the emissive flow towards the lung flasks, because the vacuum generated by the small pump bundled with the multi-component gas analyzer was insufficient to overcome the hydraulic resistance of the PTFE tubes (several tens of meters long);
- A 12-way multipoint gas sampler (MGS) to which the floating funnels in the two slurry tanks (five for the UNT and five for the SL) were connected via IN channels. An additional channel was used to sample the background air ("white") to check the atmospheric concentration of the gases under study. The multipoint gas sampler device allows the user to open one channel at a time, via solenoid valves, for a chosen time interval and to define the order of the opening and closing of the different channels. Finally, the MGS was connected via the OUT channel to the gas analyzer: a real time assessment of gases was done with a high-resolution spectrometer (ETG FTIR 9500, Chivasso, Italy) that exploits the Fourier transformed IR spectroscopy (FTIR) technique. The instrument collects a complete infrared spectrum at regular intervals, which is scanned in full, allowing the simultaneous detection and measurement of different gases present in the air at a given time. The measurement time interval, and the unit of measurement with which to express the gas concentration (mg/m^3 or ppm), can be set manually. The instrument has a resolution of 0.01 ppm for CH_4 and 1 ppm for CO_2.

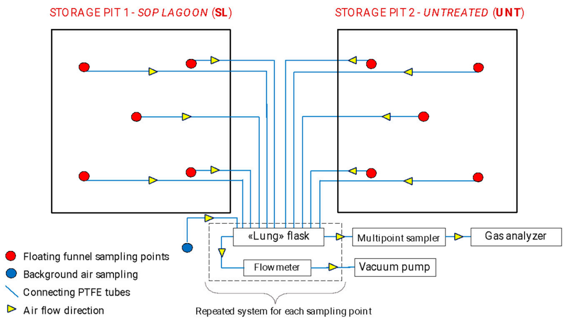

Figure 1. Scheme of the system for sampling and measuring air samples from above the two slurry storage pits under study.

Following the VERA protocol, sampling for each point was carried out for a 30 min period. The measurement period of the gas analyzer was set as low as possible, equal to approximately 43 s, resulting in a total of approximately 40 measurements per point every 30 min. The measurements were carried out by sampling alternating points from the SL tank and the UNT tank; this was done to prevent a potential bias in the data due to the daily variability of climatic conditions. Data regarding the average hourly temperature during measurement periods were retrieved from a nearby public climate control unit and considered in the data analysis. The background air was sampled for a time interval equal to 12 min between one point and another. The expected air-flow through the funnels was about 1.5 L/min, verified via electric flow meters.

The emission flows of the gases in question were finally calculated using Equation (1):

$$F = (Q \times (Cin - Cout))/A \tag{1}$$

where:

- F is the GHG flux (mg/m^2/h);
- Q is the air flow (m^3/h);
- Cin is the gas concentration in the air above the slurry surface, sampled by the funnel system (mg/m^3);
- Cout is the corresponding background gas air concentration (mg/m^3);
- A is the surface of the funnel (m^2).

2.4. Data Analysis

The data were analyzed by the analysis of variance (ANOVA) procedure using SPSS version 28. The sampling point within a single tank was considered as a replication. The average temperature during each point sampling was used as the covariate. Each sampling data were analyzed separately.

A two-way ANOVA was carried out to obtain indications about a possible interaction between the treatment and the sampling date.

3. Results

3.1. Slurry Chemical Characteristics

The results did not show numerically relevant differences in the chemical parameters between the UNT and SL. An average of the slurry characteristics is therefore reported in Table 1.

Table 1. Average slurry chemical characteristics.

Variable	Value
TS (%)	8.43 ± 0.12
VS (% TS)	74.25 ± 0.5
TKN (g/kg)	3.70 ± 0.06
TAN (g/kg)	1.69 ± 0.03
TAN (% TKN)	0.46 ± 0.01
Organic Carbon (%DM)	40.20 ± 0.20

± Standard error; TS: total solids; VS: volatile solids; TKN: total Kjeldahl nitrogen; TAN: total ammoniacal nitrogen; DM: dry matter.

The chemical composition of the slurry was in the range typically reported in the literature [36].

The pH analyses were statistically different on two of the four sampling dates, so the results are reported in Figure 2.

It is possible to observe that the pH was similar on day one and day three, but was lower and statistically significantly different ($p < 0.001$) in the UNT than in the SL on days two and four (−0.97 and −1.18, respectively), when the lowest values were recorded.

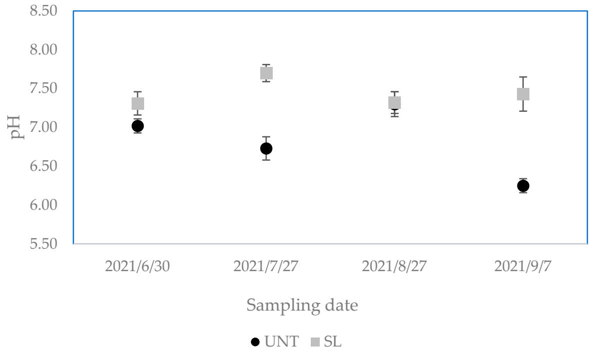

Figure 2. pH values. Bars indicate the standard error of the measurements. Plotted values represent the average per each treatment, SL or UNT, in the sampling dates.

3.2. Gas Emissions

In this experiment, the slurry additive SOP LAGOON was tested at the tank scale level to evaluate its ability to reduce CH_4 and CO_2 emissions.

The CH_4 and CO_2 fluxes were determined according to Equation (1) and are reported in the following graphs.

The CH_4 emissions (Figure 3) were almost at the same level for the SL and UNT on the first date, with slightly lower values registered for the UNT. The emissions for the UNT showed higher values in the subsequent dates, with a maximum of 59.64 mg/m^2/h (34.2% higher than the first measurement). On the other hand, the values from the SL were lower on the second, third and fourth days than what was recorded on the first day of sampling, with the minimum value (11.73 mg/m^2/h) registered on the third date. The third date measurement also showed the maximum difference between the SL and UNT −80%, ($p < 0.001$). At the last sampling date, the difference remained high at −75%, ($p < 0.001$).

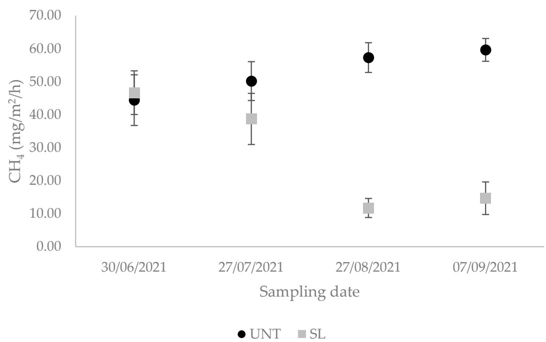

Figure 3. CH_4 emissions. Bars indicate the standard error of the measurements. Plotted values represent the average flux per each treatment, SL or UNT, in the sampling dates.

The CO_2 emissions are represented in Figure 4. As observed for CH_4, at the first sampling date, the emissions were very similar for both treatments. In the case of this gas, both fluxes registered the peak value on the second day of measurement, of 88.75 mg/m^2/h for the SL and 130.78 mg/m^2/h for the UNT, when the data were most dispersed. The untreated tank showed higher emission levels at the end of the trial period for the first sampling date. In comparison, the SL showed lower levels, resulting in differences of −75% (p < 0.05) and −46% (p < 0.001), respectively, for the third and fourth sampling dates in favor of SL over UNT.

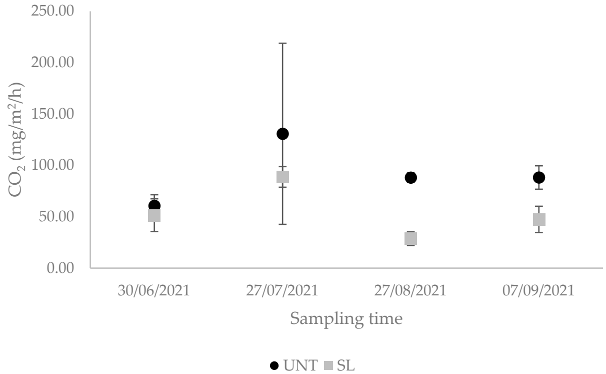

Figure 4. CO_2 emissions. Bars indicate the standard error of the measurements. Plotted values represent the average flux per each treatment, SL or UNT, in the sampling dates.

3.3. Additional Observations

As described in the Materials and Method section, a tractor-operated propeller was used to mix the manure tanks on the day before the measurements to break up the crust formed in the previous weeks and allow for easier measurement. It is worth reporting that after the slurry activation period, the agitation time for the SL was approximately a quarter of that of the UNT tank.

In addition, farm workers reported that the odors were nearly eliminated from the manure in the SL tank compared to the UNT tank at the time of spreading the manure on the fields.

4. Discussions

Slurry management and storage is a critical aspect of the livestock farming sector in generating GHG emissions, especially CH_4.

The complexity, labor intensity and the equipment requested for a real-scale on-site emission measurement are limiting factors in this research. In fact, most of the studies found in the literature are based on small- (vessels or barrels) or mid- (some tens of m^3 tanks) scale trials [37]. Often, when real tanks or lagoons are involved in the trial design, samples are collected from the farms for subsequent processing in a lab.

Measuring in the field in real time poses several technical challenges that must be addressed. The main one is the compatibility with the daily operations on the farm: for this work, this required coordination with the farmer in order to prevent safety issues and avoid damage to the measurement equipment, which was assembled and disassembled on each measurement day. For these reasons, the authors decided to maintain approximately one month between measurements.

The results of this work show an improved mitigation capacity of SOP LAGOON at scale than that previously measured [30,31], both for CH_4 and CO_2. The duration of the monitoring in this study was much longer than the previous works (5 months vs. 26 days or 1 week, respectively, for [30,31]). The experiment was performed in a commercial dairy, with pre-existing manure tanks, as opposed to the two preliminary works where the emissions were measured from manure in 220 L barrels.

The analysis of the chemical characteristics of the liquid manure in both UNT and SL does not show remarkable differences between the treatments or throughout the test dates. The values are similar to what Martínez-Suller et al. [36] found and are consistent with regular dairy farm practices, where the feed quality and composition do not vary significantly over the year.

The pH values of the liquid manure do not differ significantly when comparing one test day to the other, both for the UNT and SL. It can be noted, however, that the pH of the UNT was significantly lower than the SL at the second and fourth sampling dates. This seems to be in contrast with the emissions level, as a lower pH is generally associated with lower emissions: this is the principle of acidification. Slurry acidification (i.e., the application of strong acids to reduce the manure pH) has been investigated since 2012 [38] for its ability to curb GHG emissions, while, before that, it was mainly investigated for its ability to reduce ammonia NH_3 emissions [39]. Numerous studies [40–42] registered lower CH_4 emissions from an acidified slurry, from 49 to over 90%, obtained with the addition of different quantities of acid, from 2.4 L to 6 kg of acid (usually sulfuric acid) per m^3 of manure and with different storage conditions. The addition of SOP LAGOON shows emission reduction results in line with these numbers, without a significant change in the manure pH (Figure 2). The results of the chemical analyses in the SL showed very similar pH values in all the sampling dates, slightly basic between 7.3 and 7.6. This can be an advantage over acidification when spreading manure on soils, where the soil pH does not need to be lowered [43].

Acidification can also be achieved by adding some type of gypsum [44–46], the base material of the additive under test. In this study, the negligible pH variations measured suggest that the mechanism of action does not follow a chemical pathway. This was expected, considering that the results were obtained with the addition of only 2 g per week of SOP LAGOON per animal, producing approximately 0.3–0.5 m^3 of slurry per week, following the manufacturer's specifications. The lower quantity used here, compared to other options previously found in the literature (from 5% up to 30% on the dry weight of the manure to be treated) [44–46] necessary to achieve a significant decrease of the slurry pH, represents another advantage in terms of logistics and scalability over the use of other types of gypsum.

The manufacturer recommends the application of the product for at least three months to be able to observe the results: the duration of the treatment (103 days since the first application of the additive) is consistent with this indication and is long enough to potentially allow for the biological processes within the slurry to adapt to the treatment [47].

Given the considerations above, microbial changes seem to be the only viable mechanism of action to explain the results: how this interaction takes place appears likely due to the proprietary processing technology applied to the product and requires further investigation.

The results presented in this study showed that SOP LAGOON could reduce CH_4 emissions by up to approximately 80% during the storage phase.

Looking at Figure 3, it is possible to notice how UNT shows higher emission fluxes on the third and fourth date compared to the first measurement, i.e., after a storage period of 3–4 months, in accordance with the literature [48]. On the contrary, SL showed lower values than the first point of measure (approximately −70%, $p < 0.001$). Lowering the emissions rates of CH_4 with respect to the initial condition could have remarkable benefits for the climate, including carbon sequestration [49].

Over the years, other techniques have been investigated for their ability to reduce GHG emissions.

