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Environmental Impacts of Alternative Co-substrates for Biogas Production

– A comparative life cycle assessment

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Environmental impacts of alternative co-substrates for biogas production – a comparative life cycle assessment

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ABSTRACT

In recent years, the production of biogas from animal manure has gained increased attention in Denmark, as it has been identified as an important resource to reach the goal of a fossil free society by 2050. In addition, manure management with biogas production has been recognized as a viable way to reduce environmental impacts from animal production systems. Yet, because the methane production potential of animal manure is low, biogas plants depend on the addition of high energy organic wastes as co-substrates to manure, to make their operations profitable. The latter are in short supply and are already being imported in countries like Denmark. The use of different co-substrates and their biogas potential has been investigated, but there is presently a lack of knowledge about the environmental impacts of using one co-substrate versus another. Therefore, this study assessed the environmental impacts of three co-substrates to pig slurry, which are currently underexploited; namely extruded wheat straw, the organic fraction of municipal solid waste, and the solid fraction of liquid-solid separated slurry. A comparative LCA was carried out, where the conventional manure management scheme of slurry storage and subsequent application to arable fields was compared to this three different ways of biogas production. Upon the analysis, extruded wheat straw was identified as a superior co-substrate. This is due to its low nutrient content, high methane yield potential, and low water content, which resulted in the lowest environmental impacts for eutrophication and the most savings for climate change potential. The second best co-substrate was identified to be the solid fraction of separated slurry and lastly the OFMSW had the most environmental impacts out of all scenarios, due to its relationship to energy production from incineration. A sensitivity analysis, where different methane yield potentials were tested for each co-substrate, was performed and the results proved to be robust. However, increase detail to the model is necessary to provide more confidence to the results, since system expansion activities proved to be crucial for the performance of each scenario.

LIST OF ABREVIATIONS

CHP	Combined heat and power
CSTR	Continuously stirred tank reactor
DM	Dry matter
GHG	Greenhouse gas
HRT	Hydraulic retention time
IPCC	Intergovernmental panel on climate change
ISO	International organization for standardization
LCA	Life cycle assessment
LCI	Life cycle inventory
LHV	Lower heating value
OFMSW	Organic fraction of municipal solid waste
OLR	Organic loading rate
OM	Organic matter
ON	Organic nitrogen
PSA	Pressure swing adsorption
SP	Specific heat
TAN	Total ammoniacal nitrogen (ammonium plus ammonia)
TS	Total solids
UHV	Upper heating value
w/w	Wet waste

CHAPTER CONTENTS

Chapter 1. Introduction

The motivation for this study is introduced and broad background information is given. The relevance of the topic is discussed and objectives are presented. Also, the geographical are to which the study is relevant is announced, along with other technicalities of the project.

Chapter 2. Theoretical Background

The theoretical background gives the reader an overview about manure management practices, biogas production practices and important environmental emissions occurring through these activities. The current situation of biogas plants in Denmark is discussed, as well as the use of co-substrates and requirements that make biogas production successful.

Chapter 3. Method

The methodology used for this project, life cycle assessment, is described here. Additionally, key choices for the method are discussed. This includes the use of a ready-made scientific model for classification and characterization and use of a dedicated software package to build the model. System boundaries for the system are described and the way it has been decided to handle biogenic carbon and co-substrate specific activities. Data collection sources, impact categories and equivalency factors used are also mentioned. The time boundary for the study is also revealed.

Chapter 4. Inventory Analysis

The inventory chapter offers a thorough description, complete with flow diagrams, of each scenario modeled. Choices for system expansion are discussed more thoroughly, as well as the assumptions made to arrive at these choices. A table with the most important assumptions for this project is presented. Lastly, results for the inventory analysis, such as mass balances and chemical characterization of substrates throughout the models can be found here.

Chapter 5. Results and Discussion

The results for this project are discussed in this section. The effects of all scenarios modeled are evaluated by impact category and uncertainties found for each scenario are discussed. Also, two sensitivity analysis, performed to check the robustness of the results, are presented.

Chapter 6. Conclusion

Final remarks for the project are given. The initial aim of the project is addressed.

Chapter 7. References

All references can be found in chapter 7.

Chapter 8. Appendix A Calculations

A detailed description of the most important calculations performed to arrive at the results is found here.

Chapter 9. Appendix B Assumptions

A comprehensive list of assumptions with references used for the modeling is presented.

1 Introduction

Climate change concerns are a pronounced global issue, which is continuously discussed by the press and organizations around the world. As a consequence, the world's population faces the daunting challenge of realizing mitigation strategies and measures to counteract the effects of climate change. A common strategy expressed by several governments has been to gradually shift from the use of fossil fuels to more renewable energy sources. In Europe, an objective of 20% energy from renewable sources by 2020 is in place, while in the U.S. a legal target of 15% renewables by 2020 has recently been approved by the government (Appleyard, 2011; Watson, 2009). Additionally, several developing countries have announced similar targets. For example, Brazil aims at a reduction of 26 to 33 Mt CO_2 equivalents and China intends to have 15% of primary energy consumption from renewables. Lastly, Peru expressed a wish to have 33% of their total energy use come from non-conventional energy sources (United Nations Framework Convention on Climate Change, 2011). These choices signify many benefits, not only in greenhouse gas emissions savings, but also as lower levels of pollution and healthier ecosystems. Finding alternatives to fossil fuels and perfecting the use of renewable sources of energy is of high importance. In order to make the right choices when replacing these fuels, the appropriate studies should be carried out so that the environmental benefits desired are ensured. Many alternatives to fossil fuels exist, including wind and solar power, energy from biomass, biofuels, geothermal energy and hydropower. Among these, biomass represents a largely unexploited resource which can generate GHG emissions savings and bioenergy. In this context, biogas has an important role as it can produce energy from different types of biomass.

1.1 Biogas and Benefits

Several processes exist for the production of biogas from biomass sources. These include anaerobic digestion of energy crops, sewage sludge, agricultural residues such as manure and other waste products like industrial wastes and the organic fraction of household waste. Often co-digestion is done in order to achieve higher methane yields. During anaerobic degradation, microbes synthesize the previously mentioned substrates and biogas is produced as a waste product. Biogas is largely composed of methane (55-70%) and carbon dioxide (30-45%), but it also has small amounts of hydrogen gas, hydrogen sulfide, ammonia (together 1-2%), and trace amounts of carbon monoxide, nitrogen, and oxygen (Burton and Turner, 2003). As biogas contains methane, which is a combustible gas, it can be used to generate energy. It may also be upgraded so that mostly methane remains and then it can be integrated into the natural gas grid or used as transport fuel (Jørgensen, 2009). In recent years, the production of energy from biogas has gained increased attention as it has the potential to deal with many of the environmental problems that we currently face, such as eutrophication, acidification of aquatic ecosystems and nutrient recycling.

According to Holm-Nielsen et al. (2009) the worldwide production of animals accounts for 18% of the overall greenhouse gas emissions and 37% of anthropogenic methane emissions. Leaching of nutrients in manure from agricultural fields to water bodies is an important problem caused by this activity. In addition, global emissions of ammonia and nitrous oxide from animal production are high at 64% and 65% of anthropogenic emissions respectively (Steinfeld et al., 2006). The use of manure for the production of biogas is an important aid when seeking to mitigate these impacts. By utilizing biogas, methane and ammonia emissions can be curtailed and leaching of nutrients can more easily be controlled (Holm-Nielsen et al., 2009; Michel et al., 2010). In the EU-27, the potential to develop biogas from manure is great, as there is an annual production of 1500 million tons of manure (Holm-Nielsen et al., 2009). More importantly, it has been found that producing biogas from manure is one of the most cost effective ways to mitigate greenhouse gas emissions with a cost of 13 euro per ton of CO_2 equivalent (Hamelin et al., 2011). It is especially beneficial when all advantages such as the sales of renewable energy, better quality of digested manure as fertilizer, a solution for organic waste treatment, and lower GHG emissions are taken into account. In Europe, the countries that have developed agricultural biogas technology the most are Austria, Denmark, Germany and Sweden followed by Belgium, France, Italy, the Netherlands, the United Kingdom and Spain to a lower degree (Holm-Nielsen et al., 2009). Not only are there differences from country to country in biogas technology, but there are also many different designs for biogas plants, different uses of substrate, and different degrees of political incentives. Such variations call for a deeper understanding of the biogas process.

Very few studies have focused on the environmental implications of using new co-substrates in the biogas process. Thus, the environmental impacts of this action are presently not well quantified. The current study analyzes the environmental impacts of three different co-substrates to pig slurry for biogas production, in comparison to the conventional way to treat animal manure. This will be done by conducting a comparative Life Cycle Assessment (LCA). The co-substrates to be investigated are extruded wheat straw, the organic fraction of municipal solid waste (OFMSW), and the solid fraction of separated slurry. In this way, it is possible to later offer recommendations for the use of co-substrates in biogas plants.

