Reducepi

Development of a tool aimed at assessing the environmental impact of pig slurry management life cycle

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Preamble

This document serves as an interim report for my Master Thesis Project, which is an integral part of the broader initiative known as "Reducepi." This collaborative research project is conducted jointly by the *Universitat Autònoma de Barcelona (UAB)* and *Inèdit*, a Barcelona-based company specializing in LCA analysis, eco-design, and sustainability consultancy, which has been subcontracted for this specific project. The *UAB* team was assigned the responsibility of developing a model to estimate slurry composition based on different types of diet, and *Inèdit* with modeling the environmental impacts for various treatments selected from commonly used practices in Catalonia.

Abstract

Pig slurry, a byproduct of intensive pig farming, poses a significant environmental threat due to its high nutrient content and greenhouse gas emissions. This issue is particularly concerning in Spain, one of the EU countries with the largest pig herds. This situation has propelled the need to develop user-friendly tools that enable stakeholders to estimate the environmental impacts of the management of pig slurry without requiring a deep scientific background. In this master thesis, a first-of-its-type LCA tool was developed to estimate emissions from pig slurry management thorough all its value-chain, from storage to transport through the application of targeted treatments, and finishing with the landspreading of the resulting products. Simapro and Excel were used to model all the life cycle stages, and data was obtained both from available literature and experimental data, which was provided by the Reducepi project collaborators. Specifically, experimental data consisted on slurry composition of pigs of 54 kg and 70 kg fed with a control feed (diet-based feed) and an alternative feed (sorghum-based feed), and results were tested against three treatment scenarios: anaerobic digestion, composting and no treatment (control group). Results showed that, for all the impact categories considered, the scenario composting was by far the most impacting treatment compared with anaerobic digestion and no treatment.

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Contents

List of Figures

List of Tables

Disclaimer

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Introduction

Over the second half of the XX century, countries like Spain and Denmark experienced a transition from small pig farm holdings to large scale operations that have profoundly increased environmental impacts in a local and regional scale. This shift was part of a broader modernization of the agricultural sector and was marked by a replacement of traditional productive factors (such as land and labor) by capital investment, technological developments and efficient public policies aimed at ensuring food security and at integrating production into a competitive, global market economy (Symes & Jansen, 1994). The shift from the traditional *laizez-faire* way of the agricultural sector to an interventionist approach stemmed from the need of food security in a post-war Europe, where institutional concern led to governments recognize the strategic importance of the agricultural sector (Martin-Retoncillo & Pinilla, 2015). With the signing of the Treaty of Rome in 1957 and later, the ratification of the Common Agricultural Policy (CAP), national agricultural policies were diluted in pose of an European shared framework (Clar et al., 2018), which paved the way to the development of an intensive, production-centered European agricultural industry. Today, agriculture stands as the predominant land use in the EU-27, encompassing nearly 45% of its entire land area (Comission, 2020) and contributing up to 1.3% of the total EU GDP in 2019 (Eurostat, 2022).

However, this came at the expense of significant environmental challenges; inappropriate farming practices such as excessive fertilization, improper use of pesticides, and the use of heavy machinery have lead to acidification, nitrification, desertification, contamination (e.g., by heavy metals and agrochemicals), compaction, and erosion of soils (Alexandridis et al., 2018). This contributed greatly to the emission of greenhouse gases (GHGs) and other contaminating particles, but also had profound consequences for environmental services such as carbon sequestration, nutrient cycling, soil structure and functioning, water purification, and pollination. In addition, the intensive use of chemical fertilisers leached nitrate and phosphate into the groundwater, contributing to its deterioration (Stoate et al., 2009). In order to address these environmental impacts, the European Union (EU) has put forward over the years different initiatives and regulatory frameworks like the Nitrates Directive, the Water Framework Directive, the Sustainable use of Pesticides Directive, the Birds and Habitats Directives and the Effort Sharing Decision, among others.

5.1 Pig production in the EU

In 2021, the EU livestock population amounted to 289 million, from which 142 million accounted for pigs. Between 2001 and 2021, the EU's total livestock count for pigs fell by an estimated 9% even though more meat was produced. A majority of this livestock was held in just a few EU Member States, and some of them like Spain and Denmark are relatively specialised in terms of livestock farming; Spain accounted for one fifth while Denmark one tenth of total production (Eurostat, 2022). In 2022, the pig meat sector accounted for 8.5% of the overall agricultural output in the EU-27, the highest proportion among all meat sectors, and contributed 35% to the total meat production (European Comission, 2022). Regarding exportation, Europe holds a significant position as a major global pork producer. In 2016, the EU produced more than 20% of the world's pork meat (K.H.J. et al., 2016). In 2022, it became the second biggest producer of pig meat, only behind China, and it exported about 13% of its total production, most of it being exported to East Asia, particularly China (European Comission, 2022). Currently, it is the biggest exporter of pork and pork-products worldwide (European Comission, 2022).

The EU pig meat sector is regulated by various legislative acts regarding food safety, public and animal health, environmental protection and animal welfare throughout the whole production chain. Directive 2010/75/EU on industrial emissions applies to intensive pig rearing installations with more than 2000 places for pig production by establishing best available techniques (BAT) that cover from nutritional management to storage and processing of manure and storage of dead animals (Marie-Laure, 2020). Nitrogen emissions from pig slurry falls under the Directive 91/676/EEC, which aims to mitigate water pollution resulting from agricultural nitrate use by monitoring nitrate concentrations in water bodies, designating nitrate vulnerable zones, and implementing codes of good agricultural practices along with measures to prevent and reduce water pollution caused by nitrates (Comission, 1991). Also, Directive 2016/2284/EC on the reduction of national emissions of certain atmospheric pollutants covers contaminants related with the pig industry such as nitrogen oxides (NOx), non-methane volatile organic compounds (NMVOC), ammonia (NH3) and fine particulate matter (PM2,5) (IEA, 2022; LNV, 2020). Within the CAP, the European pig market falls under the Common organisation of agricultural markets (CMO) regulation, which ensure incomes for farmers and a continued supply for European consumers (EUR-Lex, 2023; Marie-Laure, 2020), and pig meat is subject to Regulation 178/2002, namely General Food Law Regulation.

5.2 Pig production in Spain

The agricultural sector plays a pivotal role in the Spanish industrial landscape, holding significant social, territorial, environmental, and economic importance. Approximately 50% of Spain's total land area is dedicated to agricultural and livestock activities (La Moncloa, 2017), and Spain stands as one of the major global producers of fruits and vegetables, especially in the production of olives, wine, cereals, and derived products. In 2015, approximately half of the total agricultural production from Spain found its way to international markets, with a significant 92% of that being directly exported to the EU. Regarding livestock production, Spain is the EU member with the biggest pig herd. The pig industry particularly exemplifies a production approach geared towards exportation, as it is currently the country's leading livestock sector in terms of net exportation (La Moncloa, 2017). Spain boasts the largest pig herd among all EU Member States, reaching 31 million heads in 2018 (Marie-Laure, 2020), and secured the position of the second-largest producer of pig meat in the EU, trailing only behind Germany (European Comission, 2022). The distribution of the Spanish pig herd is notably uneven, with the most populated areas collectively hosting approximately 50% of the total Spanish pig livestock census. In 2023, Spain's swine sector experienced a downward trend in swine and pork production due to lower pork exports to non-EU markets (mainly to China), continued high input costs arising from the Ukraine-Russian war and Isreal-Gaza conflict, and imposition of new regulations such as the Royal Decree 159/2023 on Animal Welfare. Production was also affected by several Porcine Reproductive and Respiratory Syndrome (PRRS) outbreaks in Catalonia and Aragon (Valverde, 2023).

Yet, the current scenario is vastly different from the past. Spain underwent one of the most significant shifts in meat production within Western Europe, specially pig meat. The increase was part of a broader plan of Franco's Regime to make meat more affordable to the ordinary citizen. This was achieved by introducing new animal breeds and large amounts of cereals (soya and corn) from the United States during the 1970's, which opened the door to foreign capital investments that enabled a rapid implementation of a vertically-integrated agroindustrial model based on cooperation between large transnational companies and small Spanish businesses (Clar et al., 2018). This gave rise to a trend towards industrial concentration that led to a reduction of production prices due to economies of scale (FWW, 2017); between 1987 and 1997, the number of farm holdings fell by 75% at the expense of a livestock herd growth of 225% (Clar et al., 2018). Despite the major production and reduction of production costs, this resulted in worsening labour conditions and in rising impacts in the environment and in animal welfare (FWW, 2017). If traditional farmers were constrained by the quantity of livestock their croplands could support (Thorne, 2007), now they produce such large quantities of manure that restrict their use as a fertiliser (Hollas et al., 2021). Spanish farmers find themselves in the dilemma of: complying with regulations by reducing livestock as a mechanism for reducing emissions in a regime that forsakes to comply production quotas; or enhance or sustain production in order to obtain higher yields and afterwards have sufficient capital to cover possible penalties in the event of exceeding the permitted nitrogen limits (Barbeta-Viñas & Requena-I-Mora, 2022). Therefore, there is an increasing need to assess the potential revalorisation of this waste, reducing the environmental impacts associated with its management while maintaining the economic competitiveness of the industry.

5.3 Environmental impacts of pig production

Modernization of agriculture has endured food security and eradicated chronic famines worldwide. However, this was only possible with the advent of artificial nitrogen fixation technologies at the beginning of the twentieth century, like the Haber-Bosch synthesis. This advancement, though transformative, came at the expense of altering in depth the nitrogen cycle. Nowadays, reactive nitrogen production rates by humans have doubled the rates of natural nitrogen fixation (UN, 2023); this has lead to eutrophication due to nutrient overloading, acidification from nitrogen deposition, groundwater contamination, and GHGs emissions, among others. In addition, the use of heavy machinery during crop production, as well as the distribution of goods to consumers, are responsible for emitting a substantial quantity of GHGs into the atmosphere, thereby contributing to climate change.

This outgrow in food production paved the way for an enhancement in livestock production, coming with the caveat of having to dispose larger volumes of faeces while the available space for that has been reduced (OECD, 2003). For example Catalunya, which is well-known for being one of the Spain's major centers of swine production, has a 39,9% of its land categorized as nitrogen sensible according with the established by the European's Water Directive and Nitrate Directive (ACA, 2023). In consequence, water quality has been compromised specially in nearby production areas (OECD, 2003), and the resulting water needs to be treated before human consumption or other uses.

Furthermore, manure landspreading without any pretreatment can lead to potential emissions of nitrous oxide (N_2O), which has the highest warming potential amongst all GHGs.

Nitrous oxide is formed amidst denitrification, and can escape to the atmosphere prior being reduced to nitrogen gas. Other compounds that can escape to the atmosphere are ammonia (NH_3), which can volatilize and further be degraded by photo-chemical reactions to other compounds that form part of what is known as nitrogen deposition. This deposition is well-known to reduce biodiversity by giving advantage to plants more nitrogen-tolerant, and can lead to variations in the pH of water bodies that, in turn, can alter their ecosystems after reaching buffer capacity.

In this sense, a great tool that has been found to be effective at assessing the environmental impacts of pig production is the LCA methodology.

5.4 Objectives

The present study aims at developing a tool that quantifies the environmental impacts associated with pig-slurry treatment technologies currently in use in Catalonia using the Life Cycle Assessment (LCA) method based on the composition of the manure.

The treatments selected as a basis for our study have been selected following the criteria presented by Prenafeta and Parera, 2020: anaerobic digestion, composting and solid-liquid separation.

The specific objectives of the study are:

1. Identify the stages in each treatment process that generate the highest environmental impacts.
2. Identify targeted optimization measures within the manure treatment process.

3. Identify how different diets affect the environmental impacts of swine slurry management.

Theory

6.1 Life cycle assessment (LCA) methodology

The Life Cycle Assessment (LCA) is a method used for quantifying the environmental impact of a product or service thorough its life cycle, from the extraction and processing of raw materials through manufacturing and distribution, to its use, recycling, and final disposal. This tool is essential for identifying improvement opportunities in the environmental performance of products over their lifespan, providing valuable information to decision-makers in industry, governments, and non-governmental organizations.

The LCA methodology facilitates decision-making regarding environmental sustainability throughout the entire production chain in a more informed and objective manner. Including the entire life cycle and supply chain prevents the transfer of environmental impacts between different phases or processes of the life cycle instead of reducing them. This not only strengthens environmental responsibility at the company level but can also lead to improvements in efficiency and competitiveness in a global market where sustainability is a key differentiating factor.

6.1.1 Stages of an LCA

In accordance with ISO 14040 and as shown in Fig. 6.1, there are four sections that are compulsory in any LCA study:

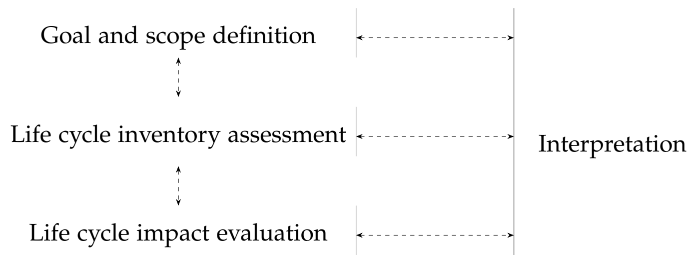

Figure 6.1: Stages of the LCA methodology and its interrelations, from ISO 14040:2006.