The most commonly proposed strategy to mitigate CH_4 emissions from liquid manure is the installation of biodigesters, which might not be economically viable for small-scale farms with less than 200 animals [50]. Moreover, inefficiencies in the plants, which often co-process manure together with agricultural residues, agro-industrial by-products or energy crops, cause the release of extra CH_4 in the atmosphere: in Italy, that accounts for approximately 1% of the total biogas production, especially from the digestate tanks, nearly offsetting the "avoided" CH_4 release from unprocessed manure [20].

Holly et al. [51] studied the different techniques to abate GHG emissions from liquid manure, including solid separation. They concluded that it could be another effective method to reduce the GHG emissions from stored manure, up to −46% compared to fresh manure. However, the direct GHG reduction can be partially offset by the carbon emissions connected to the production and use of the energy used to operate the machines.

In addition to this, the combination of the two above-mentioned techniques (digestion and separation) might even cancel the GHG mitigation potential of the two approaches taken singularly, as it can increase nitrous oxide (N_2O) emissions from the solid fraction [51,52] when compared to the unseparated manure. Another disadvantage is represented by the cost of equipment, structures and maintenance, which can impose a financial burden on the farmer, if it cannot be partially recovered by selling the gas or electricity to the market.

Kupper et al. [53] published a review of studies on the emissions from stored lagoons. They reported that manure covers could be another way to curb CH_4 emissions, with abatement rates between 10% to 60%, if they are impermeable. Guarino et al. [54] found that covers do not show a statistically significant efficacy when they are made of permeable materials. The natural crust that forms on top of a liquid manure tank is also considered a type of cover, with proven efficacy in reducing NH_3 and CH_4 emissions [55,56]. The UNT showed more significant CH_4 emissions than the SL, despite presenting a crust on the top. The higher crust thickness (indirectly measured by the longer time required to break it before each measurement day) might lead to a lower oxygen diffusion in the UNT, which can be a limiting factor in the methanotrophic activity [57].

Metanotrophy in the crust does not appear to be the only mechanism of action. The previous studies [30,31] on the same additive showed emissions reduction with little or no crust forming on top of the manure. In addition, the reduction of the odors at the time of spreading (as reported by the farm operators) suggests a different evolution of the liquid manure in the SL compared to the UNT, leading again to the conclusion that a different microbial activity occurred in the SL.

Moreover, breaking the crust in preparation for the field application generates an extra cost for the farmers regarding machinery operation, fuel consumption and manpower. As the agitators are commonly powered by tractors, the CO_2 released by the internal combustion engine partially offsets the GHG mitigation that the crust could offer. This topic deserves further study to better evaluate the scope of this trade-off.

This work also shows a significant great reduction (up to −75%) of the CO_2 emissions from the SL compared to the UNT. Scarce information is present in the literature on CO_2 fluxes, especially on the effect of additives or other containment systems. Generally, the research does not evaluate the CO_2 emissions from manure because they are considered part of a cycle that sees the plants used as feed for the animals as carbon sinks [58]. Additionally, the much greater air concentration of CO_2 compared to CH_4 (414 ppm vs. 1.8 ppm [59]) makes it difficult to separate the baseline air concentration from the contribution of the slurry. This is why, in this work, ambient air was sampled before each point on the manure surface, and the concentration was subtracted from the measured values in order to calculate the fluxes [34,35].

By analyzing the data at different times, it is possible to notice how the SL emissions were similar to those recorded on the starting date, while those from the UNT showed greater values than the first measured point, similar to what Borgonovo et al. [30] reported.

Additional Considerations

Odors connected to farming activities can be a nuisance for the surrounding communities, especially in regions with a high population density. Presently, there is no consensus among the different regions on odor regulations [60]: restrictions on management techniques, distance from inhabited areas and seasonality for the operations are already in place, even if only locally. Several strategies can be put in place to reduce the odor emissions from livestock manure, some of which are compatible with the goal of reducing GHG, such as anaerobic digestion, solid–liquid separation or covers [61].

During this test, the dairy farm operators reported that odors from the manure treated with SOP LAGOON were strongly reduced compared to the control when the manure was spread on the fields. This is in line with the observations reported by Peterson et al. [31], who measured a significant reduction of odors from the treated lagoon water. SOP LAGOON provides similar benefits to the other techniques targeting odor issues.

5. Conclusions

Liquid manure is a critical source of GHG emissions from the dairy industry.

After three months from the first additive applications, the treated storage tank showed lower and statistically significantly different emissions compared to the untreated one, with results (up to −80% for CH_4 and −75% for the CO_2) that are compatible or better than other more complex strategies such as acidification or methane digesters.

The data presented in this work shows great potential for SOP LAGOON to reduce CH_4 and CO_2 emissions from liquid manure storage, in real field scenarios, confirming and demonstrating improved results than what was shown in the previous small-scale studies. The in-field test also allowed the operators to report a reduction in the odors at the time of spreading and a reduction of the fuel consumption for the agitation.

In this work, SOP LAGOON proves to be an effective additive to help the farmers manage their stored liquid manure, which can offer economic, social and environmental benefits for the dairy industry.

Further studies could also investigate the effects of SOP LAGOON on the emissions at the moment of the manure spreading and its influence on soil and crops.

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References

1. Climate Change 2022: Impacts, Adaptation and Vulnerability. In *Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*; Pörtner, H.-O.; Roberts, D.C.; Tignor, M.M.B.; Poloczanska, E.S.; Mintenbeck, K.; Alegría, A.; Craig, M.; Langsdorf, S.; Löschke, S.; Möller, V.; et al. (Eds.) Cambridge University Press: Cambridge, UK; New York, NY, USA, 2022; p. 3056. [CrossRef]
2. WMO Update: 50:50 Chance of Global Temperature Temporarily Reaching 1.5 °C Threshold in Next Five Years. Available online: https://public.wmo.int/en/media/press-release/wmo-update-5050-chance-of-global-temperature-temporarily-reaching-15%C2%B0c-threshold (accessed on 14 December 2022).
3. Clark, M.A.; Domingo, N.G.G.; Colgan, K.; Thakrar, S.K.; Tilman, D.; Lynch, J.; Azevedo, I.L.; Hill, J.D. Global Food System Emissions Could Preclude Achieving the 1.5° and 2°C Climate Change Targets. *Science* **2020**, *370*, 705–708. [CrossRef] [PubMed]
4. Sun, X.; Wang, P.; Ferris, T.; Lin, H.; Dreyfus, G.; Gu, B.-H.; Zaelke, D.; Wang, Y. Fast Action on Short-Lived Climate Pollutants and Nature-Based Solutions to Help Countries Meet Carbon Neutrality Goals. *Adv. Clim. Change Res.* **2022**, *13*, 564–577. [CrossRef]
5. Xu, Y.; Ramanathan, V. Well below 2 °C: Mitigation Strategies for Avoiding Dangerous to Catastrophic Climate Changes. *Proc. Natl. Acad. Sci. USA* **2017**, *114*, 10315–10323. [CrossRef] [PubMed]
6. Methane Emissions. Available online: https://energy.ec.europa.eu/topics/oil-gas-and-coal/methane-emissions_en (accessed on 14 December 2022).
7. Allen, M.R.; Shine, K.P.; Fuglestvedt, J.S.; Millar, R.J.; Cain, M.; Frame, D.J.; Macey, A.H. A Solution to the Misrepresentations of CO_2-Equivalent Emissions of Short-Lived Climate Pollutants under Ambitious Mitigation. *NPJ Clim. Atmos. Sci.* **2018**, *1*, 16. [CrossRef]
8. The Biogenic Carbon Cycle and Cattle. Available online: https://clear.ucdavis.edu/explainers/biogenic-carbon-cycle-and-cattle (accessed on 14 December 2022).
9. Global Methane Pledge. Available online: https://www.globalmethanepledge.org/ (accessed on 14 December 2022).
10. GMP COP27 Joint Factsheet. Available online: https://energy.ec.europa.eu/system/files/2022-11/GMP%20COP27%20Joint%20Factsheet.pdf (accessed on 14 December 2022).
11. SB 1383. Available online: https://www.yubasutterrecycles.com/files/2793c2592/SB+1383.pdf (accessed on 14 December 2022).
12. Brandt, A.R.; Heath, G.A.; Cooley, D. Methane Leaks from Natural Gas Systems Follow Extreme Distributions. *Environ. Sci. Technol.* **2016**, *50*, 12512–12520. [CrossRef] [PubMed]
13. Regulation of the European Parliament and of the Council. Available online: https://eur-lex.europa.eu/resource.html?uri=cellar:06d0c90a-5d91-11ec-9c6c-01aa75ed71a1.0001.02/DOC_1&format=PDF (accessed on 14 December 2022).
14. European Commission. An EU Methane Strategy. Available online: https://ec.europa.eu/commission/presscorner/detail/en/qanda_20_1834 (accessed on 14 December 2022).
15. Climate Change and Land—IPCC. Available online: https://www.ipcc.ch/site/assets/uploads/sites/4/2020/02/SPM_Updated-Jan20.pdf (accessed on 31 December 2022).
16. Emissions from Livestock and Manure Management. Available online: https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Ch10_Livestock.pdf (accessed on 14 December 2022).
17. Estimated Global Anthropogenic Methane Emissions by Source 2010. Available online: https://www.igsd.org/estimated-global-anthropogenic-methane-emissions-by-source-2010/ (accessed on 14 December 2022).
18. The Role of the Dairy Sector in a Low-Carbon Future. Available online: https://www.fao.org/3/CA2929EN/ca2929en.pdf (accessed on 14 December 2022).
19. Amon, B.; Kryvoruchko, V.; Moitzi, G.; Amon, T. Greenhouse Gas and Ammonia Emission Abatement by Slurry Treatment. *Int. Congr. Ser.* **2006**, *1293*, 295–298. [CrossRef]
20. Italian Greenhouse Gas Inventory. Available online: https://www.isprambiente.gov.it/files2022/pubblicazioni/rapporti/nir2022_italy_r360.pdf (accessed on 14 December 2022).
21. Current California GHG Emission Inventory Data. Available online: https://ww2.arb.ca.gov/ghg-inventory-data (accessed on 14 December 2022).
22. McCauley, J.I.; Labeeuw, L.; Jaramillo-Madrid, A.C.; Nguyen, L.N.; Nghiem, L.D.; Chaves, A.V.; Ralph, P.J. Management of Enteric Methanogenesis in Ruminants by Algal-Derived Feed Additives. *Curr. Pollut. Rep.* **2020**, *6*, 188–205. [CrossRef]
23. Jayanegara, A.; Sarwono, K.A.; Kondo, M.; Matsui, H.; Ridla, M.; Laconi, E.B.; Nahrowi. Use of 3-Nitrooxypropanol as Feed Additive for Mitigating Enteric Methane Emissions from Ruminants: A Meta-Analysis. *Ital. J. Anim. Sci.* **2018**, *17*, 650–656. [CrossRef]
24. Honan, M.; Feng, X.; Tricarico, J.M.; Kebreab, E.; Honan, M.; Feng, X.; Tricarico, J.M.; Kebreab, E. Feed Additives as a Strategic Approach to Reduce Enteric Methane Production in Cattle: Modes of Action, Effectiveness and Safety. *Anim. Prod. Sci.* **2021**, *62*, 1303–1317. [CrossRef]
25. Carrazco, A.V.; Peterson, C.B.; Zhao, Y.; Pan, Y.; McGlone, J.J.; DePeters, E.J.; Mitloehner, F.M. The Impact of Essential Oil Feed Supplementation on Enteric Gas Emissions and Production Parameters from Dairy Cattle. *Sustainability* **2020**, *12*, 10347. [CrossRef]
26. Mosque, J.; Schils, R.; Groenestein, K.; Hoeksma, P.; Velthof, G.; Hummelink, E. Emissies van Lachgas, Methaan En Ammoniak Uit Mest.Pdf. Available online: https://edepot.wur.nl/161899 (accessed on 14 December 2022).