1.2 Goal and Scope

With the intention to create a knowledge base, to aid decision making, and contribute to research and development, different alternatives to manure management are modeled in this study. These include conventional manure management, where animal slurry is applied to arable fields and manure management with biogas production. In addition, manure management with biogas production is modeled in three different ways, with three different co-substrates to pig slurry.

The objectives of this study are:

- To evaluate the potential environmental impacts of including biogas production into manure management strategies
- To identify superior co-substrates to pig slurry in terms of environmental and biogas production benefits
- To determine which areas of the manure management continuum and biogas production are responsible for the most environmental loading
- To offer recommendations for the use of one co-substrate over another and when possible to give advice on how to reduce emissions arising from manure management

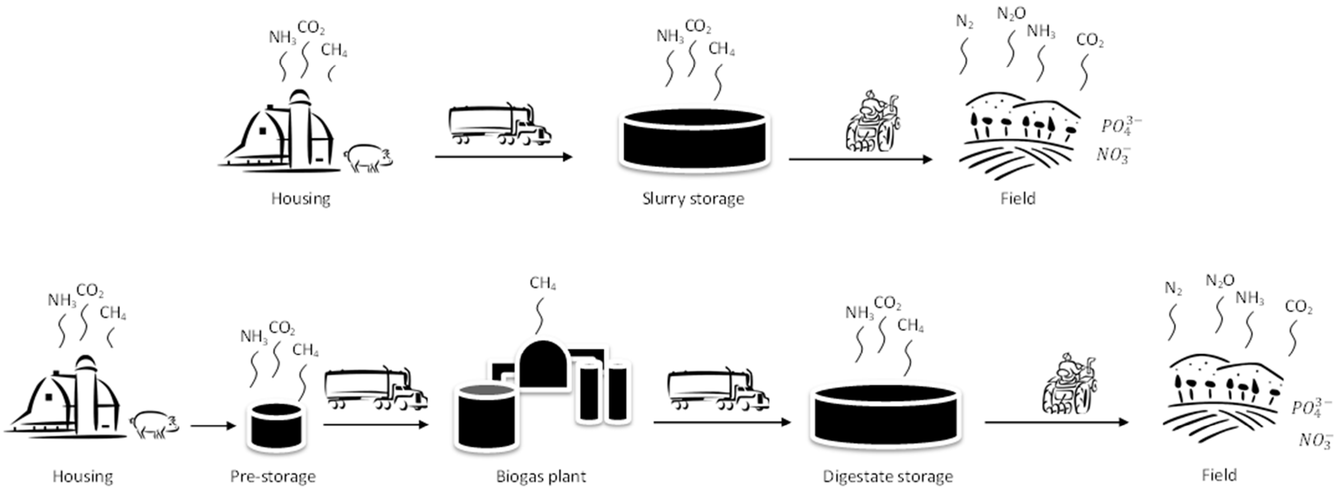
The study is relevant to Denmark only, as all data used and situation modeled represent this geographical region. Also, this study forms part of a larger project, the CLEANWASTE project, which focuses on technologies for sustainable management in the livestock industry. The intended audience here is broad and includes anyone who would benefit from acquiring this knowledge. This could be scientist involved in research and development, members of government, members of energy companies, farmers etc.

2 Theoretical Background

2.1 Manure management continuum

Different ways of handling waste from the production of animals exist. But by far, the most common method of manure disposal is its application on agricultural fields (Burton and Turner, 2003). There are several steps to the manure management continuum, which are shown in Figure 1. The main steps are the accumulation of feces and urine (slurry) in animal housing, slurry storage on farm or close to the fields and application of manure to the fields. In addition, Figure 1 shows manure management when biogas production is incorporated. An additional step is then included, where slurry is taken to the biogas plant to produce biogas and digestate (organic fertilizer).

Figure 1 **The manure management continuum and important environmental emissions occurring through the steps. Top: conventional method of manure management. Bottom: manure management including biogas production.**



Throughout the steps it can be seen that several emissions occur. These are of high importance because of their potential impact on the environment.

2.2 Emissions

Through the various steps of manure management, several compounds are emitted to the atmosphere as gaseous emissions or to the aquatic environment through leaching and runoff. Figure 1 shows the species that have been tracked in the scenarios modeled for this project.

Important emissions from animal housing and slurry or digestate storages include methane, carbon dioxide, and ammonia. Methane and carbon dioxide occur from the anaerobic decomposition of organic matter by bacteria. They are both important global warming gases where carbon dioxide has a global warming potential of 1 kg CO_2 in 100 year period, while methane has a global warming potential of 25 kg CO_2-equivalents in the same period (IPCC, 2006). Ammonia emissions arise from the mineralization of organic nitrogen containing compounds in animal excreta. In this process, organic nitrogen mineralizes into ammonium ions (NH_4^+), which will then partly dissociate into free ammonia that can easily volatize. In the atmosphere, ammonia may oxidize and contribute to acid rain formation and acidification of the environment, as well as eutrophication by increasing the nitrogen available in aquatic ecosystems (Burton and Turner, 2003; Bernet and Beline, 2009).

At the biogas plant, methane emissions are known to escape to the atmosphere through leaks in equipment. Two important sources of methane are the biogas reactor and leaks from upgrading facilities. These have been quantified on few occasions, but reliable measurements were performed by Holmgren et al., 2012 who's results are used for this project.

Finally, gaseous emissions occurring after slurry or digestate is spread on arable fields include, nitrogen gas, nitrous oxide, carbon dioxide and ammonia. Here, carbon dioxide is emitted from degradation of organic matter in a mainly aerobic environment (Jensen and Husted, 2006). Additionally, leaching of phosphorus and nitrogen occur, which are of great importance for eutrophication. All nitrogen emissions in the field are interrelated through the processes of nitrification and denitrification (Burton and Turner, 2003; Chadwick et al., 2011). Through nitrification the ammonium ion is oxidized into nitrite and nitrate; the latter being an important free ion in the aquatic environment, able to cause eutrophication. A by-product of nitrification is nitrous oxide, which is an important global warming gas, with a global warming potential of 298 kg CO_2-equivalents in 100 year period. Nitrous oxide is also produced through denitrification by the reduction of nitrate, in the anaerobic pockets of the soil matrix. This causes a reduction in the amount of nitrate that may leach to the environment. The end product of denitrification is nitrogen gas, which is released to the atmosphere (Jarvis et al., 1996). Biogas production has the ability to change how much organic matter is added to arable fields and thereby it also affects the amount of nitrogen added. The effects of this action are analyzed in this project.