A detailed description of each of these steps is presented below.

6.1.1.1 Goal and scope definition

In order for an LCA to provide relevant and actionable insights for decision-making, both the Goal and the Scope Definition must be specified. The Goal defines what the LCA aims to achieve, and ensures that the following steps are aligned with it. It can range from e.g., evaluating the environmental impact of a product's life cycle to comparing the impact between two production methods, or identifying key areas for improvement in a specific manufacturing process.

Once this is done, the scope of the assessment shall be defined. This includes the system boundaries and the level of detail. The system boundary defines the limits of the LCA and which stages of the life cycle are considered, and it is typically indicated in a diagram. Often, practitioners work with three type of system boundaries:

1. Craddle to Gate: from the extraction of raw materials (craddle) to the moment when

the product leaves the manufacturing process (gate). Therefore, it does not include any disposal phase.

2. Gate to Gate: from the input gate of a process or sequence of processes to the output gate. It is typically narrowed to analyse specific segments of a production chain.

3. Craddle to Grave: from raw material extraction to through manufacturing, distribution, use, and end-of-life disposal or recycling. It provides with a whole picture of the environmental impacts within the entire life cycle of a product.

By establishing a system boundary, the LCA can focus on the relevant processes directly associated with the life-cycle of a product or service. It is noteworthy that the selection of the system boundary must be consistent with the goal of the study, and the criteria used in its establishment shall be identified and explained, as well as the level of detail in which the unit processes are studied.

ISO 14040:2006 defines unit processes as the "smallest element considered in the life cycle inventory analysis for which input and output data are quantified". Each unit process should be initially described to define where the unit process begins (in terms of the receipt of raw materials or intermediate products), the nature of the transformations and operations that occur as part of the unit process, and where the unit process ends, in terms of the destination of the intermediate or final products. Inputs and outputs of every unit process shall be initially identified and afterwards more carefully curated after additional data is collected during the course of the study. A cut-off criteria is established to exclude inputs and outputs that are not taken into consideration, and the assumptions on which this cut-off is established must be clearly stated and described in the final report. Cut-off value can be based on the quantity of material or energy used or the environmental significance of certain components (e.g., excluding components whose impact is less than 1% of the total impact of the system).

In addition, during this step, the functional unit must be also defined (both in terms of inventory and environmental impacts) which, according to ISO 14040/44, can be defined as the "quantified performance of a product system for use as a reference unit". Therefore, it serves as a scaling factor for all calculations and therefore, it is integral to the outcome of the LCA.

6.1.1.2 Life cycle inventory Analysis (LCI)

The definition of the goal and scope of a study provides the initial plan for conducting the life cycle inventory phase of an LCA.

It starts with the data collection step, where both quantitative and qualitative data for each unit process is collected. In the case of calculated data, calculation processes must be explicitly documented and consistently applied throughout the study. Additionally, any assumptions made should be clearly stated and explained, and the same calculation procedures should be consistently applied throughout the study. In the case of data that has been collected from public sources, it shall be be referenced.

During data collection, a data validity shall be conducted to provide evidence that the data is of high-quality. This validation may involve establishing e.g., mass balances, or energy balances, since each unit process must obey the laws of conservation of mass and energy.

In some cases, it's common to face scenarios where a single unit process generates not just one primary product but also additional secondary products known as "co-products". Co-products differ from byproducts in that they are intentionally produced alongside the main product and have significant value. In the dairy industry, for instance, while the primary focus may be on milk production, the meat generated from cows also holds economic value and is typically utilized in the meat industry. This is particularly the case of industries where multiple outputs are produced simultaneously from a shared production process. Consider, for instance, the production of milk. The production of milk and meat stem from the same unit process of raising and maintaining dairy cows. When cows are raised for milk, they also produce meat as a byproduct. In this case, milk would be the primary product while meat would serve as a co-product. The understanding of co-products is essential in LCAs as it influences the environmental impacts' allocation among different products.

Wherever possible, the allocation shall be avoided by 1) dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes or 2) expanding the product system to include the additional functions related to the co-products. In the case of milk production, this can be achieved by breaking down the unit process of raising dairy cows into distinct sub-processes e.g., separate data can be gathered for the feed production, cattle breeding and management, milk extraction, and meat production stages and thus, avoiding the need of allocation; or alternatively, the system can be expanded to encompass all the relevant processes and functions associated with both milk and meat production by considering both milk and meat outputs within the same system boundary.

If allocation is unavoidable, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the underlying physical relationships between them; i.e. they should reflect the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system. In the case where physical relationship alone cannot be established or used as the basis for allocation, the inputs should be allocated between the products and functions in a way that reflects other relationships between them e.g., input allocation between co-products in proportion to the economic value of the products.

In other cases, outputs may be partly co-products and partly waste. In such cases, it is mandatory to identify the ratio between them as inputs and outputs shall be allocated to the co-products part only.

6.1.1.3 Life cycle impact assessment (LCIA)

The Life Cycle Impact Assessment (LCIA) is the third stage of the LCA, and consists in estimating the potential environmental impacts derived from the results of the LCI. LCIA differs from other type of impact assessment techniques such as the environmental performance evaluation or the environmental impact assessment in the sense that it is an approach based on a functional unit. The result is a series of environmental impact indicators for different impact categories representing specific environmental issues, such as climate change or ozone layer depletion.

The LCIA should include the following mandatory elements:

Selection: it consists in the selection of the category indicators, which can be midpoint categories or endpoint categories.

1. **Midpoint category indicators** provide detailed indicators of environmental impacts based on specific stressors or factors such as GHG emissions, acidification, eutrophication, etc. that are often expressed in physical units (e.g., kg CO_2-eq, mol H^+-eq). Examples of midpoint categories are Global Warming Potential, Acidification, Eutrophication, Ozone Depletion, etc.

2. **Endpoint category indicators** are typically broader and more comprehensive compared with midpoint indicators, as they represent the final environmental impacts that are relevant to human health, ecosystems, and resources. They are derived from the midpoint indicators through a characterization process. Examples of endpoint categories are Carcinogenicity, Biodiversity Loss, Fossil Fuel Depletion, etc.

Classification: it consists in the allocation of LCI results (input and output flows) to the selected impact categories.

Characterization: it consists in the calculation of category indicator results (characterization). This is achieved using impact factors; for each impact category, inputs and outputs are multiplied by their respective impact factors to transform them into the equivalent unit of environmental impact, resulting in the corresponding indicator.

E.g., if we choose the category of climate change, which affects ecosystem health, greenhouse gas emissions will be used as an indicator, and the IPCC (Intergovernmental Panel on Climate Change) method will be employed for calculation. The life cycle inventory will provide various inputs (e.g., carbon dioxide fixation) and outputs (emissions of methane or other greenhouse gases). Each input and output will be multiplied by its impact factor (e.g., methane has a global warming potential of 28 kg CO2 eq. per kg), resulting in a unique indicator in kg of CO2-eq.

6.1.1.4 Interpretation

This section is cross-cutting across all others and has an iterative nature. Here, any issues or conflicts between the LCA results, LCIA, and the defined objectives and scope are identified. Different parts of the study must be consistent and contribute to achieving the defined objectives. The LCA shall be critically analyzed to ensure completeness, conducting sensitivity analyses and consistency checks if necessary. Finally, conclusions shall be drawn in this section, identifying limitations and making recommendations based on the results.

6.1.2 Limitations of the LCA

While Life Cycle Assessment (LCA) is widely regarded for its robustness, it does have limitations.

In spite of the requirements outlined by the International Organization for Standardization (ISO) in protocol ISO 14040:2006, practitioners are afforded flexibility to tailor methodological choices that align with the objectives of their study. While this can be advantageous in customizing LCAs for a given purpose, it can lead to inconsistencies and having a confounding effect, with different LCAs of the same product yielding seemingly disparate results (Miller, 2022).

This criticism is summarized on four major issues:

1. The boundaries of analysis (Miller, 2022)

2. The quality of analysis may be limited by the uncertainty and quality of the data used (Anwar, 2020; Miller, 2022)

3. Appropriately capturing the environmental impacts (Anwar, 2020; Miller, 2022)

4. In some cases, the results cannot be applied to local conditions (Anwar, 2020)

6.2 Environmental impact of pig production

6.2.1 Greenhouse gases (GHGs) emissions

GHGs emissions from slurry and manure are generated thorough the pig's life cycle, from growing until manure disposal: livestock feeding, manure generated in livestock housing and on open yard areas, manure storage, manure treatment, and field-applied manure. Emissions range from carbon dioxide, methane, N-related emissions (ammonia, nitrate, and nitrous oxide), P-related emissions and K-related emissions.

The Biochemical Methane Potential (BMP) during storage is rather low because of the presence of high concentrations of ammonia, which have inhibitory effects on methanogenic bacteria (Meegoda et al., 2018). However, during land-spreading, ammonia degradation to nitrogen gas can produce nitrous oxide (EPA, 2007). In addition, the organic matter (OM) present in the slurry can be further degraded mainly into carbon dioxide, which in this case is thought to have a neutral Global Warming Potential (GWP) since it comes from biological sources that captured carbon in a short period of time.

However, in the case of an anaerobic digestor (AD), the BMP can be higher if the process is carefully optimized. The resulting biogas can be captured and converted into electricity, which reduce the Global Warming Potential (GWP).

6.2.2 Nitrogen emissions

Nitrogen in pig slurry is present in the form of urea, which is a by-product of protein breakdown. Urea can be degraded into ammonia-ammonium following two pathways: urease-catalysed hydrolysis and un-catalyzed degradation in aqueous solution (Sigurdarson et al., 2018):

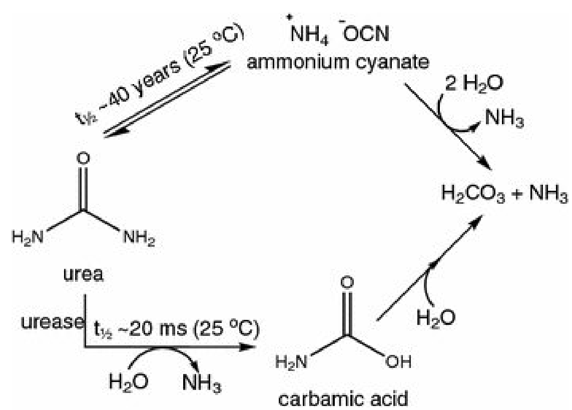

Figure 6.2: Urea degradation pathways, from Sigurdarson et al., 2018.

Sigurdarson et al., 2018 stated that urea hydrolysis of urea in animal manure slurry is complete within approximately 20 hours after the urine and feces are mixed. Therefore, in practice all the urea has been completely turned into ammonia by the time the slurry is applied in the field.

Ammonia is a highly volatile compound that can be uptaken to the atmosphere or absorbed by plants. When it is volatilised, it can react with compounds present in the atmosphere and afterwards be deposited in the environment, specially close to the source of emission. Once is deposited, ammonia can be converted into nitrite and nitrate through nitrification and denitrification or absorbed by plants and microorganisms (Sigurdarson et al., 2018).

Nitrification is a two-step microbial process by which reduced nitrogen compounds (primarily NH_3) are sequentially oxidized to nitrite NO_2^- and NO_3^- by autotrophic nitrifying bacteria (AWWA & EPA, 2002). Eq. 6.1 and Eq. 6.2 from AWWA and EPA, 2002 summarize the 2-step nitrification process:

$$NH_3 + O_2 \Rightarrow NO_2^- + 3H^+ + 2e^- \tag{6.1}$$

$$NO_2^- + H_2O \Rightarrow NO_3^- + 2H^+ + 2e^- \tag{6.2}$$

Denitrification is the second part of the transformation of ammonia into nitrogen gas. It is a process by which nitrates are reduced to gaseous nitrogen (N_2) by facultative anaerobic heterotrophic bacteria (EPA, 2007), and generally. It generally proceeds through some combination of the following half reactions (**hola**):

$$NO_3^- + 2H^+ + 2e^- \Rightarrow NO_2^- + H_2O \quad \text{(Nitrate reductase)} \tag{6.3}$$

$$NO_2^- + 2H^+ + e^- \Rightarrow NO + H_2O \quad \text{(Nitrite reductase)} \tag{6.4}$$

$$2NO + 2H^+ + 2e^- \Rightarrow N_2O + H_2O \quad \text{(Nitric-oxide reductase)} \tag{6.5}$$

$$N_2O + 2H^+ + 2e^- \Rightarrow N_2 + H_2O \quad \text{(Nitrous-oxide reductase)} \tag{6.6}$$

One of the products of denitrification is nitrous oxide (N_2O), a potent GHGs with a emission factor of 273 kg CO_2-eq (Arias et al., 2023).