27. VanderZaag, A.C.; Baldé, H.; Crolla, A.; Gordon, R.J.; Ngwabie, N.M.; Wagner-Riddle, C.; Desjardins, R.; MacDonald, J.D. Potential Methane Emission Reductions for Two Manure Treatment Technologies. *Environ. Technol.* **2018**, *39*, 851–858. [CrossRef] [PubMed]
28. Misselbrook, T.; Hunt, J.; Perazzolo, F.; Provolo, G. Greenhouse Gas and Ammonia Emissions from Slurry Storage: Impacts of Temperature and Potential Mitigation through Covering (Pig Slurry) or Acidification (Cattle Slurry). *J. Environ. Qual.* **2016**, *45*, 1520–1530. [CrossRef] [PubMed]
29. Amon, B.; Kryvoruchko, V.; Fröhlich, M.; Amon, T.; Pöllinger, A.; Mösenbacher, I.; Hausleitner, A. Ammonia and Greenhouse Gas Emissions from a Straw Flow System for Fattening Pigs: Housing and Manure Storage. *Livest. Sci.* **2007**, *112*, 199–207. [CrossRef]
30. Borgonovo, F.; Conti, C.; Lovarelli, D.; Ferrante, V.; Guarino, M. Improving the Sustainability of Dairy Slurry by A Commercial Additive Treatment. *Sustainability* **2019**, *11*, 4998. [CrossRef]
31. Peterson, C.B.; El Mashad, H.M.; Zhao, Y.; Pan, Y.; Mitloehner, F.M. Effects of SOP Lagoon Additive on Gaseous Emissions from Stored Liquid Dairy Manure. *Sustainability* **2020**, *12*, 1393. [CrossRef]
32. Baird, R.B.; Eaton, A.D.; Rice, E.W. (Eds.) *Standard Methods for the Examination of Water and Wastewater*, 23rd ed.; American Public Health Association, American Water Works Association, Water Environment Federation: Washington, DC, USA, 2017.
33. VERA Test Protocol. Available online: https://www.vera-verification.eu/app/uploads/sites/9/2019/05/VERA_Test_protocol_slurryseparation_v3_2018_final.pdf (accessed on 14 December 2022).
34. Balsari, P.; Airoldi, G.; Dinuccio, E.; Gioelli, F. Ammonia Emissions from Farmyard Manure Heaps and Slurry Stores—Effect of Environmental Conditions and Measuring Methods. *Biosyst. Eng.* **2007**, *97*, 456–463. [CrossRef]
35. Finzi, A.; Mattachini, G.; Lovarelli, D.; Riva, E.; Provolo, G. Technical, Economic, and Environmental Assessment of a Collective Integrated Treatment System for Energy Recovery and Nutrient Removal from Livestock Manure. *Sustainability* **2020**, *12*, 2756. [CrossRef]
36. Martínez-Suller, L.; Azzellino, A.; Provolo, G. Analysis of Livestock Slurries from Farms across Northern Italy: Relationship between Indicators and Nutrient Content. *Biosyst. Eng.* **2008**, *99*, 540–552. [CrossRef]
37. Regueiro, I.; Coutinho, J.; Fangueiro, D. Alternatives to Sulfuric Acid for Slurry Acidification: Impact on Slurry Composition and Ammonia Emissions during Storage. *J. Clean. Prod.* **2016**, *131*, 296–307. [CrossRef]
38. Petersen, S.O.; Andersen, A.J.; Eriksen, J. Effects of Cattle Slurry Acidification on Ammonia and Methane Evolution during Storage. *J. Environ. Qual.* **2012**, *41*, 88–94. [CrossRef]
39. Ndegwa, P.M.; Hristov, A.N.; Arogo, J.; Sheffield, R.E. A Review of Ammonia Emission Mitigation Techniques for Concentrated Animal Feeding Operations. *Biosyst. Eng.* **2008**, *100*, 453–469. [CrossRef]
40. Habtewold, J.; Gordon, R.; Sokolov, V.; VanderZaag, A.; Wagner-Riddle, C.; Dunfield, K. Reduction in Methane Emissions From Acidified Dairy Slurry Is Related to Inhibition of Methanosarcina Species. *Front. Microbiol.* **2018**, *9*, 2806. [CrossRef]
41. Sokolov, V.K.; VanderZaag, A.; Habtewold, J.; Dunfield, K.; Wagner-Riddle, C.; Venkiteswaran, J.J.; Crolla, A.; Gordon, R. Dairy Manure Acidification Reduces CH_4 Emissions over Short and Long-Term. *Environ. Technol.* **2021**, *42*, 2797–2804. [CrossRef] [PubMed]
42. Ma, C.; Dalby, F.R.; Feilberg, A.; Jacobsen, B.H.; Petersen, S.O. Low-Dose Acidification as a Methane Mitigation Strategy for Manure Management. *Agric. Sci. Technol.* **2022**, *2*, 437–442. [CrossRef]
43. Park, S.H.; Lee, B.R.; Jung, K.H.; Kim, T.H. Acidification of Pig Slurry Effects on Ammonia and Nitrous Oxide Emissions, Nitrate Leaching, and Perennial Ryegrass Regrowth as Estimated by 15N-Urea Flux. *Asian-Australas. J. Anim. Sci.* **2018**, *31*, 457–466. [CrossRef] [PubMed]
44. Febrisiantosa, A.; Ravindran, B.; Choi, H.L. The Effect of Co-Additives (Biochar and FGD Gypsum) on Ammonia Volatilization during the Composting of Livestock Waste. *Sustainability* **2018**, *10*, 795. [CrossRef]
45. SangSun, L.; HyunJin, P.; XiYing, H.; SunIl, L.; ByeongJun, J.; JinHyeob, K.; WooJung, C. Nitrogen, Carbon, and Dry Matter Losses during Composting of Livestock Manure with Two Bulking Agents as Affected by Co-Amendments of Phosphogypsum and Zeolite. *Ecol. Eng.* **2017**, *102*, 280–290.
46. Hao, X.; Larney, F.J.; Chang, C.; Travis, G.R.; Nichol, C.K.; Bremer, E. The Effect of Phosphogypsum on Greenhouse Gas Emissions during Cattle Manure Composting. *J. Environ. Qual.* **2005**, *34*, 774–781. [CrossRef]
47. Speshock, J.L.; Brady, J.A.; Eastman, J.; Roach, T.; Hays, S.; Kattes, D. Impact of Manure Storage Time and Temperature on Microbial Composition and Stable Fly (Diptera: Muscidae) Development. *Adv. Microbiol.* **2019**, *9*, 248–265. [CrossRef]
48. Sokolov, V.; VanderZaag, A.; Habtewold, J.; Dunfield, K.; Tambong, J.T.; Wagner-Riddle, C.; Venkiteswaran, J.J.; Gordon, R. Acidification of Residual Manure in Liquid Dairy Manure Storages and Its Effect on Greenhouse Gas Emissions. *Front. Sustain. Food Syst.* **2020**, *4*, 568648. [CrossRef]
49. Lynch, J.; Cain, M.; Pierrehumbert, R.; Allen, M. Demonstrating GWP *: A Means of Reporting Warming-Equivalent Emissions That Captures the Contrasting Impacts of Short- and Long-Lived Climate Pollutants. *Environ. Res. Lett.* **2020**, *15*, 044023. [CrossRef]
50. Agrisource. Available online: http://agrisource.org/fr/1/home.html (accessed on 15 December 2022).
51. Holly, M.A.; Larson, R.A.; Powell, J.M.; Ruark, M.D.; Aguirre-Villegas, H. Greenhouse Gas and Ammonia Emissions from Digested and Separated Dairy Manure during Storage and after Land Application. *Agric. Ecosyst. Environ.* **2017**, *239*, 410–419. [CrossRef]

52. Veltman, K.; Rotz, C.A.; Chase, L.; Cooper, J.; Ingraham, P.; Izaurralde, R.C.; Jones, C.D.; Gaillard, R.; Larson, R.A.; Ruark, M.; et al. A Quantitative Assessment of Beneficial Management Practices to Reduce Carbon and Reactive Nitrogen Footprints and Phosphorus Losses on Dairy Farms in the US Great Lakes Region. *Agric. Syst.* **2018**, *166*, 10–25. [CrossRef]
53. Kupper, T.; Häni, C.; Neftel, A.; Kincaid, C.; Bühler, M.; Amon, B.; VanderZaag, A. Ammonia and Greenhouse Gas Emissions from Slurry Storage—A Review. *Agric. Ecosyst. Environ.* **2020**, *300*, 106963. [CrossRef]
54. Guarino, M.; Fabbri, C.; Brambilla, M.; Valli, L.; Navarotto, P. Evaluation of simplified covering systems to reduce gaseous emissions from livestock manure storage. *Trans. ASABE* **2006**, *49*, 737–747. [CrossRef]
55. A Review of Livestock Methane Emission Factors. Available online: https://www.epa.ie/publications/research/climate-change/Research_Report_288.pdf (accessed on 14 December 2022).
56. Aguerre, M.J.; Wattiaux, M.A.; Powell, J.M. Emissions of Ammonia, Nitrous Oxide, Methane, and Carbon Dioxide during Storage of Dairy Cow Manure as Affected by Dietary Forage-to-Concentrate Ratio and Crust Formation. *J. Dairy Sci.* **2012**, *95*, 7409–7416. [CrossRef] [PubMed]
57. Duan, Y.-F.; Reinsch, S.; Ambus, P.; Elsgaard, L.; Petersen, S. Methanotrophic Activity in Slurry Surface Crusts as Influenced by CH4, O2, and Inorganic N. *J. Environ. Qual.* **2017**, in press. [CrossRef]
58. Külling, D.R.; Dohme, F.; Menz, H.; Sutter, F.; Lischer, P.; Kreuzer, M. Methane Emissions of Differently Fed Dairy Cows and Corresponding Methane and Nitrogen Emissions from Their Manure during Storage. *Environ. Monit. Assess.* **2002**, *79*, 129–150. [CrossRef]
59. USEPA. Climate Change Indicators: Atmospheric Concentrations of Greenhouse Gases. Available online: https://www.epa.gov/climate-indicators/climate-change-indicators-atmospheric-concentrations-greenhouse-gases (accessed on 14 December 2022).
60. Bokowa, A.; Diaz, C.; Koziel, J.A.; McGinley, M.; Barclay, J.; Schauberger, G.; Guillot, J.-M.; Sneath, R.; Capelli, L.; Zorich, V.; et al. Summary and Overview of the Odour Regulations Worldwide. *Atmosphere* **2021**, *12*, 206. [CrossRef]
61. Janni, K. Reflections on Odor Management for Animal Feeding Operations. *Atmosphere* **2020**, *11*, 453. [CrossRef]

Supplementary Materials

As anticipated in the presentation of the paper, the work was carried out to analyze the emissions of ammonia (NH_3), nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2), i.e., the three main greenhouse gases, plus the other gas of greater environmental importance related to agriculture, as discussed in detail in the introduction.
Nonetheless, the publication deals only with methane and carbon dioxide.
As regards nitrous oxide, all the data collected during the tests were found to be inconsistent between the different tests, as well as with the trend observed for the other gases and with what was expected compared to the bibliographical references. Although the cause of these results was not clarified, perhaps a problem in the calibration of the gas analyzer used, they were excluded from the analysis.
Ammonia, however, was excluded due to a choice by the authors in terms of the contents of the paper, which was set up with a focus on climate-changing gases. This choice was also justified by the fact that ammonia was measured during the tests with a different method than the other gases, i.e. with a so-called acid trap system, and not with an infrared gas analyzer.
For the sake of completeness, the methodology and results obtained with respect to ammonia monitoring are reported below, as a supplement, albeit brief, to the paper.
The measurement system described in detail in the paper in paragraph 2.3 "*Emission and Fluxes Determination*" was in fact also equipped with "acid traps" for each sampling point. These were each composed of two 500 ml drechsels, containing 300 ml of 1% boric acid solution, sealed and connected to each other (see Figure S1 below), and were positioned, within the scheme shown in Figure 1 of the paper, between the "lung" flasks and flow meters. The flow directed to the acid trap enters the drechsel A and then the B before passing through the flow meter, so that the acid solution captures the ammonia present in the air. Finally, the ammonia content of the solution present in the acid traps was measured through laboratory analysis at the end of each single day of the field test, by titration with sulfuric acid (N = 0.01). This allowed to determine the total amount of ammonia trapped during the test and, subsequently, through a relationship with the average air flow measured by the flow meters, the average NH_3 concentration of the air passing through the drechsels during the measurement period.
The results are reported graphically in Figure S2 and commented below.

Figure S1 - Detail of one of the acid traps used to detect ammonia emissions. In the photo it is recognizable that the solution is bubbling due to the passage of the air sucked in by the vacuum generated by the pump

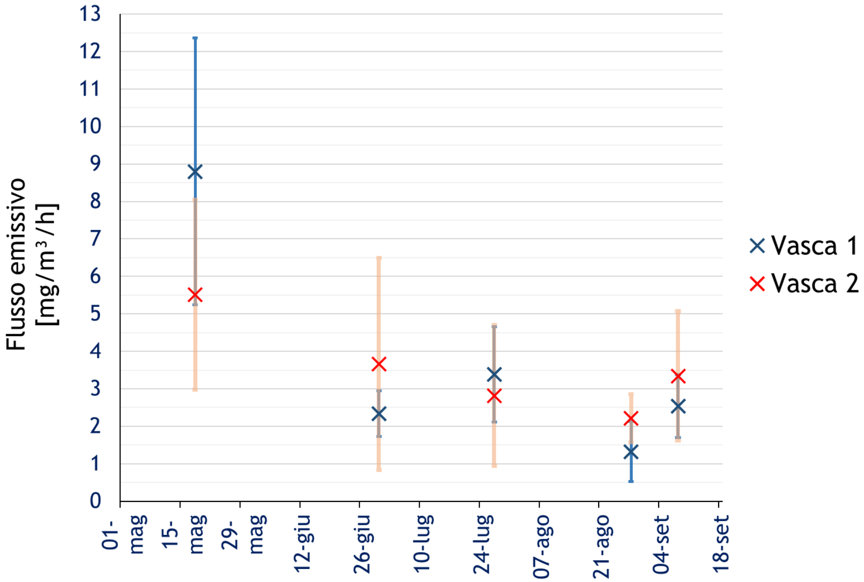

Figure S2 - Emission flow from the storage tanks detected with an acid trap system. The graph shows the mean ± standard deviation (error bars) divided by tank and sampling date.