2.3 Biogas

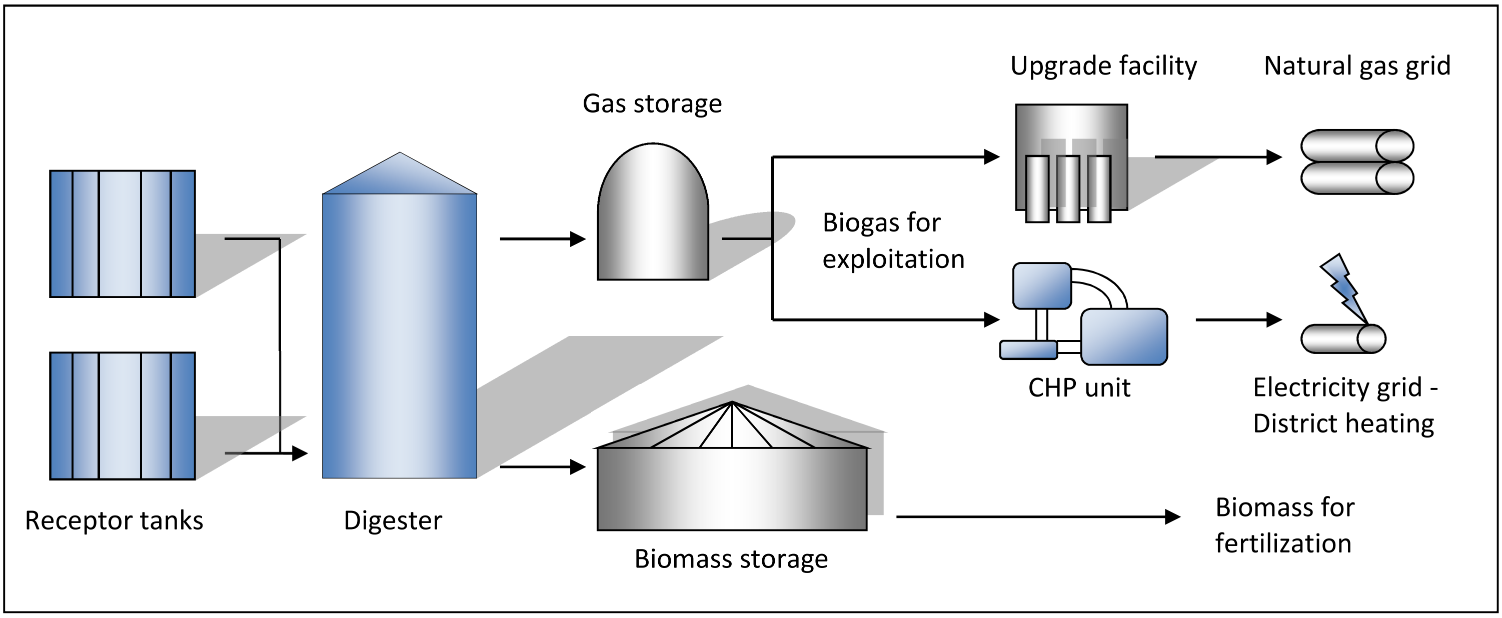
2.3.1 Biogas in Denmark

Currently, there are 22 centralized biogas plants and 60 farm scale plants in Denmark (Raven and Gregersen, 2007; Jørgensen, 2009). According to the latest Danish emissions inventory from agriculture, in 2009 a total of 2.4 million tons of manure were processed, which equals around 8% of the available manure (Mikkelsen et al., 2011). This amount is small when there is a potential to produce 30 PJ of energy from biogas, of which 80% is potential from manure alone (Angelidaki and Ellegaard, 2003). For this reason, the Danish government has recognized the immediate need to make better use of this resource for the production of green energy. In the latest Energy Strategy report, released in February of 2011 by the government, a target to use 50% of the manure to produce biogas by 2050 was communicated (Regeringen, 2011). However, it is important to note that biogas yields and the economic feasibility of plants in Denmark is currently limited by a short supply of co-substrates for slurry e.g. industrial waste and other higher yielding organic wastes. Digestion of manure alone gives low methane yield, with a potential between 10 and a maximum of 20 L CH_4 per L of manure. The value is low compared to the standard of a minimum 20 L CH_4 per L of biomass, which is often cited as the threshold of economic feasibility (Angelidaki and Ellegaard, 2003; Wang et al., 2009). Because manure is not so rich in carbon, it requires co-substrates with a high C:N ratio and high energetic potential to increase methane yields (Wu et al., 2010; Mata-Alvarez et al., 2011). This has given rise to competition between agricultural biogas plants as well as between agricultural biogas plants and waste water treatment plants (Madsen et al., 2011). It has also lead biogas plants to import the highly valued industrial waste from other countries (Jørgensen, 2009; Mata-Alvarez et al., 2011). In light of these facts, research efforts have been focused on finding new co-substrates and pre-treatments of co-substrates that lead to higher biogas potentials. Moreover, there are substrates that are currently underexploited such as, agricultural residues with high lignocellulose structures, which are hard to degrade in the biogas reactor. This includes wheat straw, as well as grasses and silages. Also, organic waste from the food industry and organic by-products from the chemical industry are still largely available for co-digestion in biogas plants (Madsen et al., 2011). These facts were taken into account when choosing to investigate the previously mentioned co-substrates.

2.3.2 Biogas Plants

Between the many configurations that exist for biogas plants, the most common distinction is made between centralized and farm scale plants. These two function quite similarly, but they differ in size and capability to process biomass (Raven and Gregersen, 2007; Jørgensen, 2009). In this study, the discussion is limited to centralized agricultural biogas plants as they function in Denmark.

Figure 2 **Schematic representation of a centralized biogas plant. Adaptation from (Jørgensen, 2009)**



A centralized biogas plant in Denmark receives manure, often in the form of slurry, from several farms in the area. Slurry is the mixture of feces, urine, and water that accumulates below the grates in animal housing units. After the slurry arrives at the biogas plant (Figure 2), it enters receptors tanks from which it is pumped semi-continuously or continuously to the reactor tank. In the reactor tank or digester, decomposition takes place under anaerobic conditions while slurry is continuously stirred. A continuously stirred reactor tank (CSTR) has the advantage of allowing for better contact between substrates and bacteria, which means higher methane yields. The dry matter content in the reactor tank is of a maximum of 12% (Jørgensen, 2009). The biogas produced in the CSTR is then stripped of H_2S, which is a corrosive agent in combination with CO_2 and water vapor, and then shortly stored before transmission to a combined heat and power plant (CHP). Most centralized biogas plants in Denmark have an on-site CHP and the electricity generated there is sold to the electricity grid. The process also generates heat, which is either used for the biogas production process or sold to the district heating grid (Madsen et al., 2011).The effluent from the reactor tank, the previously digested biomass, is stored and awaits until it can either be taken to a farm for further storage or taken to the farm for direct field application depending on the season (Hansen et al., 2006; Jørgensen, 2009). A second option for the utilization of the biogas produced is to upgrade it. That is, to clean all CO_2 and other gases out of it so that mostly methane remains. Several methods for biogas upgrade exist, the most common ones being pressure swing adsorption (PSA) and water scrubbers. With PSA, carbon dioxide is adsorbed to a surface at elevated pressure so that it can be separated from methane. A water scrubber works with the differences in solubility of carbon dioxide and methane; carbon dioxide being more soluble in water than methane, especially at low temperatures (Petersson and Wellinger, 2009). Until now, it had not been economically attractive to upgrade biogas in Denmark. But, this has recently changed with the passing of the new energy agreement, which gives monetary incentives for upgrading biogas (Energipolitik, 2012).

In Denmark, all biogas plants have been fitted so that they can co-digest manure along with other types of organic waste (Raven and Gregersen, 2007). This improves biogas yield tremendously since the digestion of manure alone has proven to lead to low methane yield (Mata-Alvarez et al., 2011). Another feature of the Danish plants is that they run under thermophilic conditions (temperatures of 50-60^{o}C) as opposed to mesophilic conditions (temperatures of 35-37^{o}C) during digestion of biomass. The thermophilic process has proven to provide many benefits such as increased amounts of degradation, shorter retention times and therefore an increased capacity to process biomass, and better sanitation results (Angelidaki and Ellegaard, 2003). However, a thermophilic process can also be more sensitive to process failure, since a higher temperature means bacteria are more sensitive to changes (Mata-Alvarez, 2003; Nielsen and Angelidaki, 2008). Therefore, it is very important that operational parameters are kept at optimal ranges in such biogas plants. Lastly, a novel feature of Danish biogas is the ingenious way in which farmers have become involved with the biogas industry. These farmers have formed non-profit cooperatives, which have in turn organized transportation and storage needs at low costs and receive income from energy sales. The cooperatives also allow the farmers to process their manure free of charge, unlike all other European countries where the farmers have to pay (Raven and Gregersen, 2007). Yet, with all of these advances, manure remains an underutilized resource for the production of biogas and demands innovations to foment its use.

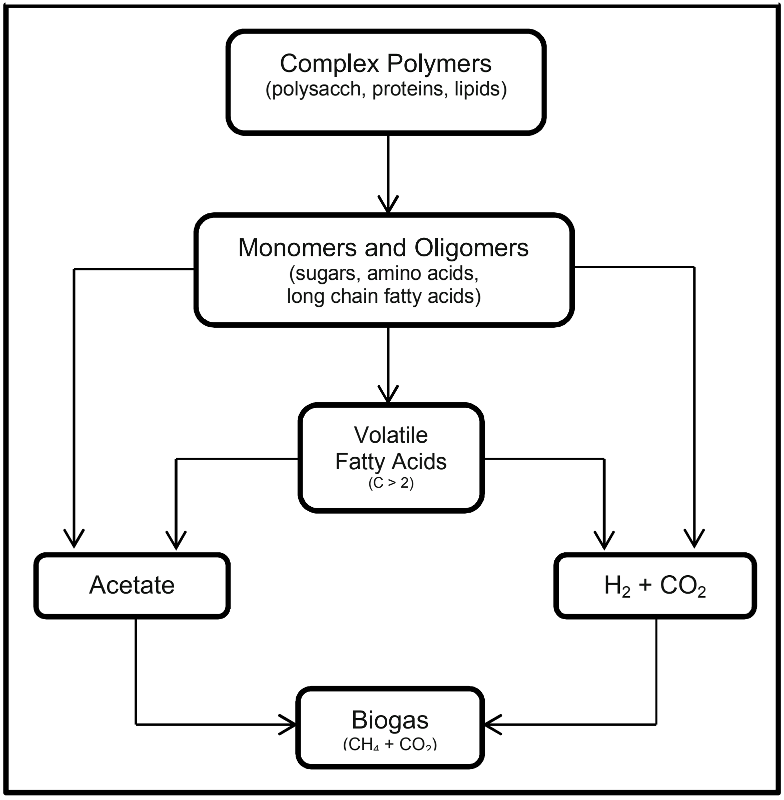
2.3.3 Microbial production of biogas

The production of biogas involves a wide consortia of bacteria, which work together to degrade complex substances into the final products of methane, carbon dioxide and water. It is essential to understand the microbial processes governing biogas production, to reach a better understanding of the importance of substrate composition and operational parameters leading to high methane yields in the biogas reactor. Figure 3 shows the main steps of biogas production, which can be divided into hydrolysis, acidogenesis, acetogenesis and methanogenesis. These processes involve hydrolytic bacteria, acetic acid forming bacteria and methanogenic bacteria (Rojas et al., 2010).