Another pathway by which ammonia can be converted into nitrogen gas is the Anaerobic Ammonium Oxidation, namely Anammox, which is an anaerobic ammonium oxidation process carried out by autotrophic bacteria in which NH_3 and NO_2^- simultaneously converted into N_2 (Rangaswamy et al., 2022). Eq. 6.7 from Reimann et al., 2015 summarizes the process (Sigurdarson et al., 2018):

$$NH_4^+ + NO_2^- \Rightarrow N_2 + 2H_2O \tag{6.7}$$

In the atmosphere, ammonia can react with sulphuric acid (H_2SO_4) or nitric acid (HNO_3) shown in Eq. 6.8 and 6.9 from Sigurdarson et al., 2018, which make up a large part of fine particles in the air known as $PM_{2.5}$:

$$2NH_3 + H_2SO_4 \Rightarrow (NH_4)_2SO_4 \tag{6.8}$$

$$NH_3 + HNO_3 \rightarrow NH_4NO_3 \tag{6.9}$$

It is important to note that anaerobic digestion does not remove nitrogen effectively and therefore, N emissions cannot be avoided. In fact, the resulting digestate exhibits higher ammonia content when compared to its raw counterpart, and can be used as an organic fertiliser.

6.3 Treatment Technologies

6.3.1 Anaerobic digestion

6.3.1.1 Introduction

Anaerobic digestion can be defined as a biological process in which organic matter, in the absence of oxygen, is converted into other products (Toerien & W., 1969). This biological process is ruled by microbial consortia that breaks down slurry through hydrolysis and fermentation, resulting in the production of biogas and digestate among other products.

The resulting biogas is predominantly composed of methane (CH_4) that, in optimal operational conditions, constitutes up to 70% of its relative mass, with the remaining portion comprised of carbon dioxide (CO_2) (Prenafeta & Parera, 2020) and other residual products like ammonia (NH_3) and hydrogen sulfide (H_2S). This biogas can then be further combusted the supply of heating and electricity or upgraded for high value applications e.g., as vehicle fuel (Fuqing et al., 2018) or upgraded for methane production in a process known as methane up-grading (Ardolino et al., 2021). The digestate can be agronomically valorized as an organic fertiliser (Li et al., 2018; Prenafeta & Parera, 2020), either directly or through its transformation into other products via other processes like composting or solar drying (Prenafeta & Parera, 2020). Though a natural process, it can indeed be recreated artificially, namely anaerobic digesters. These showcase a wide range of design approaches based on how they operate, shown in 6.1:

Passive digesters	Open lagoon	
	Covered Lagoon	
With mass retention	Continuous reactors	CSTR
		Piston flux
	Batch reactors	SBR
Without mass retention	High performance reactors	

Table 6.1: Types of anaerobic digesters currently in use in Catalonia (Prenafeta & Parera, 2020).

In passive systems, there is no active intervention or control over operational parameters like mixing or temperature control. Mass retention can take place or not depending on the needs. Reactors that lack biomass retention fall into distinct categories, including continuous reactors such as Continuous Stirred Tank Reactors (CSTR) or piston flow reactors (PFR), as well as batch systems like the Sequential Batch Reactor (SBR) (Prenafeta & Parera, 2020).

In Catalonia, the most widespread anaerobic reactor configuration for swine slurry treatment is usually based on a CSTR configuration. CSTR are fed constantly, resulting in a steady-state and a constant gas production rate. Since they are limited to substrates that can be pumped for continuous feeding (Saikat et al., 2022), it is ideal for pig slurry. Generally, reactors operate at a total solids (TS) maximum of 12% (m/m) and at a relatively constant temperature of 30-45ºC in mesophylic conditions, or at a temperature of 52-55ºC with a maximum variation of 0,5ºC in thermophylic conditions. The hydraulic residence time (HRT) usually ranges from 20 to 70 days depending if slowly biodegradable substrates, like crop residues, are added or, if adaptation to inhibitors like high ammonia or VFAs content, is needed. Additionally, to provide adequate time for the bacterial consortia to sustain themselves. Since in CSTR bacterial consortia is being expelled along with the digestate, cellular retention time (CRT) and (HRT) have the same value. Following the process, it is necessary to maintain the digestate in a covered deposit to recover the residual biogas released during this phase, which can represent between 10% and 15% of the total production (Prenafeta & Parera, 2020).

6.3.1.2 Phases of anaerobic digestion

Anaerobic digestion takes place through four phases that take place at the same time: hydrolysis, acidogenesis, acetogenesis, and methanogenesis. Fig. 6.3 shows a simplified scheme of the pathways involved in anaerobic digestion, which will be further discussed in the following sections:

Hydrolysis Hydrolysis is the first stage of the AD, where complex polymers are broken down by hydrolytic bacteria from carbohydrates, lipids, and proteins into sugars, long chain fatty acids (LCFAs), and amino acids, respectively (Li et al., 2011; Meegoda et al., 2018; Richard et al., 2019). Hydrolytic enzymes generally include amylase, cellulase, lipase, protease, and pectinase (Uddin & Wright, 2023). The resulting components can be further utilised by acidogenic bacteria (Meegoda et al., 2018). For lignin-rich substrates, the breakdown of polymers turns into the rate-limiting stage (Uddin & Wright, 2023).

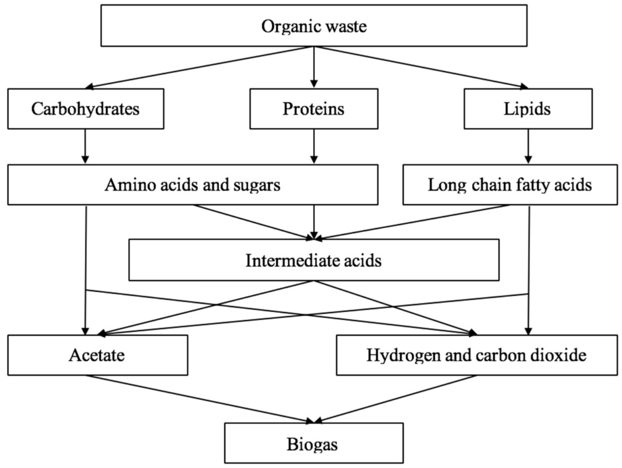

Figure 6.3: Simplified diagram of the chemical transformations taking place in an anaerobic reactor, from Gujer and Zehnder, 1983.

This is why, along with other reasons further explained, slurry from swines fed with high-roughage diets yield less methane compared with swines fed with high-grain diets (Y. Chen, 1983).

Eq. 6.10, from (Richard et al., 2019), represents the overall reaction of this stage:

$$(C_6H_{12}O_6 + H_2O)_x \Rightarrow x(C_6H_{12}O_6) \qquad (6.10)$$

Acidogenesis Acidogenesis is the fermentation stage where the products of hydrolysis (soluble organic monomers of sugars and amino acids) are degraded by acidogenic bacteria to produce alcohols, aldehydes, Volatile Fatty Acids (VFAs) and acetate together with hydrogen gas (H_2), carbon dioxide (CO_2) and ammonia (NH_3) (Richard et al., 2019). This step is generally believed to proceed at a faster rate than all other phases of anaerobic digestion (Meegoda et al., 2018).

VFAs are linear short-chain aliphatic mono-carboxylate compounds, such as acetic acid, propionic acid, and butyric acid, which are the building blocks of different organic compounds (Merrylin et al., 2020). They generally come from amino acid degradation via the Stickland reaction (Eq. 6.11) and via single amino acid degradation by hydrogenotrophic bacteria when present:

$$\text{AA1} + \text{AA2} \rightarrow \text{Fatty Acid} + \text{Ammonia} + \text{Organic Acid} \tag{6.11}$$

VFAs are an essencial compound in anaerobic digestion since their production creates direct precursors for the final stage of methanogenesis. Bacteria species found in anaerobic reactors which participate in acidogenesis via amino acid degradation are *Lactobacillus*, *Escherichia*, *Staphylococcus*, *Pseudomonas*, among others. Other species, such as *Clostridium*, *Eubacterium limosum*, and *Streptococcus*, transform sugars into intermediary fermentation products (Kamusoko et al., 2022) that can be further used as a substrate for other microorganisms.

Eq. 6.12, 6.13 and 6.14 from Richard et al., 2019 resume the stage:

$$C_6H_{12}O_6 \Leftrightarrow 2CH_3CH_2OH + 2CO_2 \tag{6.12}$$

$$C_6H_{12}O_6 + H_2O + 2H_2 \Leftrightarrow 2CH_3CH_2COOH + 2H_2O \tag{6.13}$$

$$C_6H_{12}O_6 \Rightarrow 3CH_3COOH \tag{6.14}$$

Acetogenesis A portion of the acidogenesis substrate, namely amino acids, is already been rendered into acetate, a substrate suitable for methanogens. However, other products of acidogenesis like VFAs must yet need to undergo some chemical changes to be made accessible to methanogenic bacteria. These FVAs and other intermediates are converted into acetate, with H_2 and CO_2 also being produced (Meegoda et al., 2018). 70% of the organic material passes through this pathway in the eventual formation of methane (Hill, 1982; Sravan et al., 2021).

Acetogenesis is carried out by two bacteria groups: acetogens, which form acetate, propionate and butyrate via the Wood–Ljungdahl pathway (Sravan et al., 2021) shown in Eq. and hydrogenogens, which use the latter propionate and butyrate to form acetate, formate, H_2 and CO_2.

The overall reactions of this stage are presented; Eq. 6.15, 6.16 and 6.17 (Richard et al., 2019):

$$CH_3CH_2COOH + 2H_2O \Rightarrow CH_3COOH + CO_2 + 3H_2 \tag{6.15}$$

$$C_6H_{12}O_6 + H_2O + 2H_2 \rightleftharpoons 2CH_3CH_2COOH + 2H_2O \tag{6.16}$$

$$CH_3COO^- + 4H_2O \Rightarrow 2HCO_3^- + H^+ + 4H_2 \tag{6.17}$$

Methanogenesis Methanogenesis marks the final stage of anaerobic digestion (Richard et al., 2019), where acetotrophic methanogens and hydrogenotrophic methanogens synthesise methane with the intermediate products generated during acetogenesis (Meegoda et al., 2018).

Acetotrophic methanogens degrade acetate to methane and carbon dioxide, as shown in Eq. 6.18. This pathway accounts for 70% of the total methane production in AD systems.

$$CH_3COOH \Rightarrow CH_4 + CO_2 \tag{6.18}$$

Hydrogenotrophic methanogens use carbon dioxide and hydrogen to produce methane, as Eq. 6.19 shows. This pathway accounts for 30% of the total methane production in AD systems, since hydrogen is in limited supply:

$$CO_2 + 4H_2 \rightleftharpoons CH_4 + 2H_2O \tag{6.19}$$

Methane can also be formed from ethanol by substrate oxidation, as Eq. 6.20 shows:

$$2CH_3CH_2OH + CO_2 \Rightarrow CH_4 + 2CH_3COOH \tag{6.20}$$

The resulting biogas is constituted mainly by methane with a richness oscillating between 60% and 70%, and the rest accounts for carbon dioxide and other residual gases (Prenafeta & Parera, 2020).

6.3.1.3 Nitrogen recovery

Biological activity leads to a change in some components of the slurry's composition. Tab. 6.2 shows this alterations for a CSTR reactor with a 20-day HRT of 20 days at 35ºC (Prenafeta & Parera, 2020):

Parameter	Recovery in the digestate
Flux, Q_v	95-98
Total Solids, TS (dry matter)	20-45
Volatile Solids, VS	40-60
Total Nitrogen, TN	100
Organic nitrogen, ON	60-40
Total Ammoniacal nitrogen, TAN	140-160
Total Phosphorous, TP	100
COD	40-60

Table 6.2: Nutrient content recovered in the digestate, from Prenafeta and Parera, 2020.

Total Ammoniacal Nitrogen (TAN) is the ammoniacal fraction of Nitrogen in slurry, which large amounts are excreted by livestock in urea and faeces, namely slurry (Serra-Toro et al., 2022). Contrary to volatile solids (VS), TAN increases after treatment due to protein degradation to amino acids and further transformation into NH_3 (Deng et al., 2023; Prenafeta & Parera, 2020).

Due to a lack of oxygen, the excess of ammonia cannot be nitrified into nitrate and, subsequently, into nitrogen gas (N_2) by denitrification. Since there is no NO_3^- production, denitrification cannot take place and hence, nitrogen is not reduced. Therefore, NH_3 reduction to N_2 cannot take place. Also, there is no N_2O production and hence, emission of GHG from nitrogen compounds is negligible. In addition, the annamox pathway cannot take place since it needs NO_2^- to go through. The result is an increase in NH_3 content from protein degradation added up to the initial concentration derived from urea. A fraction of this ammonia will be absorbed by plants, and another will be volatilised to the atmosphere after land spreading (Prenafeta & Parera, 2020).

6.3.1.4 Inhibitory parameters during anaerobic digestion

Overloading of organics can cause upsets in that slurry is quickly degraded resulting in an over-accumulation of VFAs and NH_3, which inhibits methanogenesis and hence, disrupts the anaerobic digestion (Meegoda et al., 2018). Y. Chen, 1983 established an OLR limit of 55 $\mathrm{kg \cdot m^{-3} \cdot d^{-1}}$ in the influent.

VFAs overload takes place when VFAs degradation is impaired, which often occurs when degradation kinetics are higher during the first steps of AD compared to methanogenesis (Maurus et al., 2023). This leads to the accumulation of VFAs and afterwards, to a decrease in pH, which in turn again negatively affects the performance of AD. This is because bacterial strains in AD only perform effectively under a determined pH range that depends on the species.