No statistically significant differences emerged in terms of NH_3 emission flux between the two tanks in question for the entire duration of the test.

Tank 1 was the one treated with the SOP Lagoon additive, which was added periodically to the slurry starting from the first half of June.

The first measurement was carried out before the start of treatment with the additive to verify whether the two tanks had statistically comparable emissions. The difference between the average NH_3 emission flux of the two tanks on this occasion was found to be not statistically significant. In this test the highest emission flow of the entire measurement period was detected, which is probably since it was the test carried out with the greatest quantity of slurry present inside the tanks, and therefore with the greatest volume stored per m^2 of emitting surface. However, in this same test the greatest variability in absolute terms of emissions was detected.

From the start of the treatment the emission flow of the treated tank was on average lower than that of the control tank (except for the test dated 27/7 in which the opposite was observed). The lower average emissions detected during the 27/8 test were probably due to a relatively low average temperature during the day in which the measurements were carried out. Nonetheless, due to the high variability of the values detected, no statistically significant differences were observed between the two tanks for any of the tests carried out. In fact, the p-values obtained from the statistical analyzes carried out were always greater than the significance level set for the statistical test (5%). This variability is graphically depicted by the error bars shown in blue for Tank 1 and red for Tank 2 in Figure S2, which in fact represent the standard deviations of the measurements and appear largely overlapped for all tests. In absolute terms, the emission values always remained constant (no seasonal trends were observed), always between 1 and 4 $mg/m^2/h$ both for the treated tank and for the control, values in line with literature for the traditional storage of bovine slurry with natural formation of surface crust.

Photographic Appendix

Figure S3 - Close up of the floating PVC funnels used for setting up the measurement system.

Figure S4 - Floating PVC funnels placed on the slurry surface in one of the slurry storages during the measurements.

Figure S5 - Floating PVC funnels placed on the slurry surface in one of the slurry storages during the measurements.

4.2. The effects of incorporating renewable energy on the environmental footprint of beef production

Submitted to: *Energy*, under review

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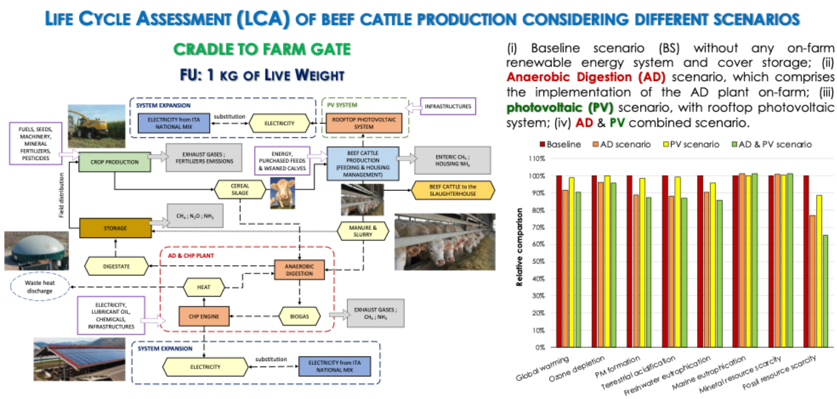
Graphical abstract



Highlights

- Modelling environmental scenarios for the implementation of anaerobic digestion and photovoltaic systems on a beef farm
- Electricity co-production managed with system expansion
- Environmental improvements in 6 out of 8 impact categories evaluated across all analysed scenarios
- Combined, the two systems achieve reductions of up to -12% in GWP and -35% in fossil resource use
- Trade-offs concern eutrophication and mineral resource use but are limited to a maximum of +1.1% of the baseline impact

Abstract

This study quantifies the influence of the on-farm implementation of different energy mitigation systems, anaerobic digestion (AD) for biogas production and rooftop photovoltaics (PV), and assesses the environmental and energy impact on beef cattle production. Data on technical aspects were collected and a cradle-to-farm gate life cycle assessment approach was adopted. Two baseline production scenarios, with conventional manure and slurry management (considering different slurry storage: open or covered), were compared with three alternatives: (i) with the implementation of AD plant only; (ii) with the implementation of a PV system only; and (iii) with both. Impacts on the infrastructure and operation of AD plant and PV systems were considered, as well as their influence on emissions and electricity generation. The latter was managed with a system expansion, considering an environmental credit. The results, expressed per 1 kg of live weight of beef cattle produced, showed widespread improvements across the impact categories assessed. The AD scenario presented larger mitigations than the PV system alone, but the best result is achieved when both energy systems are implemented, with global warming potential reduced by 12% and fossil resource scarcity by 35%. This work

represents a benchmark for future life cycle analysis of renewable energy system implementation for livestock.

Keywords

Anaerobic digestion, Manure management, Environmental Assessment, Photovoltaic system, Renewable energy, Beef production

List of acronyms and abbreviations

AD – Anaerobic Digestion

BS – Baseline Scenario

CH_4 – methane

CHP – Combined Heat and Power

EEA – European Environment Agency

FEP – Freshwater Eutrophication

FRS – Fossil Resource Scarcity

FU – Functional Unit

GWP – Global Warming Potential

IPCC – Intergovernmental Panel on Climate Change

LCA – Life Cycle Assessment

LW – Live Weight

MEP – Marine Eutrophication

MSR – Mineral Resource Scarcity

N_2O – Dinitrogen Monoxide

NH_3 – Ammonia

ODP – Ozone Depletion Potential

PM – Particulate Matter Formation

PV – Photovoltaic

TAP – Terrestrial Acidification

1. Introduction

It is now recognised that the production of beef cattle represents an environmental hotspot within the agricultural sector, in terms of carbon footprint, eutrophication, and acidification potential (LEAP, 2016), albeit with great internal variability. This is true when considering the direct comparison between product units (e.g. per kilogram of produced meat), when referring to its role within the average European diet (Notarnicola et al., 2017), and when looking at the absolute emission profile of the agri-food production sectors (Poore et al., 2018).

In Italy, beef-cattle farming only accounts for about 4% of the turnover of the agro-industry and suffers a strong dependence on imports, with a degree of self-supply of around 50%. Nevertheless, the sector is well structured, involves many stakeholders and is widespread throughout the country. In 2019, there were about 94.6 thousand farms specialising in this production, with a total of 2.635 million animals slaughtered per year. Furthermore, the number of animals reared is increasing (an increase of 8.6% in the total beef-cattle population over the 5-year period 2015-2020), despite the fact that apparent per capita consumption of beef in Italy (16.8 kg in 2019) is observing a decreasing trend (ISMEA, 2021).

At an environmental level, it is well known that manure management plays an important role in livestock production, especially impacting GHG emissions and the nitrogen cycle (McClelland et al., 2018). In this regard, the anaerobic digestion (AD) of livestock waste for biogas production is regarded as one of the most effective management techniques, from an environmental point of view (Freitas et al., 2022). All over the European Union (EU), member states' subsidies have been promoting electricity generation from bioenergy sources since 2009, following the Directive 2009/28/CE. Subsequently, the use of AD of agricultural biomass and combined heat and power (CHP) plants has become widespread. More and more livestock farms have implemented these plants, either privately owned or collectively, in agricultural consortiums (Burg et al., 2021), using livestock waste and, eventually, other agricultural biomass as feedstock due to its economic viability (Benaco et al., 2019; Lovarelli et al., 2019). The AD of biomass from waste or by-products is now established as an important pillar of the circular bio-economy of the energy sector within the EU (Khoshnevisan et al., 2021). Given that the Circular Economy Action Plan (European Commission; COM/2020/98 final) states that circularity is a prerequisite for climate neutrality, it follows that AD also plays an important role in the EU's climate goals, making the topic even more relevant.

Another topic that has attracted a lot of interest in agriculture in the last decade, particularly in Europe, is the adoption of solar power systems, or photovoltaic (PV) systems. It is a technology with enormous energy and environmental potential (Haas et al., 2023), but is still largely unexpressed for farms. This can be implemented in different ways, by means of integrated PV or rooftop PV on farming structures such as stables, warehouses, and greenhouses, and even with land-based PV (referred to as agrivoltaic) (Dinesh et al., 2016). The most innovative option, but also the most discussed, is that of agrivoltaics, on which research in this sector is mainly focusing (Chalgynbayeva et al., 2023). Rooftop-based plants are more conventional and there are already several environmental and economic analyses in this regard (Hollingsworth et al., 2019). Speaking specifically of rooftop PV systems on barns, however, the evidence in the literature is scarce. In a review of electricity use and generation in dairy farms, Mohsenimanesh et al. (2021) dealt with the energy potential of this technology on barn rooftops. The authors also mention the important economic aspects regarding the uninterrupted fall in the costs of PV modules over the last decade, which makes the technology more and more competitive.

LCA is an approach regulated by ISO 14040 (ISO, 2006) and 14044 (ISO, 2006), and subsequent amendments, to analyse products, processes, or services from an environmental perspective along the entire life cycle, or part of it. The application of LCA to agri-food supply chains is increasingly adopted for environmental analysis and claims. Regarding the beef production sector, numerous studies have been published, in an international context

(Asem-Hiablie et al., 2019) and an Italian context (Berton et al., 2017; Bragaglio et al., 2018), due to the growing attention to food sustainability and this supply chain. At the same time, several studies have investigated the impact of the AD of agricultural waste biomass on biogas production, concluding that it is a practice that, under certain conditions, has the potential to (i) reduce the impact of traditional livestock waste management; (ii) generate generally more sustainable electricity than the current European mixes; (iii) generate an attractive source of income; and (iv) generate a further series of benefits, including the reduction of treated waste odour (Burg et al., 2017; Ingrao et al., 2019). Nevertheless, the focus of most of these studies was energy production, considering the plant as a stand-alone system and, therefore, considering only inputs and outputs directly connected to it (Ingrao et al., 2019).

This study, on the other hand, aims to evaluate the environmental impact of beef cattle production by using the LCA approach, when combined with anaerobic digestion plants fed with the resulting livestock waste (manure and slurry) in an overall perspective. By using this approach, Wu et al. (2020) analysed the influence of integrated pig farming and anaerobic mono-digestion; Bacenetti et al. (2016) explored the implementation of anaerobic digestion in a cow dairy system; and Chirone et al. (2022) studied buffalo dairy systems. This study explores the implementation of PV systems on barn rooftops and any additional energy or environmental benefits derived from it. Pascaris et al. (2021) measured the combination of PV systems with animal production in a life cycle perspective, but that study was concerned with agrivoltaics on rabbit pastures. The novelty of this study is that, to the authors' knowledge, it is the first environmental analysis that deals with the effects of integrating the above-mentioned renewable energy production systems with beef cattle production.

2. Materials and methods

2.1. Goal and scope definition and scenario modelling

This study aims to quantify the mitigation potential of two renewable energy production systems widely implemented in livestock farms, namely AD for biogas production and PV systems. This undertaking considers the integration of renewable electricity generation from these two sources into a beef system, by using the LCA approach. For this purpose, this work focuses on the environmental analysis of a beef cattle farm in northern Italy, equipped with an operating plant for the AD of livestock waste and subsequent conversion of biogas into electricity and a PV system that includes multi-crystalline Si panels integrated into cattle barn roofs. The farm practices an intensive open-cycle cattle farming system, that covers only a part of the rearing cycle. Weaned calves are bought externally from pasture-based systems, mainly in France, and are directly managed through the fattening part of the process, where animals are fed a mixed diet of self-produced fodder and commercial feed, with supplements purchased externally. This production system is particularly widespread in the north of the country, its incidence on national beef production is around 44-48% (ISMEA, 2021), and it has been extensively described in the literature (Berton et al., 2017; Bragaglio et al., 2018). The farm produces maize silage, which it uses to partially satisfy its animal feed requirements. Part of the silage is also fed to AD along with livestock waste, as it is a very common practice to use it as an energy crop in co-digestion with livestock waste in agricultural biomass plants (Lijó et al., 2017).

In this work, different productive scenarios (two baseline scenarios and three alternative ones) are developed for comparative purposes:

- the two Baseline Scenarios (BS) represent the standard beef cattle production system without implementing any on-farm renewable energy generation system. The difference between the two lies in slurry management: in the first scenario (BS-open) this is stored in uncovered tanks while, in the second (BS-cover), it is stored in covered tanks. This is to represent existing farms that have not implemented livestock waste AD and are managing it, either by following best practices or not;
- the AD scenario comprises the on-farm implementation of the anaerobic digestion plant;
- the PV scenario comprises the on-farm implementation of the photovoltaic system. Since this technology does not affect maintenance management, this alternative scenario is split into two, depending on whether the system is implemented on a BS-open or BS-cover;

- the AD and PV scenario comprises the implementation of both.