In the first step, complex polymers such as polysaccharides, proteins and lipids are degraded into hydrogen, acetate and volatile fatty acids (Weiland, 2010). In each subsequent step, the products of the previous step are broken down and used for bacterial growth (Rojas et al., 2010). The final step, methanogenesis, involves two groups of methanogens, one able to make methane out of acetic acid and the other from CO_2 and H_2.

The hydrolytic step is especially important as it produces various acids that VFA-degrading bacteria must degrade. These type of bacteria have a slow growth rate. The more fat present in the substrate the more long-chain fatty acids that will be produced, with the potential to inhibit the process if it becomes too acidic. Acidification is a known inhibition to the biogas process. Thus, it is crucial that there is a steady degradation of VFAs in the reactor. On the other hand, substrates rich in protein will produce more ammonium and ammonia, which is also toxic to bacteria at high levels (Bernet and Beline, 2009; Jørgensen, 2009). This is known as nitrogen or ammonia inhibition.

Figure 3 Microbial degradation of complex matter, based on (Gujer and Zehnder, 1983)



Another substrate characteristic affecting biogas production is macronutrient content, as bacteria need these to grow. In particular, the C:N ratio should be less than 30:1. At low C:N ratio there may be an over production of ammonia, which can be lowered by addition of carbon. At high C:N ratio, nitrogen becomes the limiting factor for bacterial growth (Hashimoto, 1983; Torres-Castillo et al., 1995). Aside from substrate composition, the operation parameters of the biogas plant are another aspect which highly influences the stability of the process.

2.3.4 Operation Parameters

Operation parameters refer to those factors that may be influenced by the plant's manager, which affect the microbial degradation process, such as temperature, pH and others. The effects of such parameters are discussed next.

2.3.4.1 Hydraulic Retention Time

Hydraulic retention time (HRT) refers to the amount of time the biomass spends in the reactor from when it enters to when it exists. With longer retention times, biomass may be degraded to a higher extent and methane potentials will approach the theoretical methane potential of a substrate i.e. mathematically calculated methane potential according to substrate chemical composition. In Denmark, the most common retention time for centralized biogas plants in 15 days (Hansen et al., 2006). Thus, the biomass is not degraded to its full extent and often only 30-60% of OM of substrates containing complex molecules, such as animal manure, is decomposed (Nielsen and Angelidaki, 2008; Jørgensen, 2009).

2.3.4.2 Organic loading rate

The organic loading rate (OLR) refer to the rate at which biomass is fed to the reactor. This rate must be in line with the growth rate of methanogens and the rate of removal of organic acids to avoid process inhibition. If more biomass is added than the bacteria can degrade, there is a risk of acidification (Lindorfer et al., 2007; Jørgensen, 2009).

2.3.4.3 Temperature

Temperature influences the rate of biochemical processes. In a thermophilic process there are less bacteria species than in a mesophilic one. Also, at higher temperatures methane production is more efficient, but at the same time bacteria become more sensitive to changes in temperature (Weiland, 2010). Higher temperatures may worsen ammonia inhibition as the equilibrium between ammonium and ammonia shifts to the ammonia side in such conditions (Torres-Castillo et al., 1995; Angelidaki and Ellegaard, 2003). Bacteria are able to cope with small variations in ammonia, but cannot cope with sudden increases (Jørgensen, 2009)

2.3.4.4 pH

The optimum pH for methane production is between 7 and 8. There is severe inhibition if the process' pH falls below 6.5 or above 8.5 (Wang et al., 1999). Biogas plants digesting animal manure generally run on the higher spectrum (8-8.3) as manure is high in ammonium, which is very alkaline. A high pH will shift the ammonium-ammonia equilibrium to the ammonia side causing inhibition (Hashimoto, 1983). However, the reactor has a large buffering capacity and it is not so easy to alter the environment (Jørgensen, 2009). The accumulation of ammonia increases the pH inside a reactor, while the accumulation of VFAs decreases the pH (Weiland, 2010).

By knowing these factors, steps may be taken by the operation manager so that optimal conditions may be kept inside of the reactor. As anaerobic digestion increases the ammonium content of pig slurries, which are already high in pH, ammonia content may become problematic. As discussed earlier, optimal substrate macronutrient composition e.g. paying attention to C:N ratios, is a helpful tool to prevent inhibition (Torres-Castillo et al., 1995).

3 Method

3.1 LCA

Life Cycle Assessment is a tool that allows its users to assess the potential environmental impacts of a product or service through its whole life cycle (Baumann and Tillman, 2009). Emissions released to the environment caused by energy and materials needed for the production of a product or service, are identified and tracked from cradle to grave (Cherubini, 2010). With LCA it is possible to compare the potential environmental impact of using one product or service over another. More importantly, LCA is a tool governed by international standards, set forth in ISO 14040-14044 (ISO, 2006). This tool is also recognized by the scientific community as one of the best methodologies for the quantification of greenhouse gas emissions of biomass based energy (Cherubini, 2010). In this study, the LCA methodology has been followed to evaluate alternatives of manure management. In order to do this, the four steps of a LCA where followed.

The four steps to a LCA are:

- Goal and Scope definition: in this stage the goal of the study must be clearly stated, as well as the purpose for carrying out the study and its intended audience (section 1.2). A functional unit and boundaries for the system must be decided on and the environmental impacts which will be considered.
- Inventory Analysis: here a flow model that represents the life cycle desired is constructed. All data is collected for all inputs and outputs and calculations of emissions and resource use are carried out in relation to the functional unit.
- Impact Assessment: in this step, life cycle inventory results (LCI), that is the emissions and resource use calculated, is converted into environmental loading information. First, the LCI is classified or sorted to reveal the impact category each contribution affects and then the LCI is characterized. Characterization means that different emissions, e.g. CH_4 and N_2O are applied a factor and converted into one indicator such as CO_2 to make better sense of the results. There are ready-made scientific models for classification and characterization that have been built over the years by the scientific community. These models simplify the LCA work and eliminate subjectivity that could be present if classification and characterization was done by the user.
- Interpretation: this is the last step of the LCA where all conclusions are drawn in relation to the study's stated objectives (Baumann and Tillman, 2009; Cherubini, 2010).

The dedicated software package Gabi 5, by PE International, was used to construct a model to compare the environmental impacts of three alternatives to conventional manure management. The impact assessment method for classification and characterization used here is the ready-made ReCiPe 2008 method, at the midpoint level (Goedkoop et al., 2009). To evaluate impacts at a midpoint level means e.g. to see how a specific emission affects climate change or acidification, as opposed to an endpoint, where the result will be expressed as damage to human health or species loss. As there are higher uncertainties associated with an endpoint methods it was deemed preferable to present results at the midpoint level.

3.1.1 Functional Unit

The functional unit is chosen in order to make the results for the alternative product or service comparable. The functional unit must reflect the function of the product system (Baumann and Tillman, 2009). It is the unit, or the reference flow, to which all input and output flows are related (Hamelin et al., 2010) or as stated by ISO 2006 “a reference to which the input and output data are normalized”.

In the present study, the functional unit is the management of 1000 kg of pig slurry.

All inputs and outputs from here on are related to this unit and it is the basis by which final results will be shown.

3.1.2 System boundaries

In an LCA it is necessary to identify which processes or steps in the life cycle should be included. The system boundaries help define where the system begins and where it ends.

For this study the system begins when the pig excretes the ‘slurry’ in the animal housing and it ends when the slurry is applied on the field for fertilization, taking into account the emissions arising after field application. All processes within the beginning and the end related to slurry handling are included in the LCA. That means that processes included are: transport from one step to another, processes occurring in the animal housing, long-term storage, field application, electricity consumption needed for stirring and pumping animal slurry, and avoided mineral fertilizer production and field processes. For the biogas scenarios additional processes are: biogas production processes, pre-storage, upgrade, digestate long-term storage, electricity and heat consumption of biogas production, electricity consumption of upgrade, transport of slurry to the biogas plant, transport of co-substrates to the biogas plant, pre-treatment of co-substrates such as separation of the solid fraction, extrusion of straw and biopulping of the OFMSW, avoided processes from co-substrate alternative use etc.

The system boundaries for this study do not include: anything that happened before the animal excreted the slurry such as, production of the animal, building of facilities, production of feed, etc. Likewise, anything that happened before the co-substrates were pre-treated and ready for transport to the biogas plant is not included e.g. production of straw, source separation of the organic fraction, building of a screw-press, etc.

The LCA was modeled for a 10 year period and is applicable to Danish conditions only. The time frame was chosen on account of calculation performed by the CLEANWASTE group, which are a point of departure for this project (CLEANWASTE). This larger project uses the FARM-N model; a nitrogen fate tool that calculates field emissions after the addition of organic matter for a 10 year period (FARM-N;Hutchings et al., 2012c). In order to keep things consistent, it was deemed necessary to make the rest of the model calculation for the same time frame. The only exception to this was made when calculating avoided emissions from straw left on the field for the straw scenario. This was done for 100 year period with DAISY, an agroecosystem model that is able to simulate water transport, nutrient and carbon dynamics in agricultural systems (Abrahamsen and Hansen, 2000; Bruun et al., 2003). The latter is a limitation of the study, which had to be taken as there was no better way available to calculate these avoided emissions.