Ammonia, which is generated as a by-product of acidogenesis via amino acid breakdown, is widely reported to entail a cause for digester failure (Meegoda et al., 2018). TAN concentrations higher than 1,4 g $\mathrm{NH_3\text{-}N \cdot L^{-1}}$ can potentially halve methane production (Rivera et al., 2022). Overloading may reduce the process efficiency due to: occurrence of foaming, incomplete degradation of organic nitrogenous compounds, inhibition of methanogenesis, and deterioration of the morphological sludge properties (Deng et al., 2023). Also, the resulting digestate has high ammonia concentrations, which can pose an impact on water bodies (eutrophication and nitrate pollution) and air quality (ammonia volatilization and precursor of dinitrogen monoxide) (Rivera et al., 2022).

6.3.2 Composting

6.3.2.1 Introduction

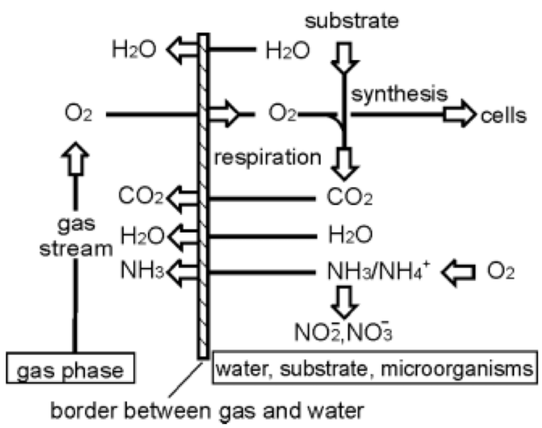

Figure 6.4: Metabolism of aerobic microorganisms at the gas/water interface, from Jördening and Winter, 2005.

Composting is a long-used technology that can be defined as the biological process where organic matter, in the form of polymeric organic compounds, is degraded by enzymatic breakdown into soluble organic carbon and sugars suitable for utilization by microorganisms (Manyapu et al., 2022). This decomposition is achieved by using carbon and nitrogen as energy sources and metabolic blockchains along with oxygen and water, producing water, carbon dioxide, and compost as byproducts (Rastogi et al., 2020).

The compost is a product free of odours (Prenafeta & Parera, 2020) that is suitable to be used as a soil-amendment (Ayilara et al., 2020) due to its nutrient richness and hygiene safeness (Papale et al., 2021). The production and use of compost constitutes an alternative to chemical fertilisers and hence, contributes to reduce the environmental impacts arising from their production, use and co-produced waste disposal.

On a practical level, different operational approaches are available. Fig. 6.3 showcases the ones currently used in Catalonia:

Open/covered systems	Turning piles	
	Static piles	Air suction
		Air propulsion
		Alternating ventilation
Closed systems		

Table 6.3: Overview of the current composting technologies currently in use in Catalonia (Prenafeta & Parera, 2020).

These systems are classified depending on their degree of protection from weather conditions – open/covered systems – and based on how passive is the air supply – turning/static piles. Turning piles are considered to be active systems since there is an ongoing turnover of the substrate to fuel aeration, while static piles are considered passive systems where air is administered through mechanic ventilation (Prenafeta & Parera, 2020). Open or covered systems, on the other hand, are more or less suitable depending on the location of the composting plant since there can be odour emissions; open or covered systems are usually located far from urban settlements, while closed systems can be placed nearby (Prenafeta & Parera, 2020).

Due to their simplicity and distance from inhabited areas, the most widespread systems in Catalonia are open, outdoors and whether with turned or static piles Prenafeta and Parera, 2020. For an optimal compost production, the initial substrate must possess a moisture content within the range of 40% to 56%, a C/N between 25 and 35, and enough porosity to promote optimal air circulation within the stacked material (Prenafeta & Parera, 2020). Because the substrate can be dense and have a high moisture content, bulking agents e.g., ligno-cellulosic material can be added to increase porosity and therefore, air circulation within the pile (Dentel & Qi, 2014; Prenafeta & Parera, 2020).

Regarding livestock manure, the composting process can be applied to all sorts of faeces, poultry litter, the digestate from the anaerobic digestion of manure and the solid fraction of slurry obtained through the solid-liquid separation process (Prenafeta & Parera, 2020). In the context of manure treatment, solid-liquid separation is a key technology that is based on physicochemical processes that allow removing part of the slurry's solids (R. Dong et al., 2022) obtaining a fraction with low solid content and another rich in solid content, namely liquid and solid fraction respectively (Prenafeta & Parera, 2020). This physicochemical processes used in the process of separation both fractions are based on decantation and mechanical processes.

6.3.2.2 Phases of composting

Composting is doubtless a self-heating process where temperatures tend to increase, to gradually decline to values comparable to environmental conditions as the substrate matures (Policastro & Cesaro, 2023). Therefore, temperature is a significant factor in determining the relative advantage of some microbial population over another (Zhang et al., 2022). In this sense, the process is usually divided into three phases (Policastro & Cesaro, 2023), even though some authors divide the process in four phases (M. Chen et al., 2011; Papale et al., 2021; Zhang et al., 2022).

The initial mesophylic phase is characterised by a rapidly increase in temperature up to 40 - 42ºC (M. Chen et al., 2011; Papale et al., 2021) due to the start of the biological decomposition of the substrate material (Papale et al., 2021). During this phase, organic polymers degrade into simpler compounds:

Eq. 6.21 and Eq. 6.22 from Jakobsen, 1992 showcases the aerobic decomposition of carbohydrates:

$$C_6H_{12}O_6 + 6O_2 \rightarrow 6CO_2 + 6H_2O \tag{6.21}$$

$$CH_3COOH \rightarrow CH_4 + CO_2 \tag{6.22}$$

Eq. 6.22 from Jakobsen, 1992 showcase the degradation of amino acids during aerobic decomposition:

$$\text{amino acid} + 3O_2 \rightarrow 2CO_2 + HCO_3^- + NH_3 + H_2O \tag{6.23}$$

The following phase, namely thermophylic phase, is distinguished by higher temperatures that can reach between 45 - 70ºC (Papale et al., 2021), and it can last days or even weeks depending on feedstock properties, pile size and environmental conditions (M. Chen et al., 2011). During this phase, decomposition of organic matter is the most rapid (Papale et al., 2021), shorting the maturity period of the substrate and the high temperatures contribute to its hygienisation (Policastro & Cesaro, 2023).

The last phase is the finishing phase, in which organic matter and biological heat production stabilizes (Papale et al., 2021).

6.3.2.3 Critical factors

Carbon and nitrogen are one of the most important factors to take into account in composting, since carbon serves both as a source of energy and elemental component for microorganisms while nitrogen is essential for the synthesis of amino-acids, proteins and nucleic acids (Azim et al., 2018). The optimal proportion between these two elements is showcased with the C/N ratio. Just like the temperature, the C/N ratio serves as an indicator of both the efficacy of the composting process and the quality of the resultant products.

The ideal C/N ratio ranges between 25-35 (Azim et al., 2018; Ho et al., 2022; Prenafeta & Parera, 2020). Below this threshold, the growth of microorganisms can be stunted and in consequence, leads to slow rates of decomposition and therefore, can lead to prolonged composting duration (Azim et al., 2018). Higher C/N ratios, on the other hand, enhance nitrogen loss ({**Azim2018**}) by releasing ammonia or nitrogen gas and nitrous oxide through nitrification and denitrification. Therefore, maintaining aerobic conditions is imperative for controlling NH3 emissions, as there is not consistent evidence to suggest whether treating waste with AD prior to composting increases or decreases NH3 emissions (Nordahl et al., 2023). pH lower than 6 ensure that ammonia stays in soluble form (Jamaludin et al., 2018).

Moisture content plays a key role in the efficiency of the composting process, as well as the quality of the resulting product. Since it affects microbial activity – water availability limits microbial activity –, as well as the physical structure of the substrate – it affects particle aggregation, matrix porosity and gas permeability thus limiting oxygen transport thorough the composting pile –, it has a central influence on organic material biodegradation (Makan et al., 2013). The ideal proportion of moisture content ranges from 40-70% (Kim et al., 2016; Makan et al., 2013); below this threshold, microorganism growth and survival is impaired, thus giving physically stable but biologically unstable composts (Makan et al., 2013), and above it can trigger anaerobic conditions as pore spaces of the solid matrix are filled with water rather than air (Kim et al., 2016). This can prevent and halt the ongoing composting activities, leading to emissions of methane and nitrous oxide by anaerobic microorganisms.

Methodology

7.1 Modelling of the pig slurry management life cycle stages

7.1.1 Storage (ST)

For ST, CH_4 emissions were estimated in line with H. Dong et al., 2006; Mller et al., 2004; Mangino et al., 2001; Zeeman, 1994 and it was based on pits below animal confinements. In addition, NH_3 emissions were also estimated in line with Griffing et al., 2007. In contrast, N_2O emissions were not considered due to the low-oxygen availability within this type of storage (H. Dong et al., 2006), and CO_2 emissions were excluded because they are considered to be biogenic and therefore, neutral in terms of GWP.

7.1.2 Anaerobic digestor (AD)

For AD, process emissions were estimated in line with Achinas and Euverink, 2016; Prenafeta and Parera, 2020. The Boyle's Busswell modified equation (see Eq. 7.1 below) was chosen as a basis for the modelling of biogas due to the requirement of less input parameters compared with other modeling approaches (such as empirical models (Y. Chen, 1983) or kinetic dependencies, like the Monod and Conto kinetic models), and its specifically designed for a Continuous Stirred-Tank Reactor (CSTR), which is the most used type of anaerobic digestor configuration in Catalonia (Prenafeta & Parera, 2020)). Methane was further converted into electricity considering a conversion factor of 52,5 $MJ \cdot kg^{-1}$ slurry (WNA, 2023), and digestate production was estimated separately using the average nutrient recovery for digestate shown in Fig. 6.2 in line with Prenafeta and Parera, 2020. In addition, electric consumption was considered, as CSTR require energy for mixing and temperature maintenance. For a more in-depth explanation of the modelling of this treatment, see Section 2.

$$C_aH_bO_cN_dS_e + C_1 \cdot H_2O \rightarrow C_2 \cdot CH_4 + C_3 \cdot CO_2 + C_4 \cdot NH_3 + C_5 \cdot H_2S \qquad (7.1)$$

7.1.3 Solid-liquid separation (SLS), composting (CP) and transport (TR)

For the SLS, pressing screw was chosen as a proxy due to its widespread use in Catalonia (Prenafeta & Parera, 2020). Nutrient balance between the solid-phase and the liquid-phase was calculated in line with Mller et al., 2004, and energy requirements for the process were derived from ROU, 2006.

For CP, active windrow was used as the basis for the model, since it is the most widespread in Catalonia (Prenafeta & Parera, 2020). CH_4 emissions were estimated in line with H. Dong et al., 2006, and are temperature-dependent. N_2O and NH_3 were considered and estimated in accordance with Klein et al., 2006, and Fukumoto et al., 2003, respectively. Energy consumption from turning up de composting pile was estimated in line with ROU, 2006. Compost production was estimated subtracting CH_4 and NH_3 emissions from the initial post-SLS solid-phase composition, and further converted into units of kg

C and kg N, respectively.

For TR, emissions were estimated through the use of EURO6-compliant vehicles, and emissions were gauged depending on the distance and weight to carry, in tkm.

7.1.4 Land-spreading (LP)

Emissions were modeled in line with Corbala-Robles et al., 2018. Mineral fertiliser that substitutes organic fertiliser and the avoided emissions from its application was calculated in line with Tonini et al., 2020. Substitution was then hulled taking into account the aggregated sum of the most used fertilisers in Spain (IFASTAT, 2021). For further explanation of the process followed, see Section 2.

1. **Emissions from land-spreading of the digestate, compost, and post-storage raw slurry**: these include both direct and indirect N_2O emissions (Clercq et al., 2015; Prenafeta & Parera, 2020; Roe & et al., 2019), NO_x emissions (Nemecek & Kägi, 2007), NH_3 emissions (EMEP/EEA, 2023; Prenafeta & Parera, 2020) and NO_3^- emissions (Roy et al., 2003; Smaling, 1993).

2. **Avoided emissions from using mineral fertiliser**. Mineral substitution was carried out in line with Corbala-Robles et al., 2018 using as a basis the most common fertilisers used in Spain to the latest available date (IFASTAT, 2021).

7.1.5 Impact methodology

The methodology chosen to conduct the life cycle impact assessment was the Environmental Footprint (EF) 3.1 methodology, from which the following impact categories were chosen (EC, 2022):

1. **Climate Change (CC)**: The results are expressed in terms of global warming potential (GWP 100) and reported in kilograms of CO_2-equivalent (kg CO_2-eq). This provides a measure of the contribution of the product or process to global warming.

2. **Marine Eutrophication (MEU)**: The results are expressed in kilograms of nitrogen-equivalent (kg N eq). This indicates the potential for nutrient enrichment in marine environments, which can lead to adverse ecological effects such as algal blooms and hypoxia.