All scenarios share the same crop cultivation and livestock management data but differ in the modelling of manure management. In the baseline scenarios, livestock waste is handled in the form of manure and slurry, as cattle are partially housed on straw bedding and partially slatted floor structures. More specifically, manure is handled with deep bedding and subsequent solid storage, while slurry is collected in open tanks (for BS-open) or covered tanks (for BS-cover). In the scenarios where the AD plant is implemented instead, both slurry and manure are managed as a feedstock for the AD plant, and all the other inputs and output flows related to the AD plant are also included. In this scenario, the digestate resulting from the waste treatment is stored in covered tanks. In fact, according to the most recent regulations, newly implemented AD plants require covered post-treatment storage. In the scenarios where a PV system is implemented, all input and output flows related to the solar power system are considered.

The outcomes of this study are aimed at researchers and stakeholders involved in the agri-food industry, to understand and quantify the potential impact caused by the implementation of anaerobic digestion plants and PV systems within a beef production system. The results can also be useful for policy makers working with agro-environmental regulations, e.g. to support decision making phases and direct price rewards or incentives for actions aimed at the mitigation of environmental impacts of agricultural activities.

2.2. Functional unit and system boundaries

This study was carried out with a cradle-to-farm gate perspective, as it focused on the agricultural phase of beef production, which is known to be the main hotspot of the whole beef life cycle impacts. The selected Functional Unit (FU) is 1 kg of live weight (LW) produced, intended as to mean the mass of cattle leaving the farm to the slaughterhouse. This FU is widely adopted in LCA literature related to the livestock agricultural phase and is also suggested by the LEAP guidelines (2016). All of the input and output inventory data and, consequently, the functional unit, refer to a specific period, i.e. 2021.

The system boundaries are schematised in **Figure 1**, where the subsystems of the alternative scenarios and the system expansions linked to them are also highlighted. Manufacture (including the extraction of raw materials), supply and use of all raw input materials consumed for crop cultivation (such as seeds, fuels, fertilisers and pesticides) are included, as well as all the derived field application emissions. The indirect environmental burdens of virtual consumption of tractors and other machinery, including maintenance and final disposal, were also considered. In contrast, the indirect impact of the farm's capital goods (buildings, warehouses) was not taken into account, as it was considered to be scarcely influential due to their long life span. As for livestock, the boundaries include the whole rearing cycle, thus considering inputs (weaned calves, raw materials, energy and fuels) and outputs such as animal-related emissions (i.e. enteric fermentations and manure-related emissions). Impacts associated with the production and usage of veterinary medicines and cleaning products were not included. Agricultural land use has remained constant for a long time in the Po Valley area, where the farms are located. Hence, soil organic carbon was assumed to be in a steady-state, not involving CO_2 emissions to the atmosphere or carbon sequestration (IPCC, 2006, 2019). Therefore, no direct on-farm land use changes (dLUC) were considered. Impacts resulting from post-production transport, processing, distribution, consumption and all related waste disposal were excluded from the assessment. Regarding the AD scenario, the boundaries were virtually extended to include the production and supply of crops used as feedstock; construction and decommissioning of AD plants and CHP engines; biogas production and conversion and related inputs (consumption of raw materials) and outputs (emissions and Ee); and digestate management. Similarly, all components for the installation of the photovoltaic plant and energy use for the mounting, as well as decommissioning, were included in the PV scenario.

In all scenarios except the baseline one, where the system involves co-production of electricity, multifunctionality was solved by system expansion, it being the first hierarchical choice among the options to manage it according to the ISO standards. Therefore, an environmental credit was considered for the avoided production of electricity, taking the Italian national mix as a reference, substituted by the electricity produced from biogas conversion in the co-generator and from the solar power system.

A sensitivity analysis related to multifunctionality was also performed. This focused on the AD scenario, testing the economic allocation between co-products. The impact of the system was, thus, divided between cattle live weight and electricity produced, based on their relative economic value. The calculations and results of this analysis are reported in the supplementary materials.

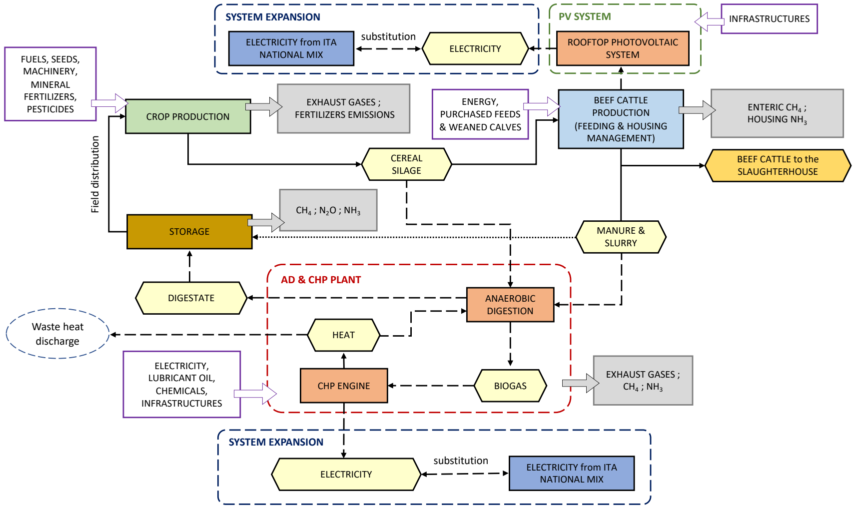


Figure 1 - *Schematic representation of the system boundaries of the study. The main processes and inputs (white boxes), outputs (yellow boxes) and emissions (grey boxes) are reported. The boundaries of the mitigation systems implemented in the alternative scenarios are shown with dashed boxes (PV system in green and AD & CHP plant in red) and the respective flows with dashed arrows. The manure management flow follows the dotted line in the baseline and PV scenarios, being directly stored and subsequently used as organic fertilizer.*

2.3. Inventory analysis and impact assessment

Primary data relating to both crop systems, cattle rearing, and renewable energy plants were collected by means of interviews with farmers and technicians. For the crop production subsystem, data was collected for each crop, including yields, quantities and types of productive factors used, the sequences of field mechanised operations, the agricultural machinery used and their fuel consumption. With regards to cattle, primary data included the number of animals bought and sold per year and their respective live weight, the division of the breeding cycle into feeding phases and their duration, feed consumption (both self produced forages and purchased mineral supplements and feeds) and productive parameters. For the biogas plant, data concerned the installed power, the hours of operation, the energy produced, the biomass ration fed daily, and the type of post-treatment storage. For the PV system, data concerned the technology used, power, the surface area and number of modules installed, and the energy produced.

Inventory data regarding the farm's structure, inputs and outputs are reported in **Table 1**; **Table 2** presents the data regarding the AD & CHP plant and **Table 3** gives data regarding the PV system. More details on crop production inventories and feed compositions are reported in the supplementary materials.

Table 1 - *Main inventory inputs and outputs data relating to the cattle rearing subsystem.*

Parameter	**Unit of measure**	**Value**
Weaned calves - average live weight	kg/head	399.4
Total weaned calves purchased	t/year	963.9
Beef cattle sold – average live weight	kg/head	649.2
Total beef cattle sold	t/year	1811
Share of animal whose waste is handled as slurry	%	71
Share of animal whose waste is handled as manure	%	29
Electricity consumption	MWh/year	81.5
Natural gas consumption	m^3/year	2445
On-farm diesel consumption (excluding field operations)	t/year	19.63

Table 2 - *Design and operating data for the AD & CHP plant considered.*

Parameter		**Unit of measure**	**Value**
Digesters/Postdigester		N	1 + 1
Total digesters' volume		m^3	1800 + 2000
Biomass	Cattle manure	t/day	22

supply	Cattle slurry	t/day	35
	Maize silage	t/day	1.4
Sodium hydroxide		kg/year	134
Electrical capacity		kW	299
Specific volume		m^3/kW	13.3
Process temperature		°C	40.5
Operating time		h/year	7345
Annual Electricity generation		MWh	2196
Electricity self-consumption		%	9.98
Lubricating oil		kg/year	600

Table 3 - *Design and operating data for the PV plant considered.*

Parameter	**Unit of measure**	**Value**
Peak electrical power	949.4	kWp
Total modules area	6640	m^2
Number of modules	4040	-
Annual Electricity generation	MWh	1080

Secondary data mainly concern pollutant emissions from (i) crop cultivation, (ii) cattle rearing and (iii) the AD and CHP plant. These were estimated through models and literature data. More specifically, on-field nitrogen compound emissions due to fertiliser application were computed based on the model proposed by Brentrup et al. (2000), considering climatic data, soil conditions and fertiliser characteristics (manure, slurry, and synthetics), and digestate (for the AD scenario). Phosphate (PO_4^{3-}) emissions were calculated following Prahsun (2006) and Nemecek et al. (2007), by considering two different emission sources: leaching to ground water and run-off to surface water.

Finally, based on Bacenetti and Fusi (2015), a loss of chopped product (maize whole plant) during ensiling was assumed to be equal to 10%.

As for the emissions from the cattle rearing subsystem, the Tier 2 approach from the IPCC guidelines (2006; 2019) was used to estimate methane (CH_4) and dinitrogen monoxide (N_2O) emissions from enteric fermentations and manure management. Ammonia (NH_3) emissions from animal housing and manure management, as well as particulate emissions from housing, were estimated based on the EEA air pollutant emission inventory Guidebook instead (EEA, 2019a). Regarding methane emissions from manure management in the AD scenario, the emission factors for an anaerobic digester were considered under conditions of low leakage, high quality gastight storage, and best complete industrial technology (IPCC, 2019). Where relevant in the models, the 'warm temperate, moist' IPCC climate zone was considered. Further details of the emission estimation process are reported in the supplementary materials.

With regard to the NH_3 emissions from the AD plants, the estimates were made following the EEA air pollutant emission inventory Guidebook (EEA, 2019b), taking into consideration the amount of nitrogen input (both from livestock waste and from other biomass), and pre-treatment storage losses. Emissions from CHP, in the form of the average amount of pollutant emissions per MWh produced, were retrieved from NERI (2010). Some information regarding the modelling of biogas plant infrastructures, such as the lifespan of digesters and CHP, was recovered from Bacenetti et al. (2019), who analysed 10 AD plants in Northern Italy and integrated the inventory of biogas plants.

In the scenarios with the PV system, on the other hand, no changes to beef production or manure management processes were considered. For the PV system infrastructures and all the impacts related to their supply and assembly, a 30-year lifespan was considered, as a standard duration reported in the Ecoinvent® database (Weidema et al., 2013; Moreno-Ruiz et al., 2021).

Background data were retrieved from the established Ecoinvent® database v. 3.8 (Weidema et al., 2013; Moreno-Ruiz et al., 2021). These refer to crop seeds, fertilisers, chemicals, diesel fuel and lubricating oil, agricultural machinery, weaned calves, purchased feed, digester infrastructure, CHP engines, solar power systems and the Italian electricity mix used as avoided products in the alternative scenarios. A list of the main Ecoinvent® processes used is given in the supplementary materials.

In the Life Cycle Impact Assessment phase, all of the collected inventory data were processed and converted into indicators that reflect environmental pressures, as well as resource scarcity. The dataset was characterised by means of the ReCiPe 2016 Midpoint (H) method, version 1.04 / World (Huijbregts et al., 2017), considering eight impact categories. The analysis was performed using SimaPro® LCA software v 9.2 (Pré-Sustainability, 2018).

3. Results and Discussion

Table 4 shows the absolute results of the baseline and AD scenarios, as well as the relative comparisons for the assessed impact categories. For the impact categories affected by the emissions of greenhouse gases and ammonia, an obvious difference between BS-open and BS-cover clearly appeared, indicating the benefit given by the implementation of the AD plant.

The mitigation offered by the installation of the solar power system was minor (**Table 5**): in the PV scenario, the two categories with the greatest reductions were FEP and FSR (-4.5% and -12.7%, respectively); the others, including global warming potential, showed limited reductions of less than 2%.

Finally, the results of the AD and PV scenario, and the relative comparisons with the baselines, are shown in **Table 6**. As expected, this is the scenario where the impact reductions obtained were greater. The trends for this alternative scenario reflected the scenarios where the single mitigation strategy was implemented, with a marked reduction in the improved impact categories, and a marked trade-off for the others (MEP and MSR).

In general, most of the impact categories had an improved environmental performance across the mitigation scenarios, except for marine eutrophication and mineral resource scarcity, which increased in all three alternative scenarios (even if only slightly), with a maximum increase of +1.1%.