While other studies make a distinction between anthropogenic carbon emissions and biogenic carbon emissions, no such distinction is made here. The distinction between these two was considered, but it was decided that for simplicity’s sake biogenic carbon would not be included. To include it could mean a heightened level of subjectivity enters the study. For example, to make a decision that says the methane emission from storage units is considered anthropogenic, but the carbon dioxide emission is biogenic complicate the matter. A lack of knowledge to make such decisions made it apparent that to treat all carbon in the same way was better for this study.

The data gathered comes from Danish studies, the slurry is of Danish chemical characterization based on Danish feed standards, the co-substrates are all of Danish chemical characterization, emission factors, biogas plant operational conditions, average crops, soils, and temperatures all represent Denmark. The study results assess alternatives for the future of biogas production and it is estimated that they could apply for the next 5 to 7 years.

3.1.3 System expansion

When multiple products are produced from one process and have to be related to one functional unit, this is called an allocation problem. In such a case, a decision must be made as to which of the products is responsible for the emissions produced. This often requires a somewhat arbitrary partitioning of the emissions and subsequent allocation of them to each of the products (Baumann and Tillman, 2009).

The ISO standards state that allocation should be avoided, whenever possible. This can be done in two ways: by increasing the level of detail of the model, or by system expansion (ISO, 2006). Through system expansion, the additional functions related to the co-products are included in the model, thus avoiding allocation (Cherubini, 2010).

In the present study, the ISO recommendations have been followed and allocation has not been done. System expansion, especially in the biogas scenarios, has given rise to many 'co-substrate specific activities', which are discussed in detail under 'Scenario Descriptions'. In the results, the environmental loadings and savings from these co-substrate specific activities have been attributed to biogas production. The reasoning behind this is that these activities are a consequence of using each co-substrate for biogas production and must then be grouped to this action.

It is important to realize that when system expansion is done, though preferable to allocation, an uncertainty is introduced. This is due to the decisions that must be made on how to model the additional processes (Baumann and Tillman, 2009).

3.1.4 Impact Assessment

The impact categories taken into account in this study are climate change potential, fossil depletion potential, marine water eutrophication potential, freshwater eutrophication potential and terrestrial acidification potential.

When the classification and characterization step is done using ReCiPe 2008, the equivalency factors used for CO_2, N_2O, and CH_4 are those specified by the IPCC for 100 year period (Goedkoop et al., 2009). This means that the emissions occurring after 10 years of field application, at the end of the 10th year, are tracked for another 100 years as part of the impact assessment. The equivalency factors are 1, 25, 298 kg CO_2 equivalents for CO_2, N_2O, and CH_4, respectively.

4 Inventory Analysis

Below the scenarios modeled in this project are described in detail. System expansion and co-substrate choices are discussed. Mass balances resulting from the model calculations are also shown. A full list of assumptions with references can be found in APPENDIX B.

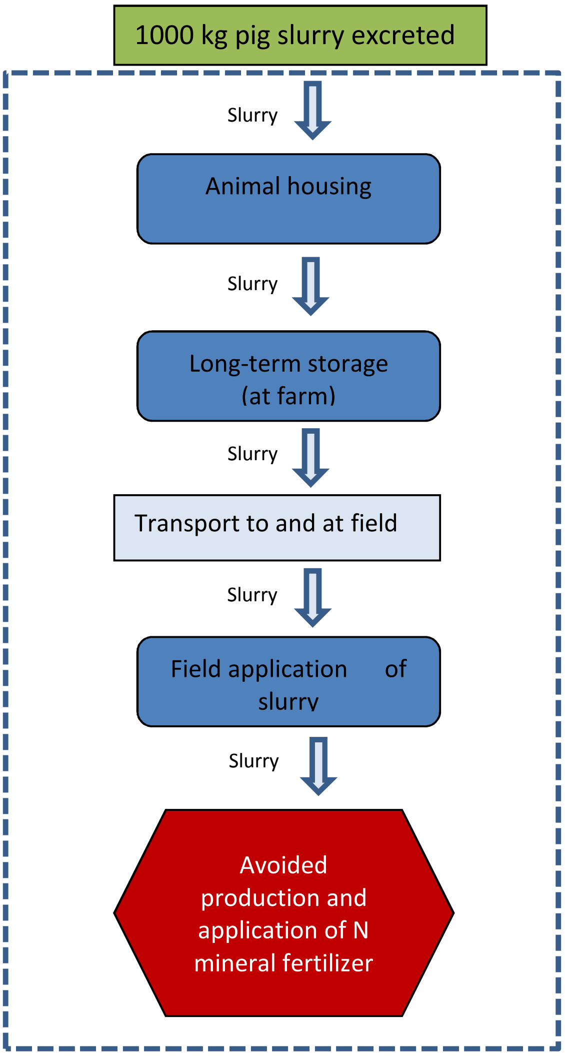
Data for the inventory was collected from many sources. Results from Danish studies where used in most cases, with a few exceptions. When exceptions were made, studies with similar conditions to Denmark were used (temperature, retention time, storage period etc…). As this project is part of the CLEANWASTE project, several of the assumptions applying to this larger project were adopted to keep consistency. Also, calculations for the baseline scenario come from this group, where the model FARM-N was used.

4.1 Scenario Descriptions

4.1.1 Baseline Scenario

The baseline scenario modeled here, represents conventional manure management practice in Denmark. In short, this means that animal manure is excreted in-house, stored at the farm in covered storage and finally applied to the field in April as organic fertilizer.

Figure 4 **Process flow diagram for Baseline Scenario. In green, the functional unit, major steps of the life cycle in blue, avoided processes in red.**



- Animal housing: in the animal house, finishing pigs excrete urine and feces that accumulate under a fully slated floor. The slurry mixes with drinking water spilled by the pigs and some of this water evaporates. N_2 and N_2O emissions are assumed to be negligible due to short residence time (FARM-N). After this time, the slurry is stirred and pumped into a long-term storage at the farm.
- Long-term slurry storage: the slurry is stored outdoors in a concrete storage unit covered by a tent. There is no crust formation over the slurry as this is pig slurry. As the storage unit is covered, it is assumed that no water is added by precipitation and no water leaves by evaporation. The slurry is stored for 6 months, until it can be applied to the field. Note that all storages in this study have the same physical characteristics.
- Transport to and at field: the slurry is stirred and loaded for transport. The total distance from the long-term storage to the field, and at the field while spreading, is assumed to be 8 km. The slurry is spread by trailing hose.
- Field application: at the field, slurry organic matter degrades, giving rise to several emissions. An average cereal crop grows on an average Danish soil in the field.
- System Expansion:
    - Avoided production and application of mineral N-fertilizer: slurry application replaces production and emissions from N-mineral fertilizer application. The mineral fertilizer ammonium nitrate is replaced by the slurry at an efficiency rate of 75% (Hutchings et al., 2012c). It is assumed that the fields near the animal farms have a surplus of phosphorus built up in the soil. This is based on the fact that these fields have been receiving pig slurry, high in phosphorus, for many years and it has been estimated that they won’t need P-fertilizing for years to come (Whalen et al., 2001). Therefore, P-mineral fertilizer is not replaced by the application of slurry, only N-mineral fertilizer. Only fields far away, 100km away, are in need of phosphorus fertilization, since there is no animal production near these fields in Denmark. Additionally, potassium content of the slurry has not been included in this study.

Since emissions from the field have been identified to be a very important for the project, field emission rates for all scenarios and efficiencies of digestate, slurry and solid fraction to replace mineral fertilizer are presented in Table 1. Note: information about avoided mineral fertilizers is found in the first column.

Table 1 Nitrogen and phosphorus emission rates in the field and mineral fertilizer replacement efficiencies of the various substrates utilized throughout the model.

	Mineral Fertilizer Ammonium nitrate or Single superphosphate	Substrate			Reference
		Slurry	Digestate	Solid fraction	
N-replacement efficiency, %	n/a	75.00	80.00	65.00	FARM-N, Chantigny et al. 2007[a]
NH_3 rate, kg NH_3-N kg^{-1} N	0.03	0.16[b]	0.16[b]	0.39[b]	FARM-N
N_2 rate, kg N_2-N kg^{-1} N	0.02	0.04	0.03	0.04	FARM-N
N_2O rate, kg N_2O-N kg^{-1} N	0.02	0.02	0.02	0.02	FARM-N, IPCC 2006[c]
NO_3 rate, kg NO_3-N kg^{-1} N	0.41	0.39	0.48	0.33	FARM-N
Crop N uptake	0.59	0.52	0.49	0.44	FARM-N
P-replacement efficiency, %	n/a	100.00	100.00	100.00	FARM-N
P rate, kg P kg^{-1} surplus P	0.05	0.05	0.05	0.05	Nielsen and Wenzel 2007
Crop P uptake, kg P kg^{-1} P	0.28	0.61	0.64, 0.64, 0.59[d]	0.53	FARM-N, Calculation

[a] reference valid for digestate replacement efficiency
[b] unit is kg NH3-N kg^{-1} TAN
[c] reference valid for 0.02 rate
[d] calculation of crop P uptake for straw, OFMSW, and Separation scenarios respectively

The chemical characterization of the 1000 kg of pig slurry was obtained from calculations done in the FARM-N model. A scientific description of this model, which specializes in losses of nitrogen into the atmospheric and aquatic environments from arable land, can be found by following the link in the references and also in the future publication (Hutchings et al., 2012c). The chemical characterization includes total nitrogen content (both organic nitrogen and ammoniacal nitrogen) of the slurry, water, volatile solids (organic matter), ash without phosphorus and phosphorus. These species are tracked through each step of the life cycle for all scenarios.