The remaining impact categories from EF 3.1 were included in Reducepi but not selected for our assessment because the primary focus of the analysis was to address the impacts associated with the emission of GHGs and nitrogen compounds, since they are critical indicators of environmental performance. On one hand, CC analyses wide-ranging effects related to emission of GHGs, and MEU is also of significant concern, as it measures the effects of nitrogen in aquatic ecosystems'. Other impact categories, while important, were deemed less critical for the specific scope and objectives of this study. Additionally, limiting the number of impact categories simplified the analysis and allowed for a more in-depth examination of the chosen impacts.

7.2 Scenarios

The three treatment lines were considered in our assessment and were considered in this assessment:

1. Scenario AD (anaerobic digestion)

2. Option SLS+CP (solid liquid separation + composting)

3. Option NT (no treatment), "control group"

Fig. 7.1 showcases the system boundaries considered in this assessment:

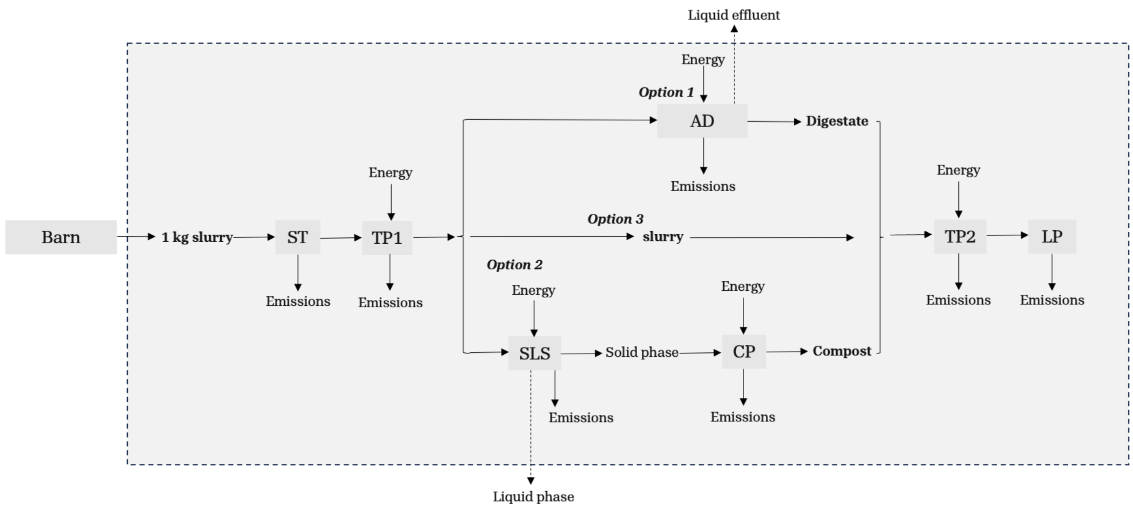

Figure 7.1: System boundaries considered.

1. **Manure Storage (ST)**: containment of waste prior to subsequent phases.

2. **Transport (TR1)**: Potential transportation to treatment facilities.

3. **Treatment**: treatment used.

4. **Transport (ST2)**: Post-treatment, the processed waste potentially transported to site to further be land-spreaded.

5. **Land-spreading (LP)**: The final stage involved the application of treated waste onto land surfaces, serving purposes such as soil enrichment or agricultural fertilization.

7.2.1 Scenario AD

Scenario AD involved the impact assessment of treating pig slurry with anaerobic digestion. The storage residence time was considered to be more than 1 month by default, and an annual average temperature of 12ºC was considered for this step. For the anaerobic digestor, the residence time was assumed as 25 days, which is within the range stated in Prenafeta and Parera, 2020, and process temperature was assumed to be in the range of mesophylic conditions (35ºC). An aggregated temperature variation of 1ºC/day was considered when calculating the electric consumption from temperature maintenance, and 2 mixing times of 1 hour with a Rushton impeller at 2,50 rps were considered for the estimation of the the energy required for mixing within the reactor. Both TR1 (transport from farm to treatment facilities) and TR2 (from the treatment facilities to the field) was considered, and a default distance of 30 km was assumed for both TR1 and TR2. Finally, digestate was land-spreaded (LP).

7.2.2 Scenario SLS+CP

Scenario SLS+CP involved the impact assessment of treating pig slurry with composting. The storage residence time was considered to be more than 1 month by default, and an annual average temperature of 12ºC was considered for this step. Slurry underwent SLS, and the solid fraction was used for CP. TR1 was considered null, since treatment was assumed to take place within the farm. The process temperature of the composting process was assumed to be 25ºC. TR2 was considered in this scenario, and a default distance of 30 km from the farm to the field was considered. Finally, digestate was land-spreaded (LP).

7.2.3 Scenario NT

Scenario NT involved the impact assessment of not treating the pig slurry and, therefore, acted as a control group. The storage residence time was considered to be more than 1 month by default, and an annual average temperature of 12ºC was considered for this step. Afterwards, slurry was transported (TR2) to the field, and a default distance of 30 km from the farm to the field was considered.

7.3 Slurry composition

Slurry composition was provided by the Reducepi project and involved a modelling approach based on the initial feed makeup including factors like the animals' protein retention capacity, food digestibility, and feed conversion ratio, alongside other variables. Reducepi project provided us with four composition cases which were classified in two groups:

1. **SC-A(d)**: slurry composition from pigs weighing 54kg fed with 30kg of diet-based feed (control group).

2. **SC-A(s)**: slurry composition from pigs weighing 54kg fed with 30kg of sorghum-based feed.

3. **SC-B(d)**: slurry composition from pigs weighing 70kg fed with 97kg of diet-based feed (control group).

4. **SC-B(s)**: slurry composition from pigs weighing 70kg fed with 97kg of sorghum-based feed.

Slurry composition, kg	SC-A(d)	SC-A(s)	SC-B(d)	SC-B(s)
C	1,136	1,600	3,334	3,455
H	0,206	0,267	0,739	0,771
O	1,161	1,636	3,408	3,532
TN	0,226	0,245	1,263	1,417
S	0,052	0,049	0,189	0,189
K	0,051	0,032	0,158	0,158
P	0,033	0,052	0,196	0,196
Total	2,813	3,880	9,287	9,717

Slurry composition, %	SC-A(d)	SC-A(s)	SC-B(d)	SC-B(s)
C	40,38%	41,24%	35,90%	35,56%
H	7,33%	6,87%	7,96%	7,93%
O	41,28%	42,16%	36,70%	36,35%
TN	8,03%	6,31%	13,60%	14,58%
S	1,85%	1,27%	2,03%	1,94%
K	1,16%	1,33%	2,11%	2,02%
P	1,82%	0,82%	1,70%	1,62%
DM	10,00%	10,00%	10,00%	10,00%
OM	9,12%	8,76%	9,20%	9,17%

Table 7.1: Absolute and relative composition of pig slurry used to test Reducepi.

This approach facilitated a clear understanding of how diet impacts both nutrient utilization and slurry composition as pigs develop in size, maturity, and biological complexity. Tab. 7.1 presents the results of the slurry composition modelling.

Results and discussion

8.1 Reducepi overview

8.1.1 Introduction

As outlined in Chapter 5, the primary objective of this master thesis is to identify environmental impact hot-spots during the slurry life cycle and examine how different diets influence these impacts. In order to achieve this, a flexible, modifiable and comprehensive Excel-based calculator has been developed following the ISO 14040:2006 standard for life cycle assessment (LCA). This calculator enables stakeholders to model the slurry management supply chain from Storage of the slurry to transport, land-spreading and in addition, offers different treatment options. For the sake of simplicity, the tool will be named from now on "Reducepi".

8.1.2 Structure of the tool

Fig. 8.1 shows the structural data flow of Reducepi:

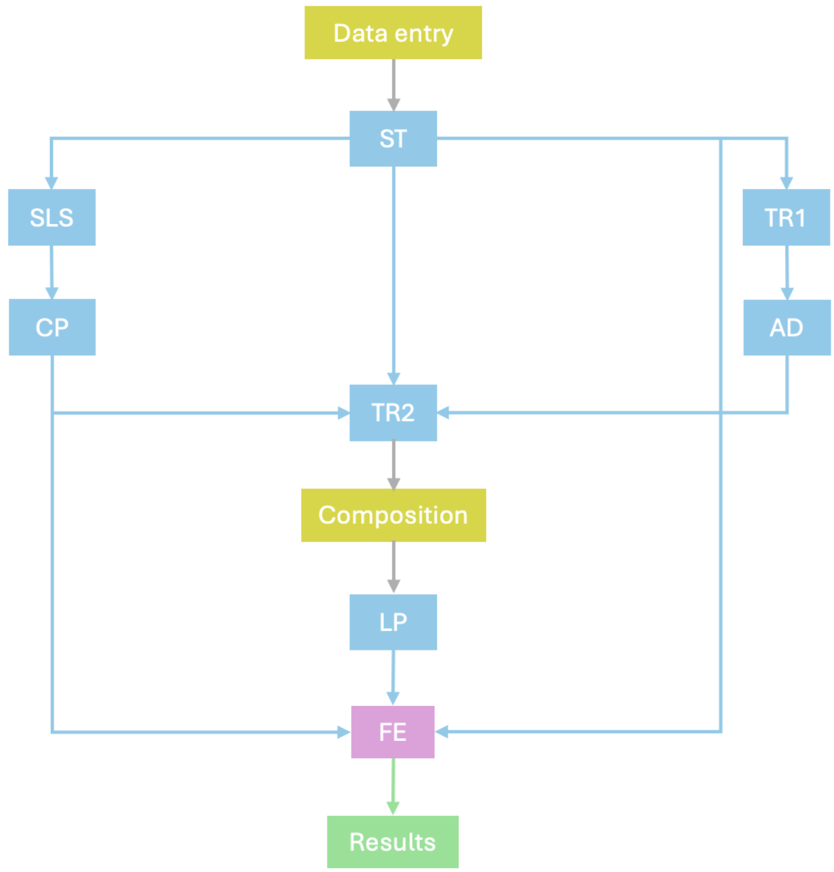

Figure 8.1: Overview of the structural data flow of Reducepi.

Sheets coloured with yellow are the **Data modifiable sheets**, where data is introduced and can be changed. Sheets coloured in blue correspond to the **Modelling sheets**, where both calculation of operation emissions are calculated. Electric consumption for every treatment is also included. Sheets coloured in purple correspond to "emission factor sheets", where the LCI (FE) and LCIA (FE2) are developed from results of "modelling sheets". Finally, sheets coloured in green present the results of "emission factor sheets".

8.1.2.1 Data modifiable sheets

Data modifiable sheets consists on two sheets: Data Entry sheet and Composition Sheet.

Fig. 8.2 presents the Data Entry sheet:

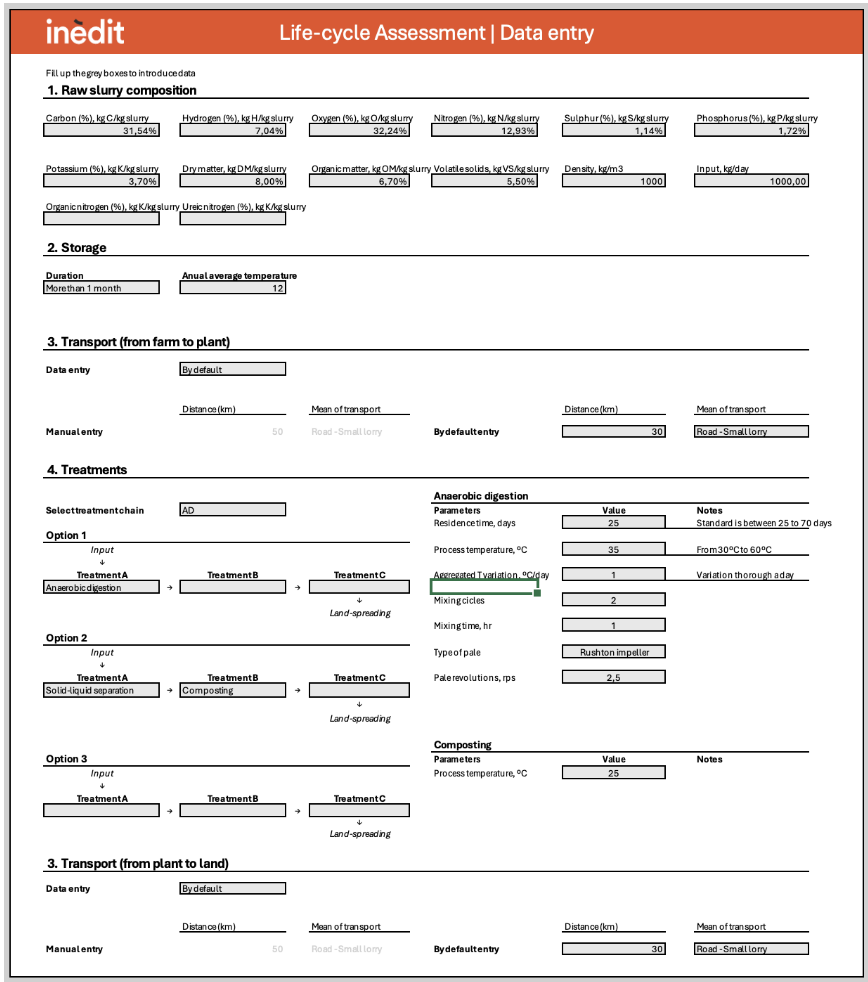

Figure 8.2: Overview of the data entry sheet.