Table 4 - *Environmental results of baseline and AD scenarios for the assessed impact categories, with relative variations of the mitigation scenario compared to the baseline ones. Results are expressed per 1 kg of live weight leaving the farm to the slaughterhouse.*

Category	**Unit**	**BS-open**	**BS-cover**	**AD Scenario**	**Delta BS-open vs AD (%)**	**Delta BS-cover vs AD (%)**
GWP	kg CO_2 eq	15.16	14.81	13.57	-10.5	-8.4
ODP	g CFC11 eq	0.120	0.121	0.117	-2.5	-3.3
PMF	g PM2.5 eq	11.95	11.17	9.91	-17.1	-11.3
TAP	g SO_2 eq	74.50	68.14	59.86	-19.7	-12.2
FEP	g P eq	1.10	1.09	0.99	-10.0	-9.2
MEP	g N eq	23.70	23.70	23.93	+1.0	+1.0
MSR	g CU eq	15.14	15.14	15.24	+0.7	+0.7
FRS	kg oil eq	0.55	0.55	0.42	-23.6	-23.6

Table 5 - *Environmental results of baseline and PV scenarios for the assessed impact categories, with relative variations of the mitigation scenario compared to the baseline ones. Results are expressed per 1 kg of live weight leaving the farm to the slaughterhouse.*

Category	**Unit**	**BS-open**	**BS-cover**	**BS-open & PV**	**BS-cover & PV**	**Delta BS-open vs PV (%)**	**Delta BS-cover vs PV (%)**
GWP	kg CO_2 eq	15.16	14.81	14.96	14.62	-1.3	-1.3
ODP	g CFC11 eq	0.120	0.121	0.120	0.121	-0.1	-0.1
PMF	g PM2.5 eq	11.95	11.17	11.77	10.99	-1.5	-1.6
TAP	g SO_2 eq	74.50	68.14	73.89	67.53	-0.8	-0.9
FEP	g P eq	1.10	1.09	1.05	1.05	-4.5	-3.7
MEP	g N eq	23.70	23.70	23.70	23.70	0.0	0.0
MSR	g CU eq	15.14	15.14	15.20	15.20	+0.4	+0.4
FRS	kg oil eq	0.55	0.55	0.48	0.48	-12.7	-12.7

Table 6 - *Environmental results of baseline and AD & PV scenarios for the assessed impact categories, with relative variations of the mitigation scenario compared to the baseline ones. Results are expressed per 1 kg of live weight leaving the farm to the slaughterhouse.*

Category	Unit	BS-open	BS-cover	AD & PV Scenario	Delta BS-open vs AD & PV (%)	Delta BS-cover vs AD & PV (%)
GWP	kg CO_2 eq	15.16	14.81	13.37	-11.8	-9.7
ODP	g CFC11 eq	0.120	0.121	0.116	-3.3	-4.1
PMF	g PM2.5 eq	11.95	11.17	9.73	-18.6	-12.9
TAP	g SO_2 eq	74.50	68.14	59.2	-20.5	-13.1
FEP	g P eq	1.10	1.09	0.94	-14.5	-13.8
MEP	g N eq	23.70	23.70	23.92	+0.9	+0.9
MSR	g CU eq	15.14	15.14	15.31	+1.1	+1.1
FRS	kg oil eq	0.55	0.55	0.36	-34.5	-34.5

Detailed results of the contribution analysis can be found in the supplementary materials. A focussed analysis of the AD and PV scenario is graphically shown in **Figure 2**. The items *feed & bedding*, *energy consumption*, *enteric methane*, *GHG manure management* and *ammonia & PM emissions* refer exclusively to the farming phase managed by the Italian farm under investigation. The impact of forage self-production, including manure field applications, is embedded in the item *feed & bedding*. All of the raw materials and emissions linked to calf production were included within the item *weaned calves* instead. The contribution analysis showed that the mitigation of the GWP impact, due to the AD implementation, occurred because of the combined effect of strongly reducing GHG emissions with respect to conventional manure management and the environmental credit given by electricity co-production. The same drivers were responsible for ODP reduction but to a lesser extent.

The contribution of the AD plant (light blue in the graph) and the PV system (dark blue in the graph) appeared with a negative sign in some impact categories and with a positive sign in others. This is because, in some cases, the environmental credit given by system expansion was greater than the impact given by the inputs and outputs (emissions) of the infrastructure and operation of the AD plant and the PV system, thus generating an overall credit; in other cases, the credit did not offset the impacts. Notably, fossil resource scarcity obtained the greatest benefit in all the alternative scenarios, down to -34.5% of the absolute results per FU in the AD & PV scenario.

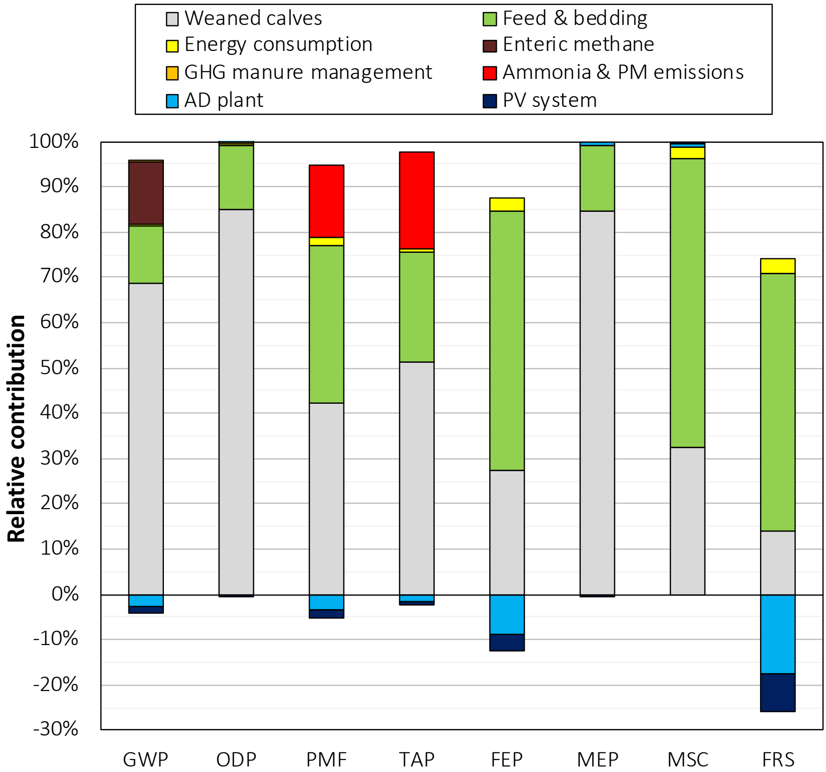
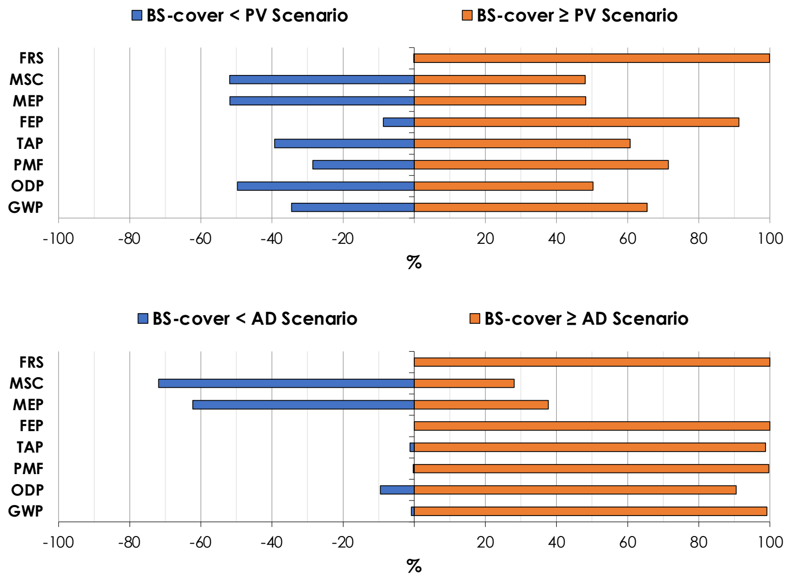


Figure 2 – *Contribution analysis for the AD & PV scenario. (Note: AD – Anaerobic Digestion; PV – Photovoltaic; GWP – Global Warming Potential; ODP – Ozone Depletion Potential; PMF – Particalate Matter Formation; TAP – Terrestrial Acidification; FEP – Freshwater Eutrophication; MEP – Marine Eutrophication; MSR – Mineral Resource Scarcity; FRS – Fossil Resource Scarcity)*

The contribution analysis revealed the supply of weaned calves and feeding during fattening as the two main hotspots for beef cattle production. In the baseline scenarios (**Tables S1** and **S2**), the former dominated the contribution of GWP, ODP, PMF, TAP and MEP; while the latter dominated FEP, MSR and FSR. These impacts remained unchanged in absolute terms in the alternative scenarios, as the plants had no influence on the rearing cycle. For GWP, an average contribution of enteric emissions of about 13% was observed in the baseline scenarios, still unchanged in absolute terms in the alternative ones. GHG emissions from manure management (including N_2O and CH_4) had a share of 8% in BS-open, already reduced to 6% due to the storage coverage in BS-cover and, finally, greatly reduced to less than 1% with the implementation of the AD (its contribution is almost negligible in **Figure 2**). Ammonia and particulate emissions contribute up to 35% to PMF and 36% to TAP in BS-open; reduced to 18% and 22% in the AD scenario, respectively. For these two impact categories, even more than for the GWP, a great reduction was already observable, depending on the management of the slurry in the baseline. The consumption of energy and fuels on the farm during fattening, on the other hand, played a minor role, reaching a maximum of 7% of the share for FRS in the AD & PV scenario. The results in the baseline scenario were in line with other LCA studies carried out in the Italian context: for this production system Bragaglio et al. (2018) observed a value of 17.62 ± 1.78 kg CO_2 eq./kg LW; Berton et al. (2017) reported a lower value of 13.1 ± 0.8 kg CO_2 eq./kg LW. However, since the contribution analysis of the latter is comparable to the present study, the observed differences were likely to have been dependent on the impact assessment method (e.g. 64% of the impact contribution was due to the weaned calves supply from France; 14% and 13% to enteric methane and feed in the fattening Italian phase, respectively). It should be noted that the impact assessment method used in this study also included climate-feedback, attributing a characterisation factor of 34 kg CO_2 eq. to biogenic methane emissions.

To test the robustness of the results obtained, when comparing the different scenarios, a quantitative uncertainty analysis was performed using the Monte Carlo technique (1,000 iterations and 95% confidence interval) as a sampling method. For parameters of the inventory where the distribution was not known, this was estimated based on the data quality pedigree approach, according to Muller et al. (2016); this was the same approach used to construct uncertainty in the database datasets used for the current study. The results are shown in **Figure 3**. The bars represent the probability that the environmental impact of the baseline was greater than, or equal to, the alternatives, while those on the left represent the opposite probability. The orange bars represent the probability that the environmental impact of the baseline is greater than or equal to the alternative scenarios, while the blue bars on the left represent the opposite probability. The results show that there are some trends in the comparison between the PV scenario and the BS-cover scenario, but the only significant difference between the two concerns fossil resource scarcity. On the other hand, when comparing the AD scenario with the BS cover scenario, the results show that the



differences are significant for 5 out of 8 impact categories, except for ODP, MEP and MSC. This confirms the environmental benefits of the AD installation previously presented and that these are not affected by the uncertainty due to data selection from databases, partial model adequacy and data variability.

Figure 3 – Uncertainty analysis results regarding the comparison between Baseline Scenario and Alternative ones. *(Note: AD – Anaerobic Digestion; PV – Photovoltaic; GWP – Global Warming Potential; ODP – Ozone Depletion Potential; PMF – Particalate Matter*

Formation; TAP – Terrestrial Acidification; FEP – Freshwater Eutrophication; MEP – Marine Eutrophication; MSR – Mineral Resource Scarcity; FRS – Fossil Resource Scarcity)

In comparison with previous studies, which analysed the influence of anaerobic digestion implementation in livestock farms, the mitigations observed in the present study were minor. In fact, in Bacenetti et al. (2016), reductions of -22% in GWP, -29% in acidification potential and -18% in eutrophication potential per kg of fat and protein corrected milk were observed for a dairy system with an implemented 300 kW AD plant fed exclusively by livestock waste. The minor reduction observed in the present study suggests that, since the life cycle of beef production generally has a higher carbon footprint than that of milk per product unit, the benefit obtainable due to the implementation of AD is lower in relative terms.

The contribution analysis highlighted the important role of weaned calves within the life cycle impact. This translates into the fact that only a minority share of the impact is directly linked to the Italian fattening farms for most of the impact categories, which reduces their possibilities of intervention in the supply chain for technical-productive and environmental improvements; this is in line with the findings in Berton et al. (2017) and Bragaglio et al. (2018). Linked with this, it is interesting to note that the on-farm energy consumption, despite being equal to an average of 58.4 kWh and 14.07 kg of diesel per head on a farm per year, did not remarkably affected any of the impacts.