Table 2 Chemical characterization, slurry

Chemical characterization of reference slurry amounts per 1000 kg slurry	
Total mass, kg	1000.00
Total Solids, kg	78.70
Organic matter, kg	52.58
Total nitrogen, kg	6.94
TAN, kg	4.96
Organic nitrogen, kg	1.98
Ash, kg	24.97
P, kg	1.15
H2O, kg	916.34

Changes to the initial 1000 kg of slurry through the steps of the manure management continuum are shown in Table 3.

Table 3 Baseline scenario mass balance through life cycle, amounts per 1000 kg slurry

	Slurry ex-animal	Slurry ex-house	Slurry ex-storage	Field
Total mass, kg	1000.00	939.35	929.41	
Total Solids, kg	78.70	68.97	61.04	
Organic matter, kg	52.58	42.86	34.93	
Total nitrogen, kg	6.94	5.61	5.56	
TAN, kg	4.96	3.99	4.25	
Organic nitrogen, kg	1.98	1.61	1.32	
Ash, kg	24.97	24.97	24.97	
P, kg	1.15	1.15	1.15	
H2O, kg	916.34	866.39	864.12	
OM degradation rate, kg kg^{-1}	n/a	0.19	0.19	0.93

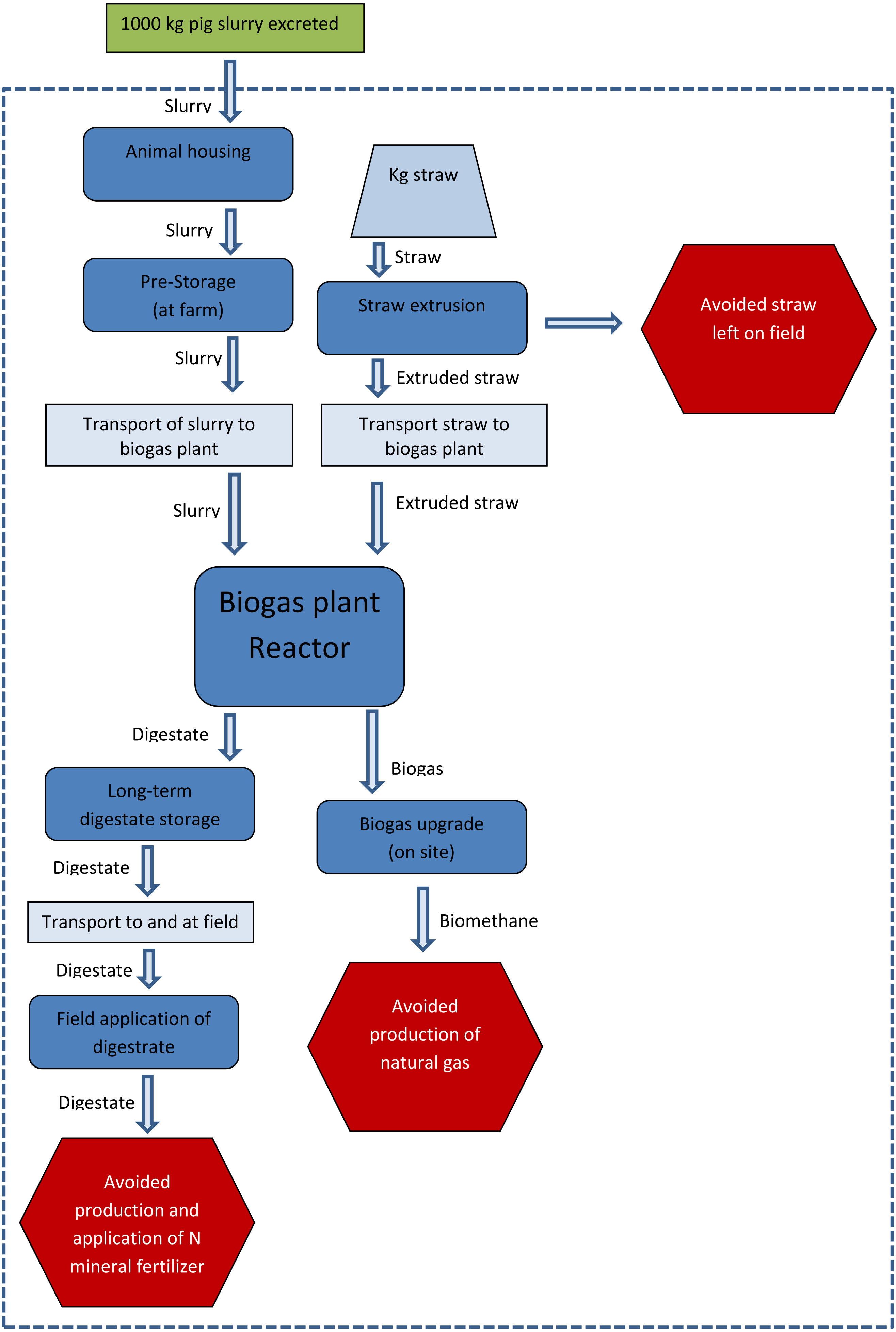
4.1.2 Co-Digestion with Straw

The straw scenario is one of three scenarios which include the production of biogas as a manure management strategy. This means that the 1000 kg of pig slurry excreted in the baseline scenario is taken to a biogas plant for co-digestion with another substrate, in this case extruded wheat straw. Straw extrusion is a process that increases the break-down of complex compounds in the straw, by causing depolymerization of cellulose, hemicellulose, lignin and protein. The potential methane yield for straw is increased by 70% by the extrusion process (Hjorth et al., 2011). For a 15 HRT, the methane yield for extruded straw is 474.7 L CH_4 kg^{-1} OM; calculated based on the methane yield of wheat straw for 15 retention time found in Wang et al., 2009. A total of 5% extruded straw per w/w was added to the digester for co-digestion with slurry (Møller, 2012). This resulted in an addition of 53.47 kg of total solids. For comparability, the same amount of total solids was added to the digester in each biogas scenario.

The animal housing step is the same for all biogas scenarios as for the baseline. After animal housing the following steps occur:

- Pre-storage of slurry: after the slurry leaves the animal housing it is pumped and stirred to a pre-storage at the farm. Here it is stored for a brief period of 10 days. This storage is also made of concrete and covered by a tent. After said period it is picked up and taken to the biogas plant.
- Extrusion of straw: in preparation for digestion in the biogas plant, straw is gathered at the farm and extruded. Straw is assumed to be in surplus and would have otherwise been left on the field.
- Transport of slurry and straw to biogas plant: both the straw and the slurry are transported 5.6 km to the nearest biogas plant. This is the average distance of farms contributing manure to biogas plants in Denmark (Al Seadi, 2000). The slurry is pumped and stirred before transport.
- Biogas plant reactor: the slurry and straw arrive at the biogas plant to be co-digested. It is assumed that they are co-digested immediately (Jørgensen, 2009). The biogas reactor has a hydraulic retention time of 15 days and works at thermophilic temperature (Hansen et al., 2006). The reactor produces two products: biogas and organic fertilizer, referred to as digestate.

Figure 5 Process flow diagram of Co-Digestion with straw Scenario. In green, the functional unit, major steps of the life cycle in blue, avoided processes in red.



- Biogas upgrade: the upgrading facility is assumed to be on-site at the biogas plant. After the biogas is upgraded to biomethane quality, it is injected into the natural gas grid. Thereby, production of natural gas is avoided. Biogas is upgraded, rather than combusted in a CHP, which is the most common practice at the moment. The decision to upgrade was made as a consequence of recently passed legislation, which favors biogas upgrade (Energipolitik, 2012). Fugitive emissions from biogas production can be found in two places; from leaks in the reactor tanks and from leaks in the upgrading facility (Holmgren et al., 2012).
- Transport of digestate to farm: after it leaves the reactor, the digestate is pumped and stirred and taken back to the farm where it will be stored until application in the field. The distance traveled to the farm is equal to the distance traveled from the farm to the biogas plant.
- Long-term storage of digestate: the digestate is stored in a covered storage, equal to the storage for the baseline scenario. The time of storage is estimated to be around 9 months at low Danish temperatures.
- Transport to and at the field: the same as for the baseline, refer to section 4.1.1.
- Field application of digestate: the same as for the baseline, refer to section 4.1.1.
- System Expansion:
    - Avoided production and application of mineral N-fertilizer: the same as for the baseline with the exception that digestate replaces mineral fertilizer with an efficiency of 80% (Chantigny et al., 2007), refer to section 4.1.1.
    - Avoided straw left on the field: if the straw was not taken to the biogas plant, its alternative use would be to leave it on the field. This assumption is justified by the large amounts of straw that are left on Danish fields each year, around 2.1 million tons or 38% of the total straw production (Skøtt, 2011). The avoided impact of the straw on soil carbon is taken into account for 100 year period, modeled with DAISY.