All the modifiable information present in the calculator is presented in this sheet. This includes:

Raw slurry composition: shown in Fig. 8.3, this section includes all the variables needed to undergo the calculations in Modelling sheets.

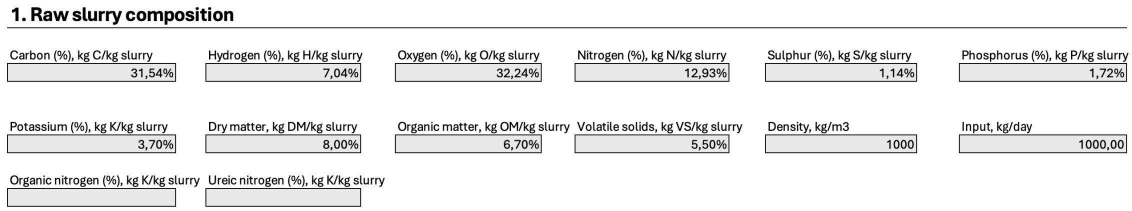

Figure 8.3: Overview of section 1 (Raw slurry composition) of the Entry Data sheet.

In this case, neither organic nitrogen nor ureic nitrogen are not considered in our calculations, but added for the final version of the calculator.

Storage (ST): in this section, two variables can be modifiable:

1. **Duration**: it specifies the storage residence time of the raw slurry. In line with EMEP/EEA, 2023, "< than 1 month" and "> than 1 month" are available as options to choose.

2. **Annual average temperature:** that is, the temperature in ºC of the process, which is assumed to be the same as the environmental temperature.

Fig. 8.4 shows this section of the sheet in detail:

Figure 8.4: Overview of the Storage section of the Entry Data sheet.

Transport 1 (TR1) and transport 2 (TR2: this encloses the transport from farm barns to treatment facilities. For SLS+CP and NT treatment options, it is assumed that no transport takes place. Fig. 8.5 shows in detail this part of the sheet:

Figure 8.5: Overview of the Transport section in Data entry Sheet.

This can also be applied to the transport from the treatment facility to the field where compost, digestate, or raw slurry will be applied. There are three options for entering the transport distance:

1. **Default**: Use a default value when the distance between farms, treatment facilities, and the field is unknown.

2. **Manual**: Enter the known distance manually.

3. **On-site**: Select this option when no transportation is needed, such as when composting occurs within the farm facilities.

Treatments (AD, SLS, CP): detailed in Fig. 8.6, these include the selection of the proposed treatment lines, which are:

1. **Anaerobic digestion (AD):** the modifiable parameters used to model the operation of the anaerobic digestion process are detailed here.

2. **Solid-liquid separation (SLS):** this process is no modulable.

3. **Composting (CP):** the modifiable parameter used to model the operation of the composting (temperature) is found here.

The type of treatment can be selected by clicking the "Data Entry" field (shown in grey color).

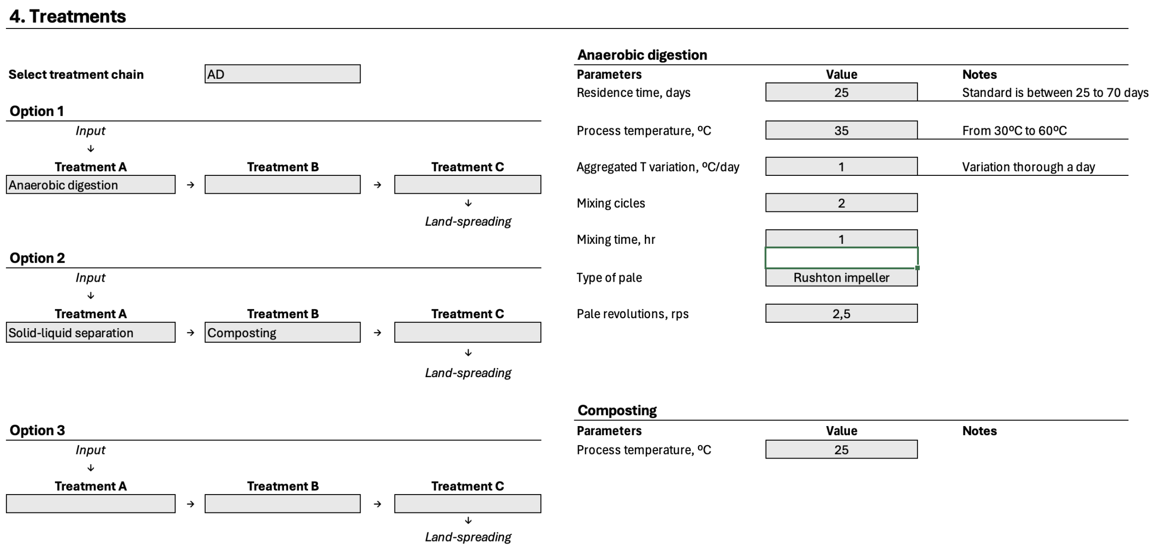

Figure 8.6: Overview of the Treatment section within the Data Entry sheet.

Land-spreading (LP): the modifiability of this section depends on the "Composition sheet", as it will serve as the basis for the calculations of this section. Therefore, there is no section for LP in the Data Entry Sheet.

The **Composition sheet**, shown in Fig. 8.7, summaries the composition of the products resulting from the aforementioned treatment applications (digestate, compost and post-storage raw slurry):

Composition

Summary

In this sheet, the composition of the raw slurry, along with the products generated thorough the entire value-chain, are showcased:

	OM	DM	VS	N	P	K	S	Density
Raw slurry	7%	8%	5%	12,9%	1,7%	3,7%	1,1%	1000 [1]
Anaerobic digestion		2,6% [2]	2,7%	10,1%	1,7%	0,50%	0,90% *	999,71 [1]
Composting				9,24% [4]	0,8% *	0,50% *	0,90% *	500,00
Solid-liquid separation [3]		6,0%		10,0%	1,5%	0,50% *	0,90% *	999,34 [1]

OM = Organic matter; DM = Dry matter; VS = Volatile Solids; N = Nitrogen (total); P = Phosphorus (total); K = Potassium (total); and S = sulphur (total). Units are showcased in relative mass except density, which is showcased in kg/m3.

References

[1] M. Hjorth et al. (2010). Solid–liquid separation of animal slurry in theory and practice. A review. Agron. Sustain. Dev. 30 (1) 153-180.

[2] Prenafeta-Boldú, F.X. i Parera, J. (2020) Guia de les tecnologies de tractament de les dejeccions ramaderes a Catalunya. Departament d'Agricultura, Ramaderia, Pesca i Alimentació (DARP), Barcelona, 72 pàgines.

[3] Møller et al. (2000). Solid-liquid separation of livestock slurry: efficiency and cost. Bioresource Technology 74, 223-229.

[4] Estimated based on N loss from the composting process.

Figure 8.7: Overview of the Composition sheet.

8.1.2.2 Modelling sheets

Modelling sheets are blocked in the tool, since no information needs to be modified. This is because all the modifiable parameters can be found in the Data Entry sheet, as explained above. There are 6 Modelling sheets which go accordingly with the sections from the Data Entry sheet:

Storage sheet: The Storage sheet, shown in Fig. 8.8, encloses the calculations used to model emissions from this stage:

Storage

Process emissions

Storage emissions were modelled in accordance with IPCC and EMEP/EEA guidelines.

Methane

Temperature, ºC	<1 month	>1 month
10	3%	17%
11	3%	19%
12	3%	20%
13	3%	22%
14	3%	25%
15	3%	27%
16	3%	29%
17	3%	32%
18	3%	35%
19	3%	39%
20	3%	42%
21	3%	46%
22	3%	50%
23	3%	55%
24	3%	60%
25	3%	65%
26	30%	71%
27	30%	78%
28	30%	80%

CH4 emissions	**0,01340**	**kg CH4/kg slurry**	[1]
OM loss	**0,01003**	**kg C/kg slurry**	

Ammonia

NH3 emissions	**0,0350126**	**kg NH3/ kg slurry**	[2]
N loss	**0,028795**	**kg N/kg slurry**	

Figure 8.8: Overview of the Storage sheet.

Anaerobic digestion sheet: The Anaerobic digestion sheet, shown in detail in Fig. 8.9 encompasses the procedure related to the emissions estimation for this treatment.

Anaerobic digestion

Process emissions

Emissions from anaerobic digestion were based on the Boyle's Busswell modified equation in line with [1]. It was assumed that the entire substrate composition was organic matter, producing methane, carbon dioxide, ammonia, and hydrogen sulfide. Since the model assumes a full conversion of the substrate, a efficiency factor of 0,8 was included to take into account imperfections during the process. Emissions are given in units of kg/kg VS and were further converted into our functional unit. The results are also shown as a chemical balanced equation in which stoichiometry is modified in real time depending on the initial substrate. Electric consumptions were also considered, namely mixing and temperature maintenance. Energy requirements for temperature maintenance were modelled based on [3], and for mixing in line with [2]. For temperature maintenance, it was assumed a daily absolute temperature variance of 1ºC and the specific heat of swine manure was estimated in line with [3]. For mixing, the type of pale considered was the Rushton pale (curve 1 in Figure AD.1.), and was assumed to have a diameter of 1/3 of the total diameter of the reactor [2].

Energy output, kJ/kg CH4	52500	
Volatile solids, VS	5,50%	
Methane production	**0,010807476**	**kg CH4/kg slurry**
Energy production	**0,157609029**	**kwh/kg slurry**

Stoichiometry

1 *OM* + 0,5 *H2O* ⇒ 1,3 *CH4* + 1,3 *CO2* + 0,9 *NH3* + 0,04 H2S

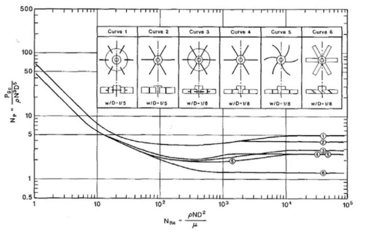

Figure AD.1. Relationship between reynolds number and power number for different type of pales.

Composition

Element	mass ratio, %	molar mass, g/mol
C	30,54%	12,017
H	7,04%	1,008
O	32,24%	15,999
N	12,93%	14,007
S	1,14%	32,065

Coefficients

Coefficient	Value
a	2,54
b	6,98
c	2,02
d	0,92
e	0,04

Mass balance

Reactants & products	molar mass, g/mol	mass, g	mass/FU, kg/kg
OM	83,9	84	--
H2O	18,015	9,0	0,11
CH4	16,049	20,6	0,25
CO2	44,015	55,3	0,66
NH3	17,030	15,7	1,87E-01
H2S	34,081	1,2	0,01

Electric consumptions

Input parameters	Value
Load capacity, kg	25000
Ideal volume, m3	25
Volumetric flow rate, m3/d	1

Temperature control	
Electricity, kWh/kg slurry	2,91E-02

Mixing	[2]
Reynolds number, a.d.	1464,1
↓	
Ψ, a.d.	5
Power, kW	0,10
Electricity, kWh/kg slurry	0,0206%

Figure 8.9: Overview of the Anaerobid digestion sheet.

By introducing the slurry composition in the Data Entry sheet, the model is automatically filled and gives the estimation of electricity generated.

Composting sheet The composting sheet encompasses all the procedures related to emissions estimation for this treatment:

Composting

Process emissions

Methane

Active windrow was chosen as the composting technology since it is the most widespread in Catalonia [5]. Methane emissions have been considered based on temperature, whereas ammonia and nitrous gas in regards of average values. The energy required to turn over the pile was considered as electric consumption based on available literature [3].

Temperature	Vessel	Static Pile	tensive windr	Static Windrow
10	0,5%	0,5%	0,5%	0,5%
11	0,5%	0,5%	0,5%	0,5%
12	0,5%	0,5%	0,5%	0,5%
13	0,5%	0,5%	0,5%	0,5%
14	0,5%	0,5%	0,5%	0,5%
15	0,5%	0,5%	1%	1%
16	0,5%	0,5%	1%	1%
17	0,5%	0,5%	1%	1%
18	0,5%	0,5%	1%	1%
19	0,5%	0,5%	1%	1%
20	0,5%	0,5%	1%	1%
21	0,5%	0,5%	1%	1%
22	0,5%	0,5%	1%	1%
23	0,5%	0,5%	1%	1%
24	0,5%	0,5%	1%	1%
25	0,5%	0,5%	1%	1%
26	0,5%	0,5%	1,5%	1,5%
27	0,5%	0,5%	1,5%	1,5%
28	0,5%	0,5%	1,5%	1,5%

Emissions, kg CH4/kg manure 0,0002748 [1]

Ammonia

Emissions, kg NH3/kg slurry 9,29E-03 [2]

Nitrous gas

Emissions, kg N2O/kg slurry 0,0203 [3]

Electric consumptions

Active windrow, kWh/kg slurry 1,30E-04 [4]

Figure 8.10: Overview of the Composting sheet.

Solid-liquid separation sheet The Solid-liquid separation sheet, depicted in Fig. 8.11, encompasses all the procedures related to emissions estimation for this treatment.