Another factor to consider is the continuous increase in the renewables share of the national energy mix, which tends to reduce the impact per kWh for most of the impact categories year by year. In fact, Gargiulo et al. (2020) reported that, by 2030, the carbon footprint of 1 kWh of electricity of the Italian mix could be reduced from the current 0.42 kg CO_2 eq down to 0.36-0.23 kg CO_2 eq, according to different energy and technological transition scenarios and mix evolutions. In an analysis carried out within the setting of this study, this would result in an increasingly reduced environmental credit in the future, as the energy produced replaces an electricity mix that emits between 14% and 45% less CO_2 eq. Indeed, this is a reason to look for further improvements in the management of the AD plant and keep its environmental benefit high. In this sense, one of the energy and environmental improvements overlooked by both policies and plant managers is the recovery and enhancement of the surplus heat generated by the biogas conversion process (Mistretta et al., 2022). Greenhouses, dryers, domestic heating, ORC turbines (Bacenetti et al., 2019; Arslan et al., 2022) and absorption groups are some possible uses of the surplus heat from AD plants. Another innovation in this sector, on which the EU has focussed on recently, is the upgrading of biogas into biomethane, to favour a growing diffusion of this biofuel (European Commission; COM/2022/230 final), which would also be a scenario to be explored, in environmental terms, for agri-food waste-fed AD plants. The strategic planning of AD plants in this direction is required in the coming years, as well as the need for further research (Mallikarjuna et al., 2021; Pappalardo et al., 2022).

At the same time, it must be kept in mind that anaerobic digestion treatment alone does not make manure management sustainable. The best management practices must be applied, starting with the removal of animal housing structures and through to the field distribution, passing through storage and treatments (Saajev et al., 2018; Khoshnevisan et al., 2021). Field distribution is extremely important, in order not to invalidate all the efforts made upstream to avoid GHG and NH_3 emissions (Ricco et al., 2021). The technique, modality, and timing of manure application can lead to important variations within application impacts (Andersson et al., 2023). Future studies could expand this comparative analysis by also adding different manure application scenarios.

Regarding the PV system, even if the environmental and energy potential for the farm was lower, compared to the AD system, barns are perfect for placing solar panels. Such investments will be more and more prioritised under future CAP Strategic Plans (European Commission; COM/2020/381 final). Future studies should investigate the influence of the possible design and operational variables when implementing this on-farm technology.

4. Conclusions and prospects

This work reported the impact of the integration of renewable energy generation systems, namely the anaerobic digestion (AD) of agricultural biomass and waste and photovoltaic (PV) systems installed on barn roofs, on beef cattle farming. The results showed that the on-farm implementation of anaerobic digestion systems generally leads to significant improvements in the environmental and energy impact of beef production. GWP was reduced by 10.5% in the AD scenario, compared to a baseline scenario of conventional manure management and slurry open storage (BS-open), and by 8.4% when compared to an improved baseline scenario with slurry cover storage (BS-cover). These are noticeable reductions, given the high absolute impact of this supply chain when compared with other agri-food products, between 14.81 and 15.16 kg CO_2 eq. per kg of live weight produced in the baseline production scenarios. The mitigation provided by the PV system was more contained, above all for the GWP, but still a further improvement. The most improved impact category concerned the replacement of the use of energy from fossil fuels: fossil resource scarcity is reduced by -35% in the scenarios with both the AD plant and the PV system. Mineral resource scarcity and marine eutrophication potential are the only categories in which trade-offs have been highlighted, albeit very limited.

The main methodological assumption of practicing system expansion, and considering an environmental credit for avoided electricity production, was evaluated with an economic-based sensitivity analysis that showed similar trends in the results. In conclusion, this study quantified the positive environmental effects on the whole beef farming system given by electricity produced by livestock waste anaerobic digestion and photovoltaics, showing good results for both. The limitations of the present study open up opportunities to deepen and broaden our understanding of the topic in future studies: many technical-productive parameters and their combination with different farming systems need to be explored. With regard to AD plants, this study provided some interesting insights into the influence of the plant on the entire farming system; however, the possible variability given by factors, such as the power of the plant and its feeding, need to be better explored. Regarding PV systems on barn rooftops, future studies could provide a deeper comparison between farms at different locations, in terms of irradiance, a factor that strongly influences their energy and, consequently, mitigation potential.

References

Arslan M., Yilmaz C., 2022. Thermodynamic Optimization and Thermoeconomic Evaluation of Afyon Biogas Plant assisted by organic Rankine Cycle for waste heat recovery, Energy, 248, 12348. 10.1016/j.energy.2022.123487

Asem-Hiablie S., Battagliese T., Stackhouse-Lawson K. R., Alan Rotz C., 2019. A life cycle assessment of the environmental impacts of a beef system in the USA. International Journal of Life Cycle Assessment; 24, 441-455. 10.1007/s11367-018-1464-6

Bacenetti J., Fusi A., 2015. The environmental burdens of maize silage production: Influence of different ensiling techniques. Animal Feed Science and Technology, 204, 88 – 98. 10.1016/j.anifeedsci.2015.03.005

Bacenetti J., Bava L., Zucali M., Lovarelli D., Sandrucci A., Tamburini A., Fiala M., 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. 10.1016/j.scitotenv.2015.09.015

Bacenetti J., Fusi A., Azapagic A., 2019. Environmental sustainability of integrating the organic Rankin cycle with anaerobic digestion and combined heat and power generation. Science of the Total Environment, 658, 684-689. 10.1016/j.scitotenv.2018.12.190

Benato, A.; Macor, A. 2019. Italian Biogas Plants: Trend, Subsidies, Cost, Biogas Composition and Engine Emissions. Energies 12, no. 6: 979. https://doi.org/10.3390/en12060979

Berton M., Agabriel J., Gallo L., Lherm M., Ramanzin M., Sturaro E., 2017. Environmental footprint of the integrated France-Italy beef production system assessed through a multi-indicator approach. Agricultural Systems, 155, 33-42. 10.1016/j.agsy.2017.04.005

Bragaglio A., Napolitano F., Pacelli C., Pirlo G., Sabia E., Serrapica F., Serrapica M., Braghieri A., 2018. Environmental impacts of Italian beef production: a comparison between different systems. Journal of Cleaner Production, 172, 4033-4043. 10.1016/j.jclepro.2017.03.078

Brentrup, F., Küsters, J., Lammel, J., Kuhlmann, H., 2000. Methods to estimate on-field nitrogen emissions from crop production as an input to LCA studies in the agricultural sector. International Journal of Life Cycle Assessment, 5(6), 349-357.

Burg V., Bowman G., Haubensak M., Baier U., Thees O., 2017. Valorization of an untapped resource: Energy and greenhouse gas emissions benefits of converting manure to biogas through anaerobic digestion. Resource, Conservation and Recycling, 136, 53-62. 10.1016/j.resconrec.2018.04.004

Burg V., Troitzsch K.G., Akyol D., Baier U., Hellweg S., Thees O., 2021. Farmer's willingness to adopt private and collective biogas facilities: an agent-based modeling approach. Resources, Conservation and Recycling, 167, 105400. 10.1016/j.resconrec.2021.105400

Chalgynbayeva, A.; Gabnai, Z.; Lengyel, P.; Pestisha, A.; Bai, A., 2023. Worldwide Research Trends in Agrivoltaic Systems — A Bibliometric Review. Energies, 16, 611. https://doi.org/10.3390/en16020611

Chirone R., Paulillo A., Salatino P., Salzano A., Cristofalo B., Cristiano T., Campanile G., Neglia G., 2022. Life Cycle Assessment of buffalo milk: A case study of three farms in southern Italy.

Journal of Cleaner Production, 365, 312816. 10.1016/j.jclepro.2022.132816Dinesh H., Pearce J.M., 2016. The potential of agrivoltaic systems. Renewable and Sustainable Energy Reviews, 54, 299-308. https://doi.org/10.1016/j.rser.2015.10.024

EEA (European Environmental Agency), 2019a. EMEP/EEA air pollutant emission inventory Guidebook 2019. Volume 3.B – Agriculture; Manure management.

EEA (European Environmental Agency), 2019b. EMEP/EEA air pollutant emission inventory Guidebook 2019. Volume 5.B.2 – Waste; Biological treatment of waste – anaerobic digestion biogas.

Freitas F.F., Furtado A.C., Piñas J.A.V., Venturini O.J., Barros R.M., Lore E.E.S., 2022 Holistic Life Assessment of a biogas-based electricity generation plant in pig farm considering co-digestion and an additive. Energy, 261 (B), 125340. 10.1016/j.energy.2022.125340

Gargiulo A., Carvalho M.L., Girardi P., 2020. Life cycle assessment of Italian electricity scenarios to 2030. Energies, 13, 3852. 10.1016/j.jclepro.2023.136563Haas R., Duic N., Ajanovic A., Ramsebner J., Knapek J., Zwickl-Bernhard S., 2023. The photovoltaic revolution is on: How it will change the electricity system in a lasting way, Energy, 265, 126351. 10.1016/j.energy.2022.126351

Hollingsworth J.A., Ravishankar E., O'Connor B., Johnson J.X., DeCarolis J.F., 2019. Environmental and economic impacts of solar-powered integrated greenhouses. Journal of Industrial Ecology, 24(1), 234-247. https://doi.org/10.1111/jiec.12934

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., van Zelm, R., 2017. ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. International Journal of Life Cycle Assessment, 22, 138–147. 10.1007/s11367-016-1246-y

Ingrao C., Bacenetti J., Adamczyk J., Ferrante V., Messineo A., Huisingh D., 2019. Investigating energy and environmetal issues of agro-biogas derived energy systems: A comprehensive review of life cycle assessments. Renewable Energy, 136, 296-307. 10.1016/j.renene.2019.01.023

ISMEA (Istituto di Servizi per il Mercato Agricolo Alimentare), 2021. Allevamento bovino da carne – Scheda di settore (Beef cattle – Sector report) 2020. https://www.ismeamercati.it/flex/files/1/a/2/D.2e8af1c92fc3e2b3b1fa/Skeda_bovino2021.pdf (accessed 9/2022).

IPCC (Intergovernmental Panel on Climate Change), 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and other land use.

IPCC (Intergovernmental Panel on Climate Change), 2019. 2019 Refinement to the 2006 Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and other land use.

ISO 14040, 2006. Environmental management — Life cycle assessment — Requirements and guidelines. International Organization for Standardization. Geneva, Switzerland.

ISO 14044, 2018. Environmental management — Life cycle assessment — Principles and framework. International Organization for Standardization. Geneva, Switzerland.

Khoshnevisan B., et al., 2021. A critical review on livestock manure biorefinery technologies: Sustainability, challenges, and future perspectives. Renewable and Sustainable Energy Reviews, 135, 110033. https://doi.org/10.1016/j.rser.2020.110033

LEAP, 2016. Environmental performance of large ruminant supply chains – Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. FAO, Rome, Italy. Lijó L., González-García S., Bacenetti J., Moreira M.T., 2017. The environmental effect of substituting energy crops for food waste as feedstock for biogas production, Energy, 137, 1130-1143. 10.1016/j.energy.2017.04.137

Lovarelli D., Falcone G., Orsi L., Bacenetti J., 2019. Agricultural small anaerobic digestion plants: Combining economic and environmental assessment. Biomass and bioenergy, 128, 105302. 10.1016/j.biombioe.2019.105302

Mallikarjuna C., Dash R.R., Bhunia P., 2021. Sustainability of Gaseous Biofuels: Potential Uses Technological Constraints and Environmental Concerns. In: Circular Bioeconomy - Current Status and Future Outlook, 247-268. https://doi.org/10.1016/B978-0-12-821878-5.00025-8

McClelland S.C., Arndt C., Gordon D.R., Thoma G., 2018. Type and number of environmental impact categories used in livestock life cyle assessment: a systematic review. Livestock Science, 209, 39-45. 10.1016/j.livsci.2018.01.008

Mistretta M., Gulotta T.M., Caputo P., Cellura M., 2022. Bioenergy from anaerobic digestion plants: energy and environmental assessment of a wide sample of Italian plants. Science of the Totale Environment, 843, 157012. https://doi.org/10.1016/j.scitotenv.2022.157012

Mohsenimanesh A., LeRiche E.L., Gordon R., Clarke S., MacDonald R.D., MacKinnon I., VanderZaag A.C., 2021. Review: Dairy farm electricity use, conservation, and renewable production – a global perspective. Applied Engineering in Agriculture. 37(5): 977-990. 10.13031/aea.14621

Moreno Ruiz E., FitzGerald D., Symeonidis A., Ioannidou D., Müller J., Valsasina L., Vadenbo C., Minas N., Sonderegger T., Dellenbach D. (2021). Documentation of changes implemented in ecoinvent database v3.8. ecoinvent Association, Zürich, Switzerland.