The chemical characterization of the wheat straw is based on Wang et al. 2009, whose study is based on Danish conditions.

Table 4 **Chemical characterization, wheat straw**

Chemical characterization of wheat straw amounts per 1000 kg slurry	
Total mass, kg	58.12
Total Solids, kg	53.47
Organic matter, kg	49.98
Total nitrogen, kg	0.36
TAN, kg	0.04
Organic nitrogen, kg	0.32
Ash, kg	3.47
P, kg	0.02
H_2O, kg	4.61

Changes to the reference slurry and co-substrate through the manure management continuum that includes biogas production can be seen in Table 5. From ‘Slurry ex-prestorage’ to ‘Digestate ex-reactor’ it can be seen that the total mass increases. This is due to the addition of the co-substrate for degradation in the reactor. Even though OM is added at this point, it can be seen that a large part of it is degraded in the reactor and a lower amount of OM goes to 'Digestate ex-storage'.

Table 5 Straw scenario. Mass balance through life cycle, amounts per 1000 kg slurry

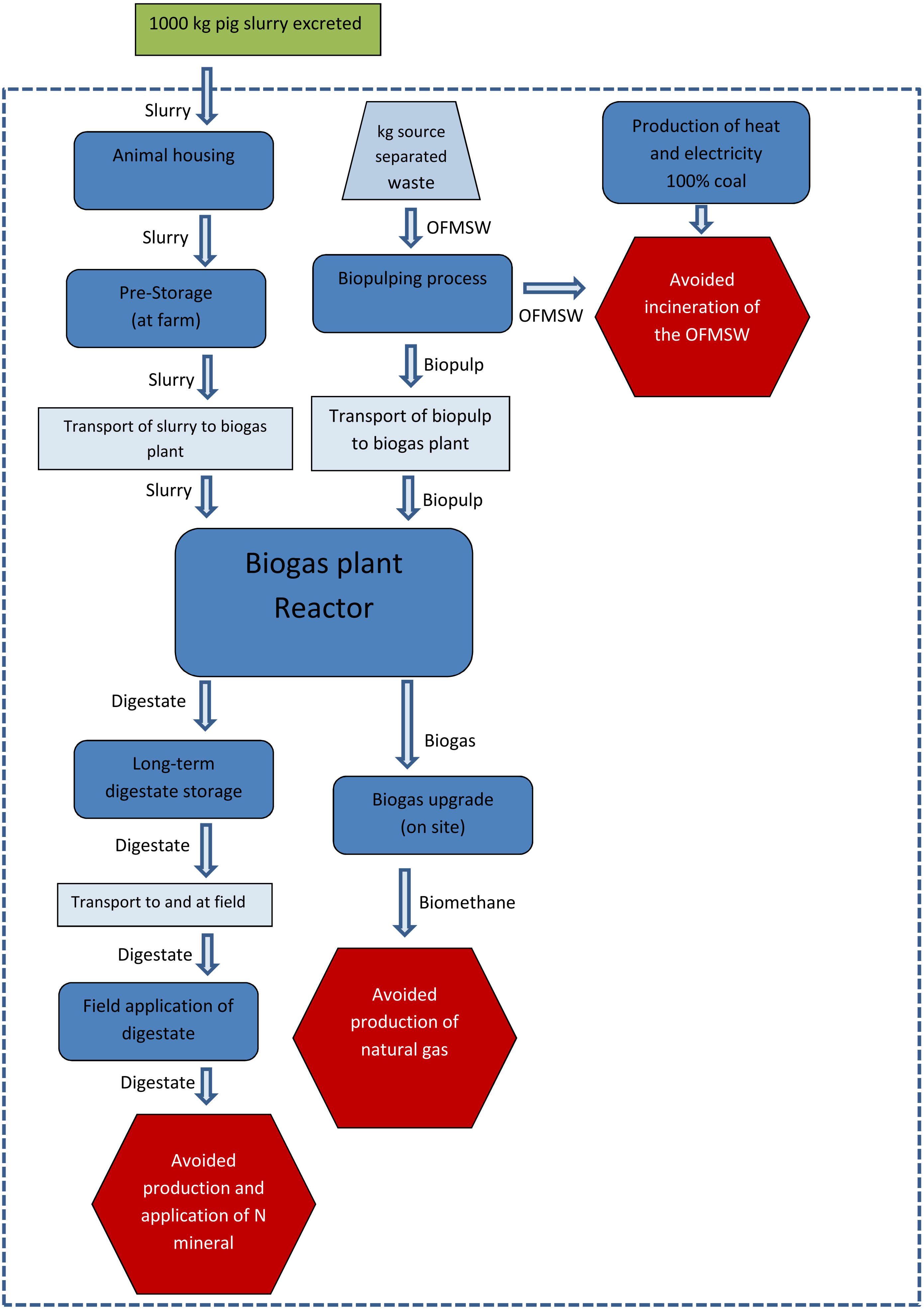
	Slurry ex-animal	Slurry ex-house	Slurry ex-prestorage	Digestate ex-reactor	Digestate ex-storage	Digestate Field
Total mass, kg	1000.00	939.35	875.39	926.94	920.46	
Total Solids, kg	78.70	68.97	67.74	65.35	60.17	
Organic matter, kg	52.58	42.86	41.63	35.75	30.56	
Total nitrogen, kg	6.94	5.61	5.55	5.91	5.85	
TAN, kg	4.96	3.99	3.98	5.04	5.11	
Organic nitrogen, kg	1.98	1.61	1.57	0.87	0.74	
Ash, kg	24.97	24.97	24.97	28.44	28.44	
P, kg	1.15	1.15	1.15	1.16	1.16	
H2O, kg	916.34	866.39	803.67	856.54	855.18	
OM degradation rate, kg kg^{-1}	n/a	0.19	0.03	0.59	0.15	1.08

4.1.3 Co-Digestion with the OFMSW

For the OFMSW scenario the 1000 kg of slurry is co-digested with a product, here referred to as biopulp, which arises from the source-separated organic fraction of municipal solid waste. The Biopulp product is produced by a company called KomTek, which takes the source-sorted OFMSW and runs it through a 'meat hacker-like' device. The end product is a high-quality organic fraction with only around 0.01% reject, in other words plastics and other non-organics (Lorentzen, 2012a, 2012b). Unfortunately, the biopulp modeled in this study does not fully represent the KomTek product. Due to difficulties in determining the biopulp's exact methane yield potential, a value for this was taken from a Danish study that quantified the methane yield potential of source-separated OFMSW. The methane potential of the OFMSW is 340.2 L CH_4 kg $^{-1}$ OM (Davidsson et al., 2007). Thus, it cannot be said with certainty that the results obtained for this scenario represent the biopulp. The major steps for this scenario are as follows after animal housing which is the same as the baseline:

- Pre-storage of slurry: this step is the same for all biogas scenarios, refer to section 4.1.2.
- Biopulping of the OFMSW: it is assumed the organic fraction has already been source-sorted when it enters the biopulping process. The OFMSW goes through this process to produce a high-quality organic fraction.
- Transport of the OFMSW and slurry to the biogas plant: transport of the OFMSW is assumed to cover a wider distance. It is expected that waste coming from the cities will have to travel further to the more remote areas where biogas plants are found. A distance of 40 km was chosen. Transport of the slurry remains as stated in section 4.1.2.

Figure 6 **Process flow diagram of Co-Digestion with OFMSW Scenario. In green, the functional unit, major steps of the life cycle in blue, avoided processes in red**



- System Expansion:
    - Avoided transport of the OFMSW to the incinerating facility: since the OFMSW is taken to a biogas plant instead of to the incinerating facility, there is avoided transport of the OFMSW. The distance to be avoided is assumed to be 20 km. It is expected that the incinerating facility is closer to the city than a biogas plant.
    - Avoided incineration of the OFMSW: as previously stated, because the organic fraction is taken to the biogas plant then incineration of it will not occur and neither will emissions from incineration. Nitrogen emissions are assumed to be in the same rate as emissions from whole municipal solid waste (Møller et al., 2008). Also, it is assumed that there are no changes to the fate of the reject, which continues to be incinerated for the production of heat and electricity.
    - Production of heat and electricity from the OFMSW: the heat and electricity that would have been otherwise produced if the organic fraction was incinerated must still be produced. The electricity and heat are produced from hard coal.
- The steps, ‘biogas plant reactor’, ‘biogas upgrade’, ‘transport of digestate to farm’, ‘long term storage of digestate’, ‘transport to and at field’, and ‘field application of digestate’ are the same for all biogas scenarios, refer to section 4.1.2.