Solid-liquid separation

Pressing screw was chosen as a proxy due to its widespread use in Catalonia [1]. Separation efficiency was calculated in line with [2] taken into account the rduced separation index (the larger the separation index, the greater the amount of compound x in the solid fraction). Energy requirement for the process was reviewed from [3]. Final composition was further converted in regards of the functional unit.

Partition

	DM (%)	TN (%)	TP (%)
Pressing screw (1)	26%	2%	12%
Pressing screw (3)	25%	-1%	11%
Average	**26%**	**0,5%**	**12%**

Solid Fraction	**SF (kg X/kg X total)**	**kg/kg manure**
DM	75%	6%
TN	99,5%	13%
TP	89%	2%
MC	26%	94%

Liquid Fraction	**LF (kg X/kg total)**	**kg/kg manure**
DM	26%	2%
TN	0,50%	0,06%
TP	11,50%	0,20%
MC	75%	98%

Electricity consumption

Type	**Consum (kWh/kg slurry)**	
Pressing screw	0,00053	[3]

Figure 8.11: Overview of the Solid-liquid Separation sheet.

Transport sheet The Transport sheet, shown in Fig. 8.12, encompasses calculations for transport 1 (TR1, from farm to treatment facilities) and transport 2 (TR2, from treatment facilities to the field). In the case of composting or not applying any treatment, transport is accounted as TR.

Transport

Transport 1		Transport 2	
distance (km)	30	Distance (km)	30
tonne-kilometer (tkm)	**0,03**	tonne-kilometer (tkm)	**0,03**

Figure 8.12: Overview of the Transport sheet.

Land-spreading sheet The Land-spreading sheet encompasses all the emission estimations related to the land-spreading of the products resulting from treatment application or post-storage raw slurry.

8.1.2.3 Emission Factor sheets

The emission factor sheets comprehensively entail all the mass and energy flows considered in the LCA. Recollection was based on quantitative data gathering from various sources, including literature reviews, on inputs (e.g., materials and energy) and outputs (e.g., materials and emissions) within every stage. They are divided in:

1. **FE sheet**: provides the emission factors (FE) related to the impact categories chosen for each flow in unitary units.

2. **FE2 sheet**: provides a thorough account of how each flow contributes to the chosen environmental impact categories.

8.1.2.4 Result sheet

In the Result sheet, disclosed in Fig. 8.13, the results (namely LCIA) of the LCA are presented in a clear and organized manner, summarizing the environmental impacts associated with each stage of the slurry's management life cycle. Additionally, it provides a detailed breakdown of the contributions from each stage.

By presenting the results in this structured format, the Result sheet facilitates an easy comparison of the environmental impacts across different stages and impact categories, helping at identifying environmental impact hot-spots and key areas for improvement.

Results

Overall results

The LCIA was performed in line with the Environmental Footprint 3.1., where the following categories were selected to estimate the environmental impact of swine slurry treatment:

Stage	AC	CC	MEU	FEU	TEU	OD	PCOF	RUF	RUMM	WU
Storage	0,00E+00	3,99E-01	0,00E+00	0,00E+00	0,00E+00	0,00E+00	1,35E-04	0,00E+00	0,00E+00	0,00E+00
Transport 1	7,44E-06	3,01E-03	2,03E-06	2,22E-07	2,08E-05	6,82E-11	1,21E-05	4,56E-02	8,40E-09	2,18E-04
Anaerobic digestion - Impacts	2,15E-05	7,23E-03	5,11E-06	7,34E-07	5,42E-05	1,88E-10	2,16E-05	1,67E-01	1,04E-08	3,63E-03
Solid-liquid separation	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00
Composting	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00
Transport 2	7,44E-06	3,01E-03	2,03E-06	2,22E-07	2,08E-05	6,82E-11	1,21E-05	4,56E-02	8,40E-09	2,18E-04
Landspreading - Impacts	1,20E-04	1,03E+00	6,37E-04	5,80E-06	4,25E-03	0,00E+00	7,96E-04	0,00E+00	0,00E+00	0,00E+00
Landspreading - Avoided	-7,77E-03	-9,22E-01	-8,47E-04	-1,97E-04	-1,45E-02	-3,02E-08	-3,01E-03	-1,52E+01	-1,21E-05	-1,11E+00
Anaerobic digestion - Avoided	-1,15E-04	-3,89E-02	-2,75E-05	-3,95E-06	-2,92E-04	-1,01E-09	-1,16E-04	-8,97E-01	-5,58E-08	-1,96E-02
Total	**-7,73E-03**	**4,86E-01**	**-2,28E-04**	**-1,94E-04**	**-1,04E-02**	**-3,09E-08**	**-2,15E-03**	**-1,59E+01**	**-1,21E-05**	**-1,13E+00**

AD = Acidification (mol H+-eq); CC = Climate change, total (kg CO2-eq); MEU = Marine Eutrophication (kg N eq); FEU = Freshwater Eutrophication (kg P eq); TEU = Terrestrial Eutrophication (mol N eq); OD = Ozone depletion (kg CFC11-eq); PCOF = Photochemical ozone formation, human health (kg NMVOC eq); RUF = Resource Use, Fossils (MJ); RUMM = Resource Use, Minerals and Metals (kg Sb eq); WU = Water Use (m3-world equivalents).

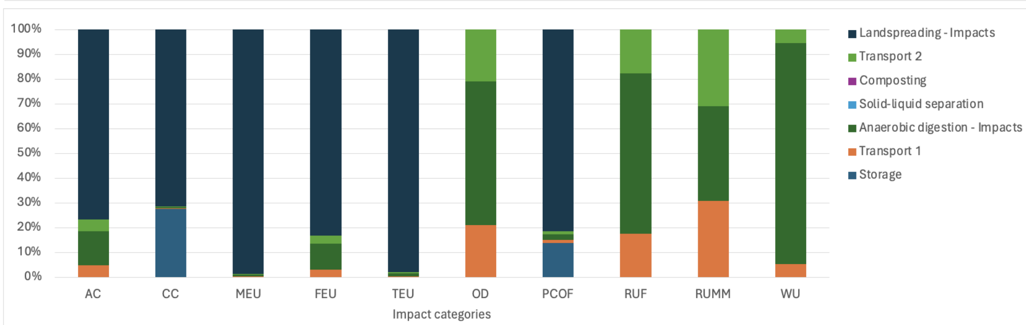

Figure 8.13: Overview of the Result sheet.

8.2 Reducepi testing

Reducepi was tested using SC-A(d), SC-A(s), SC-B(d) and SC-B(s) as input data for CC and MEU impact categories. Fig. 8.14 shows the relative contribution to the impact for every scenario considered:

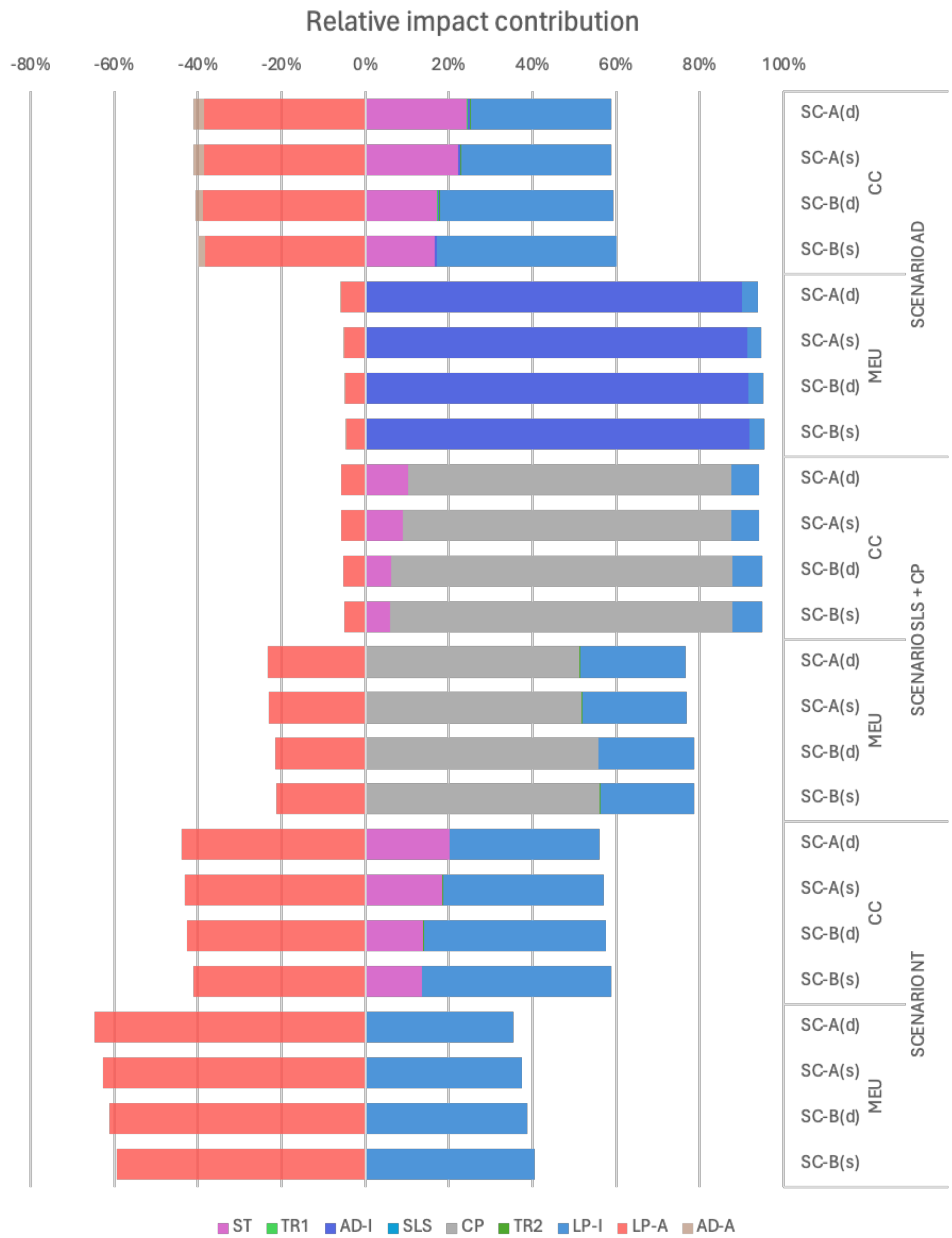

Figure 8.14: Relative impact contribution between cases for scenario AD, scenario SLS+CP and scenario NT.

8.2.1 Climate Change (CC)

Results for impact category CC are showcased in Fig. 8.15. A detailed description of each scenario is provided below.

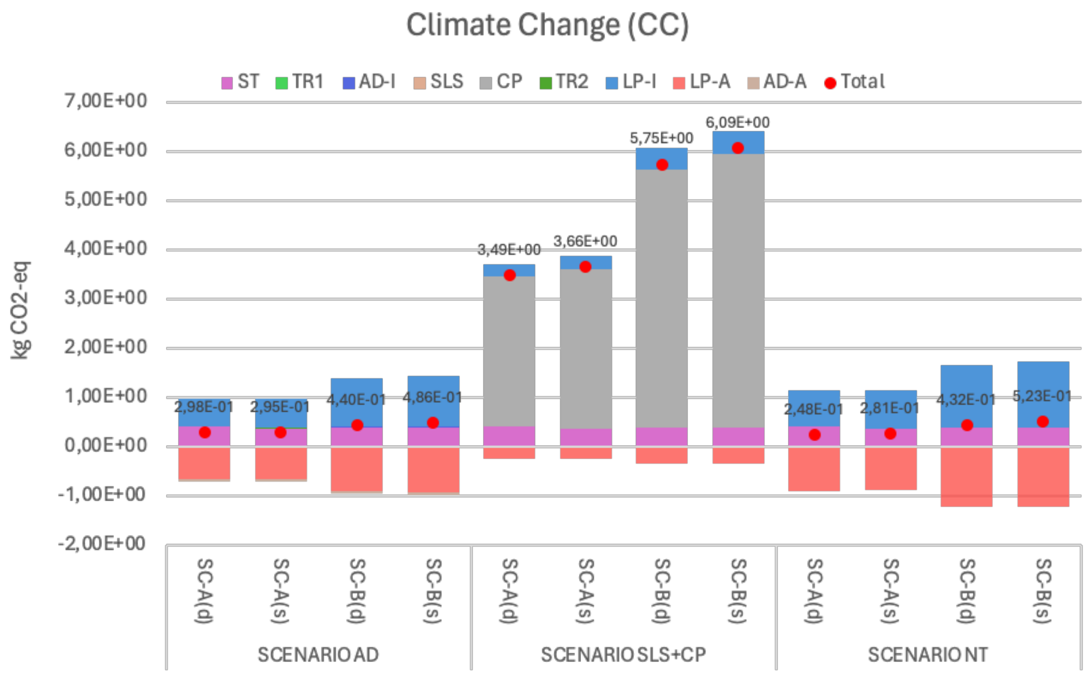

Figure 8.15: Results for CC.