Muller S., Lesage P., Ciroth A., Mutel C., Weidema B., Samson R., 2016. The application of the pedigree approach to the distributions foreseen in ecoinvent vs. The International Journal of Life Cycle Assessment, 21, 1327-1337. https://doi.org/10.1007/s11367-014-0759-5

Nemecek, T., Kägi, T., 2007. Life cycle inventories of agricultural production systems, data v2. 0, Ecoinvent report (No. 15). Ecoinvent Center, Zürich and Dübendorf Switzerland.

NERI (NATIONAL ENVIRONMENTAL RESEARCH INSTITUTE, Aarhus University); 2010. Emissions From Decentralized CHP Plants, p. 113. http://www2.dmu.dk/pub/FR786.pdf

Notarnicola, B., Sala, S., Anton, A., McLaren, S. J., Saouter, E., Sonesson, U., 2017. The role of life cycle assessment in supporting sustainable agri-food systems: A review of the challenges. Journal of Cleaner Production, 140, 399-409. 10.1016/j.jclepro.2016.06.071

Pappalardo G., Selvaggi R., Pecorino B., 2022. Biomethane production potential in Southern Italy: An empirical approach. Renewable and Sustainable Energy Reviews, 158, 112190. https://doi.org/10.1016/j.rser.2022.112190

Pascaris A.S., Handler R., Schelly C., Pearce J.M., 2021. Life cycle assessment of pasture-based agrivoltaic systems: Emissions and energy use of integrated rabbit production. Cleaner and Responsible Consumption, 3, 100030. 10.1016/j.clrc.2021.100030

Poore J., Nemecek T., 2018. Reducing food's environmntal impacts through producers and consumers. Science, 360, 6392, 987-992. 10.1126/science.aaq0216

Prahsun, V., 2006. Erfassung der PO4-Austrage fur die Okobilanzierung SALCA Phosphor. Agroscope Reckenholz – Tanikon ART, pp. 1–20.

Pré-Sustainability, 2021. SimaPro LCA software v. 9.2. https:// simapro.com/. Rivera Huerta A., Güereca L.P., Rubio Lozano M.S., 2016. Environmental impact assessment of beef production in Mexico through life cycle assessment. Resource, Conservation and Recycling, 109, 44-53. 10.1016/j.resconrec.2016.01.020

Ricco C.R., Finzi A., Guido V., Riva E., Ferrari O., Provolo G., 2021. Evaluation of ammonia emissions from filtration of digestate used for fertigation. Journal of Agricultural Engineering, 52,3. https://doi.org/10.4081/jae.2021.1187

Saajev E.P.M., Winiwarter W., Amon B., 2018. Greenhouse gas and ammonia emissions from different stages of liquid manure management chains: abatement options and emission interactions. J. Environ. Qual. 47:30–41. 10.2134/jeq2017.05.0199

Weidema, B.P., Bauer. C., Hischier, R., Mutel, C., Nemecek, T., Reinhard, J., Vadenbo, C.O., Wernet, G., 2013. Overview and methodology. Data quality guideline for the ecoinvent database version 3. Ecoinvent Report 1(v3). St. Gallen: The Ecoinvent Centre.Wu W., Cheng L.-C., Chang J.-S., 2020. Environmental life cycle comparison of pig farming integrated with anaerobic digestion and algae-based wastewater treatment. Journal of Environmental Management, 264, 110512. 10.1016/j.jenvman.2020.110512

Wu W., Cheng L.-C., Chang J.-S., 2020. Environmental life cycle comparison of pig farming integrated with anaerobic digestion and algae-based wastewater treatment. Journal of Environmental Management, 264, 110512. https://doi.org/10.1016/j.jenvman.2020.110512

CHAPTER 5 – General discussion & conclusions

In this thesis, focus have been made on some environmental impacts of livestock farming, namely global warming and air pollution, whose solution is more urgent than ever. Intensive livestock production across Europe faces the challenge of meeting increasingly stringent environmental standards and public and retailers and consumers opinion. It must be emphasized that these are not the only environmental impacts of the livestock sector, there are many others, both positive and negative, which have not been studied as a matter of both scope and time.

Insights have been provided into the complex relationship between livestock production, environmental impacts, and mitigation strategies in Italy. As discussed, there are different approaches to (i) measure, (ii) avoid, (iii) reduce these impacts. The selected papers dealt with environmental assessments, carried out through different approaches, of various mitigation strategies applicable to different steps of the supply chain. The focus was in particular on the two main sources of direct emissions from intensive livestock systems, namely (i) housing facilities and (ii) livestock waste, and the structures intended for its management.

An important topic covered was that of air treatment in pig housing facilities. Despite some trade-offs, the mitigation potential for the European Union deriving from the widespread implementation of studied air treatment technologies (wet and dry scrubbers) is very large, if below the best operational performance, as was highlighted in the Papers 2 & 3. Focusing on Italy, according to Eurostat, the majority (about 80%) of pigs and sows are kept on farms with more than 1000 heads, reflecting intensive farming practices. For both environmental and economic reasons, it is reasonable to assume that it is precisely large farms that could be affected by the future introduction of air treatment technologies. As a result, there are huge transfer opportunities for the tested technologies, which can affect up to thousands of pig farms and millions of pig heads, leading to significant and widespread reductions in air pollutant emissions across the country, fostering air quality improvement. There is ample room for improvement in scrubber efficiency to achieve greater emission reductions on the one hand and to optimize the use of consumables on the other hand, which would be crucial especially for the wet scrubber to limit trade-offs. In conclusion, scrubbers are both environmentally interesting technologies and can provide benefits in areas where ammonia emissions and particulate matter formation are locally relevant issues. When considering the balance between emissions avoided and trade-offs created, the dry scrubber was found to be the best solution. However, these are technologies that can contribute to the reduction of air pollution, but they are not intended to be the only solution to the environmental impact of pig farming, which still requires various interventions at different levels of the supply chain.

It should be underlined that no statistical tests were performed to verify the significance of the differences observed between the baseline and mitigation scenarios analyzed in both papers 2 & 3. In both cases, sensitivity analyses were used to assess the robustness of the results. The latter are commonly used in environmental assessments to verify the effect of varying the inventory data or methodology on the results; in fact, they allow to highlight the key factors that contribute most to determining the results, and their influence. However, the application of statistical analyses such as a Monte Carlo simulation would provide a consolidated confirmation of the trends observed, thus increasing the reliability, objectivity and interpretability of the results. Therefore, this represents an important possibility for improvement regarding future developments based on these publications.

Another important issue in areas with high livestock densities is the large amount of livestock waste generated that needs to be managed. In an area that is already fragile due to its intrinsic geopedological characteristics, such as northern Italy, this condition can exacerbate environmental pressures on soil and freshwater. For this reason, the efficient and careful handling of manure and slurry at each step of the chain is extremely important and must be supported by the implementation of techniques and technologies aimed at (i) preventing nutrient losses through emissions and (ii) optimizing field efficiency to make the most of their fertilizing and soil improving properties. Papers 4 & 5 explored this topic, albeit with different declinations, both in terms of subtopic, as the former focused on additives to be added to raw slurry stored in outdoor tanks, and the latter mainly on anaerobic digestion of livestock waste and its benefits, and in terms of analytical approach.

Precisely regarding the analytical approach, advantages and disadvantages to conducting direct measurement campaigns have emerged and been discussed. On the positive side, direct measurements provide highly accurate and precise data, enabling a detailed understanding of emission sources, trends, and their spatial distribution. Direct measurements are essential and necessary to verify the effectiveness of innovative emission reduction strategies, such as that tested in Paper 4, for which there are no literature references or models available. This information is invaluable for designing effective mitigation strategies and policies. However, there are also drawbacks to consider. Direct measurements can be resource intensive. They have limited spatial coverage, making it difficult to fully capture emissions from large areas. In this regard, it is important to note that the measurement campaign carried out for Paper 4 concerns a single dairy farm, and the results obtained can be extrapolated to other realities with extreme caution. The first possibility for improvement to integrate and enrich the study would be the development of an LCA case study of the cattle farm where the experimental tests took place. The result would be a more comprehensive assessment of the environmental sustainability of the tested additive. In this way, in fact, it would be possible to obtain not only the reduction in emissions relating to the slurry tanks alone but also the reduction that this entails in overall terms for the entire farm, and consequently of its output products, also having the possibility of evaluating the impact of direct (example: animal waste management) and indirect (fertilizer management) operational changes caused by the use of the additive. Future improvements could also include the use of a combination of direct measurements, remote sensing, and modeling to gain a holistic understanding of emissions and air quality, as well as a broader effect of the mitigation measure studied.

In Paper 2 and Paper 5, the introduction of technologies to mitigate the impact of livestock farms, either through emission abatement or through waste treatment and renewable energy generation, was evaluated through the application of LCA. This tool is able to provide valuable information on the environmental impacts of the livestock sector. First, it plays a critical role in promoting a comprehensive understanding of supply chains, processing methods and delivery systems, shedding light on the complexities of how products or services move through these systems. LCA also excels at identifying areas within the value chain that need improvement, providing information on potential efficiency gains and reductions in environmental impact. It is also effective in highlighting trade-offs, ensuring that improvements in one aspect of a product's life cycle do not inadvertently shift environmental burdens to another phase or impact category. Finally, LCA can support benchmarking and facilitate the substantiation of environmental claims, provided it is applied under certain standards. However, LCA is not without limitations. One major drawback is its inability to address missing impact pathways, such as biodiversity loss, which are difficult to quantify and incorporate into LCA models. LCA does not provide direct operational guidance to companies seeking to implement environmentally friendly practices, as it is primarily an analytical rather than prescriptive tool. It also fails to address complex ethical issues typically linked to this sector such as animal welfare, as these factors often go beyond the scope of conventional environmental assessments. All of these represent current gaps in the method, as well as opportunities for improvement and future study.

Establishing data collection protocols and results reporting would enhance the use of LCA in livestock, the reliability of the studies and the comparability of evaluations between products, processes (such as mitigation practices) and scenario analyses. Integrating LCA within policy frameworks (for example represented by certification or incentive systems) relating to this sector with strict and clear guidance could push a growing adoption and standardization at the same time of the methodology.

Speaking of standardization, there would be a need for clear guidelines on (i) representing the dynamic nature of livestock systems (e.g. how to estimate the average annual population on a farm, or how to account for animals that change weight during the year); (ii) representing the geographical variability of livestock systems (e.g. when to prefer specific localized rather than general inventory data and/or characterization factors); (iii) defining the appropriate system boundaries and scope for a livestock LCA (e.g. which processes and impacts should be mandatorily included/excluded, which cut-off approach to use); (iv) defining the appropriate way of collecting secondary and background data (e.g. binding emission estimation calculations, prioritizing measured data); (v) defining the appropriate way of collecting secondary and background data (e.g. which processes and impacts should be mandatorily included/excluded, which cut-off approach to use).); (iv) defining the appropriate way of collecting secondary and background data (e.g. binding emission estimate calculations, prioritizing measured data over estimated data, somehow limiting the selection of the most relevant background databases to the detriment of the outdated/unreliable/out-of-scope); (v) defining the appropriate life cycle impact assessment methodology (e.g. which categories to include, which characterization methods to include for those categories where more than one is possible).

In conclusion, there is ample room for refinement of the method and research in this direction. However, despite the limitations highlighted and discussed, the LCA methodology is already proving to be sound for the analysis of livestock production and will be increasingly adopted. As environmental awareness and demand for ethical and environmentally friendly food choices continue to grow, LCA plays a crucial role in shaping the sector and reporting.

Technological advances, such as precision agriculture and precision livestock farming, will allow for more accurate and complete assessments of the sector, thanks to the ability to collect an increasing amount of data with great accuracy. This will not only help to optimize the environmental performance, but also to improve animal welfare and minimize negative externalities. A challenge remains also to integrate the social and economic dimensions.

This work underscores the importance of a holistic approach to assessing the environmental impacts of livestock production. A key challenge is the need to capture the complexity and variability of farming systems, where local practices and conditions vary widely. LCA provides valuable insights into the environmental footprint of livestock production, but due to its limitations highlighted in the studies presented and in the discussion above it cannot stand alone in guiding decision making and environmental policy development. It should be complemented by broader considerations, given the multifaceted nature of the sector, including techno-productive, social and economic data that consider sector-specific, time-sensitive and geographically specific factors. A relevant way to achieve this goal is to use a multicriteria decision analysis (MCDA) method, which explicitly considers multiple criteria to help individuals or groups explore relevant decisions. It can combine objective measurements and value judgments using quantitative or qualitative indicators, makes subjectivity explicit, and manages that subjectivity by organizing stakeholder input, helping decision makers choose between more sustainable options or scenarios, and fully evaluating and balancing the positive and negative consequences of each. Further research and policy support on the application of this approach are essential to promote comprehensive assessments to touch all relevant facets of an analyzed system and fully evaluate the possible trade-offs. This, in turn, when applied to the sector under study, could lead to the widespread adoption of innovative mitigation strategies and technologies such as those analyzed in this thesis, and to consecrate the benefits of those mitigation strategies and technologies that already exist and have been adopted, and to promote their potential for expansion.