In Scenario AD, the largest impact contributor to CC in all cases is LP, and it is followed by ST. AD has a marginal impact compared with the aforementioned stages. Regarding types of feeds, diet-based feeds tend to outdo sorghum-based feeds in SC-A while in SC-B, the opposite happens. However, differences are minimally discernible. There are also differences between SC-A and SC-B; the former tends, on average, to hold a greater impact than the latter. Variations among cases are mainly caused by LP, as it produces highly variable results stemming from N_2O emissions. Methane emissions were not considered in LP, as it was assumed a total conversion of the OM into biogenic CO_2 and the rest was captured within the soil. Regarding avoided impacts, the biggest contributor is LP, while AD also contributes marginally.

In Scenario SLS+CP, the biggest contributor is CP, followed by LP and ST. Results show that SC-B clearly outdoes SC-A, and sorghum-based feeds tend to outdo diet-based feeds. Variation in CP is mainly caused by N_2O emissions. Even though CP produces CH_4, its impact accounts for around 0,1% of the impact related to N_2O emissions during operation. In addition, CP also has electric consumption during operation, but compared with operation emissions, its contribution is marginal. Regarding avoided impacts, they mainly come from LP and, on average, their contribution is lesser than of in Scenario AD.

In Scenario NT, the biggest contributor in all cases is LP, followed by ST. For SC-A(d), SC-A(s), SC-B(d), and SC-B(s), ST holds 56%, 48%, 32%, 30% of the impact coming from LP, respectively. In regards of avoided impacts, they also mainly come from LP and, on average, their contribution is higher than of Scenario SLS+CP but more or less equal than that of Scenario AD.

In summary, aims at reducing CC should be focused on avoiding CP and prioritizing AD over NT. However, it should be noted that the method to calculate emissions in AD is based in the Boyle's modified Busswell equation and, therefore, it is based on lots of assumptions. Even though a corrector factor of 0,8 was added in line with Achinas and Euverink, 2016, the result may be overestimated as imperfections in mixing and bacterial growth population are not modelled. Therefore, the use of empirical data should be prioritized over using secondary data, if possible. On the other hand, emission factors for LP modelled in line with Corbala-Robles et al., 2018 were generalists and don't take into account local and regional particularities. The impact coming from LP is due to transformation of N compounds in the digestate into N_2O via nitrification-denitrification processes. The total GHG emitted depends on the type of bacterial consortium and strains, optimal biological temperature, and pH. Corbala-Robles et al., 2018 does not delve into the procedure followed by the IPCC guides, and the latter tend to use generalist emission factors rather than locally or regionally focused. As a consequence, this increase the uncertainty in the contribution of the overall impact. In addition, choosing diet-based fees over sorghum-based feeds would also make a huge difference, specially in open and covered composting facilities. Anaerobic digestion would be a suitable candidate since it reduces CC impact and, at the same time, re-values the resulting waste. Furthermore, its implementation can be economically feasible by converting storage lagoons used for aerobic slurry storage into anaerobic lagoons. On the other hand, composting is an effective method for stabilizing nitrogen within the feedstock and, along with anaerobic digestion, is more cost-effective. However, this comes with the drawback that greenhouse gas (GHG) emissions cannot be controlled and may increase significantly. Closed-system composting offers an alternative to this risk, but its high cost makes widespread deployment challenging.

8.2.2 Marine Eutrophication (MEU)

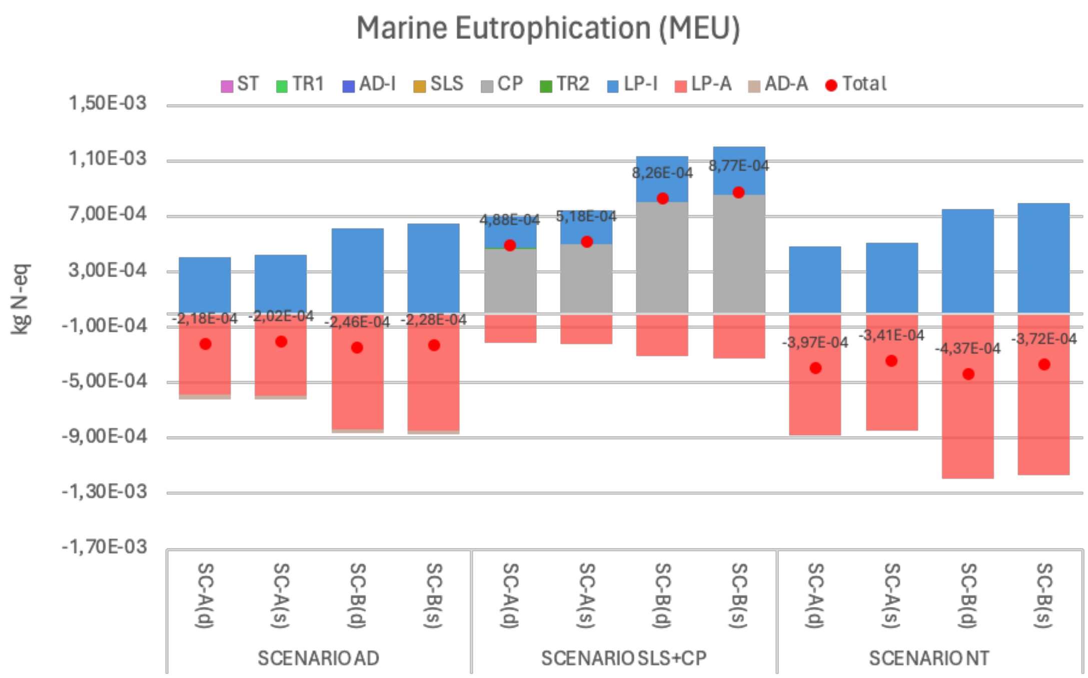

Figure 8.16: Results for MEU.

In Scenario AD, the largest impact contributor in all cases is LP. It is followed by AD and TR, even though they have lower values and their contribution can be dismissed. Regarding types of feeds, sorghum-based feeds tend to outdo diet-based feeds, and SC-B outdoes SC-A. Variation among cases are mainly caused by AD's operation emissions, specifically NH_3; this is supported by the TN content among cases, as seen in Tab. 7.1. Regarding avoided impacts, the highest contributor is LP.

In Scenario SLS+CP, the main contributor in all cases is CP, followed by LP. CP emissions come from operation emissions, while the contribution of electricity consumption to the impact attributed to this stage is marginal. Regarding avoided impacts, the biggest contributor is LP. In Scenario NT, the main contributor in all cases is LP, followed by TR.

Aims at reducing MEU should be focused on favouring diet-based feeds over sorghum-based feeds, and prioritizing anaerobic digestion or no treatment at all over composting. Composting comes with the caveat that emissions cannot be controlled and flee freely into all compartments. This is why LP has lower values in Scenario SLS+CP than in the rest, because part of the nitrogen stored in the substrate is emitted due to NH_3 volatilization while the rest is volatilised or transformed into NO_3^- and NO_x during LP. However, formation and further emission of NO_3^- during CP should have been taken into consideration for a more precise impact analysis. In addition, CSTR reactors are efficient at converting organic material into biogas, but also generate a digestate rich in NH_4^+ through OM breakdown during the digestion process. Land-spreading of the digestate to the field and subsequent contact with air can lead to greater NH_3 emissions through atmospheric volatilization. This NH_3 will react through different pathways to form a wide range of compounds that will return to the land through nitrogen deposition, which can lead to acidification. Since the type of reactor chosen was an isolated CSTR reactor, the NH_3 formed does not escape to the atmosphere and stays within the resulting digestate. This is why emissions of LP in Scenario AD are higher than in Scenario SLS+CP. In addition, Scenario NT may be able to compete with Scenario AD, as the avoided use of mineral fertiliser is greater than in Scenario AD.

8.2.3 General discussion

The Scenario that has the biggest impact in the impact categories considered is Scenario CP. Conversely, the scenario with the least impact is NT. Scenario AD goes in par with NT. Results for MEU are quite surprising because according to scientific literature, composting is a great OM stabiliser, and results for MEU show the opposite. This may be explained because unlike CC, MEU is not inherently tied to a specific time-frame but rather focus on the nutrient content and the immediate environmental impacts associated with them. Therefore, MEU does not reflect the impact that nutrient stabilisation has on the emissions. Overall, efforts at reducing emissions should be focused on choosing diet-based feeds over sorghum-based feeds. Conversely, anaerobic digestion should be chosen over not applying any treatment due to the benefits of producing electricity from biogas and also should be chosen over composting, specifically windrow composting, as it is the treatment with the highest impact.

The biggest drawback that has driven uncertainty in the results is the lack of primary data, which is a problem that LCA specialists and practitioners face on a daily basis. Data holes need to be filled whether with secondary data or with estimations that do nothing but increase the uncertainty in the results. During the development of the tool, efforts were focused on obtaining the most locally and regionally EF possibles but more general EF had to be used and, when that was not possible, models had to be resorted to and therefore, this increased even more the results. For example, the Busswell equation is based on lots of assumptions (exposed in detail in Section 6), and it does not take into account, among other things, the effects of ammonia and VFAs excess in the performance of the anaerobic digestion process. Conversely, composting is a complex process to model. Unlike in anaerobic digestion, an homologous of the Busswell equation could not be found and therefore, other approaches for estimating the possible emissions that could arise from the process were taken into consideration. Even though H_2S is emitted during composting and, in fact, is a main cause of odour, in the testing of the calculator it was not part of the assessment. Therefore, in order to have more accurate emission estimates, efforts should be focused on the use of experimental data, if possible. If that is not possible not the case, the use of more local and regional data should be prioritized.

Conclusion

This master thesis analysed the environmental impact of pig slurry management through two impact categories, namely Climate Change and Marine Eutrophication, from storage through transport, followed by treatment application and finishing with transport and further land spreading of the products. Two treatments and a group control were considered: anaerobic digestion (AD), composting (SLS+CP) and no treatment (NT, group control). The results are presented in regards of the all range of treatments considered: Scenario AD, Scenario SLS+CP and Scenario NT. In addition, two types of feeds were also considered in this assessment: a control group (diet-based feeds) and an alternative group (sorghum-based feeds) divided in two groups: pigs weighing 54 kg (SC-A) and 70 kg (SC-B), respectively. Therefore, this results in four different cases that served as the basis for our slurry composition: SC-A(d), SC-A(s), SC-B(d) and SC-B(s).

The assessment of the CC impact category across the three scenarios considered reveals distinct patterns. In Scenario AD, LP is the largest contributor, with diet-based feeds generally yielding a higher impact in SC-A and sorghum-based feeds in SC-B (however, these differences are minimal). The significant variability in LC is attributed to N_2O emissions, with CH_4 emissions being negligible due to assumption of complete OM conversion into biogenic CO_2. In addition, avoided impacts in this scenario mainly stem from LP. In Scenario SLS+CP, CP is the primary contributor, followed by LP and ST. In all the cases considered, sorghum-based feeds yields a higher impact compared to diet-based feeds, and SC-B surpasses SC-A. In CP, the primary contributor to the impact is N_2O emissions, with CH_4 and electric consumption having a marginal role. Regarding avoided impacts, in this scenario are less significant than in scenario AD. Finally, for Scenario NT, LP is again the main contributor, with ST having a substantial impact share. Avoided impacts are more significant in Scenario NT compared to Scenario SLS+CP, but comparable to AD. Overall, reducing CC impacts should be focused on avoiding scenario CP and prioritizing scenario AD over scenario NT. It's important to note that the AD emission calculations involve numerous assumptions and could potentially overestimate results due to un-modeled imperfections in mixing and bacterial growth. Therefore, the use of primary data should be prioritized over secondary data. Additionally, overly general emission factors used for LP might not account for local and regional particularities, something that increases the results' uncertainty. Conversely, diet-based feeds should be preferred over sorghum-based feeds in all the scenarios considered.

In MEU impact category, Scenario AD identifies LP as the largest contributor, followed by AD and Transport (TR), though their contributions are minimal. Sorghum-based feeds generally have a higher impact than diet-based feeds, and SC-B tends to surpass SC-A. Variations are primarily due to NH_3 emissions during AD operations, correlated with the total nitrogen content. The highest avoided impacts are from LP. Scenario SLS+CP also shows CP as the main contributor, followed by LP. CP emissions are mainly from operation emissions, with electric consumption playing a minor role. The avoided impacts in this scenario are less significant compared to NT. In Scenario NT, LP is again the primary contributor, followed by TR. To mitigate MEU impacts, diet-based feeds should be favored over sorghum-based feeds, and AD or NT should be prioritized over CP. Overall, reducing MEU impacts should be focused on favouring AD or the lack of application of any treatment (NT) over CP. This is because in windrow CP, emissions are uncontrolled and can volatilize freely, resulting in higher operation emissions. On the other hand, this results in lower LP values compared to other scenarios. In addition, control SC - SC-A(d) and SC-B(d) show a better performance than the alternative SC.

At first sight, the LCIA indicates that Scenario CP generally has the highest impact across the impact categories assessed. However, it is noteworthy mentioning that most of the data gathered in the LCI was secondary data since primary data was not available. Reliance on secondary data or models increases uncertainty. For more accurate emission estimates, primary should be developed and, when this is not possible, local and regional data should be prioritized. Even though regionalised EFs were a key effort in this study, general EFs were often used. Overall, prioritizing empirical and localized data will enhance the accuracy and reliability of LCA results, ultimately informing better environmental management decisions and for a more reasonable decision-making.

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