The chemical characterization for the OFMSW scenario is based on KomTek’s chemical analysis of the biopulp.

Table 6 **Chemical characterization, biopulp**

Chemical characterization of biopulp amounts per 1000 kg slurry	
Total mass, kg	381.93
Total Solids, kg	53.47
Organic matter, kg	45.98
Total nitrogen, kg	0.96
TAN, kg	0.18
Organic nitrogen, kg	0.78
Ash, kg	7.36
P, kg	0.13
H2O, kg	328.29

Changes to the reference slurry and co-substrate through the manure management continuum that includes biogas production can be seen in Table 7. From ‘Slurry ex-prestorage’ to ‘Digestate ex-reactor’ the total mass increases, as this is the point where the co-substrate is added. However, unlike the straw scenario the organic matter going into ‘Digestate ex-storage’ is still larger than that of ‘Slurry ex-prestorage’, which points to the fact that the degradation of the OFMSW in the reactor is not as effective as that of extruded straw.

Table 7 **OFMSW scenario. Mass balance through life cycle, amounts per 1000 kg slurry**

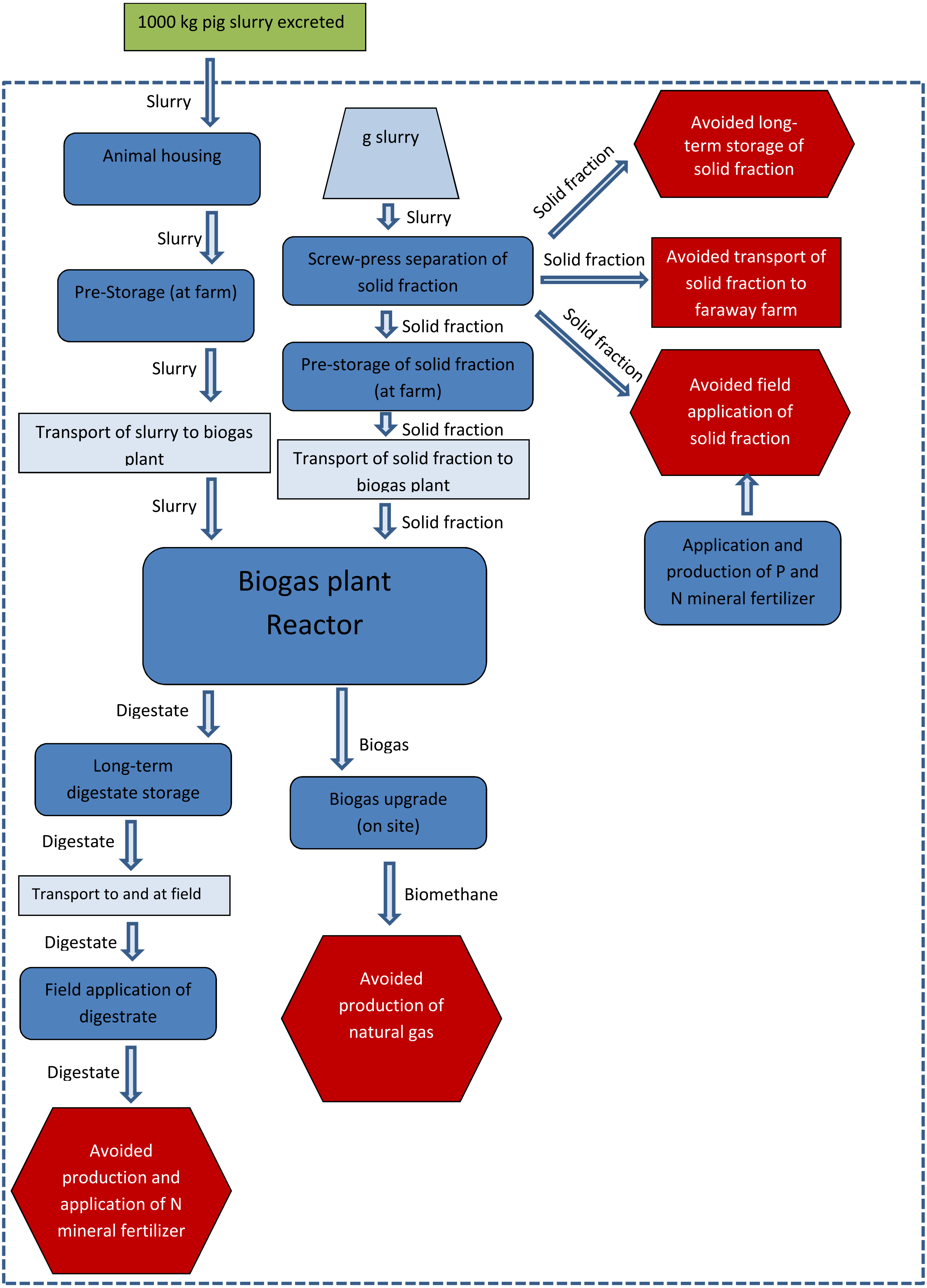
	Slurry ex-animal	Slurry ex-house	Slurry ex-prestorage	Digestate ex-reactor	Digestate ex-storage	Digestate Field
Total mass, kg	1000.00	939.35	875.39	1264.10	1252.54	
Total Solids, kg	78.70	68.97	67.74	78.40	69.58	
Organic matter, kg	52.58	42.86	41.63	44.80	35.97	
Total nitrogen, kg	6.94	5.61	5.55	6.51	6.45	
TAN, kg	4.96	3.99	3.98	5.32	5.49	
Organic nitrogen, kg	1.98	1.61	1.57	1.19	0.95	
Ash, kg	24.97	24.97	24.97	32.33	32.33	
P, kg	1.15	1.15	1.15	1.27	1.27	
H2O, kg	916.34	866.39	803.67	1180.37	1177.47	
OM degradation rate, kg kg^{-1}	n/a	0.19	0.03	0.49	0.20	1.08

4.1.4 Co-Digestion with the solid fraction of slurry

In the separation scenario, other slurry than the functional unit is put through a screw press separator and yields a solid fraction of separated slurry. Since it is assumed the farmer always separates his slurry, the liquid fraction, which will be applied on the field no matter what, is not taken into account. The potential methane yield of the solid fraction is rather low, 170 L CH_4 kg^{-1} OM (Møller et al., 2007). This co-substrate is co-digested with the reference flow in the biogas reactor. As with the other biogas scenarios the animal housing step is the same as for the baseline. Additional steps for this scenario are:

- Pre-storage of slurry: this step is the same for all biogas scenarios, refer to section 4.1.2.
- Screw-press separation of slurry: slurry other than the reference slurry, but with the same chemical characteristics is separated using a screw-press with efficiency to separate nutrients as stated in (Hjorth et al., 2010). A liquid and a solid fraction are the outputs of this process. It is assumed that the farmer always separates his slurry and always applies the liquid fraction to the field. Thus, the effects of applying the liquid fraction are not modeled, since there is no change in the farmer's behavior due to the solid fraction being used for biogas production. Separation happens on-site at the farm.
- Pre-storage of solid fraction of separated slurry: immediately after separation the solid fraction is taken to an outdoor storage unit where it is stored for a 15 day period.
- Transport of the solid fraction and reference slurry to biogas plant: The solid fraction is loaded to a truck and transported to the biogas plant. The distance of transport is equal as that of the reference slurry, since it is assumed that the solid fraction originates at the same farm or a nearby farm, equidistant to the biogas plant. Transport of the slurry remains as stated in section 4.1.2.

Figure 7 **Process flow diagram of Co-Digestion with the solid fraction of separated slurry Scenario. In green, the functional unit, major steps of the life cycle in blue, avoided processes in red**



- System Expansion:
    - Avoided transport of solid fraction to a far-away field: It is assume that the farmer transports the solid fraction to a far-away field in need of phosphorus fertilization. As stated before, the fields where the animals are produced have a surplus of phosphorus built up in the soil (Whalen et al., 2001). Thus, if the farmer sends the solid fraction to the biogas plant, then emissions from this transport, 100km, will be avoided.
    - Avoided long-term storage of solid fraction: since the solid fraction is taken to the biogas plant there is no need to store it until it is allowed to be applied on the field. Long-term storage of the solid fraction is assumed to be 270 days long, or around 9 months.
    - Avoided field application of solid fraction: the solid fraction is not transported to be applied on a field far away. Thus, emissions from application and all related activities such as tillage, loading, and ploughing are avoided.
    - Production and field application of P and N mineral fertilizer: because the solid fraction is not taken to the field far-away, this means that said field will need to synthetically fertilize with N and P mineral fertilizers. The production of these fertilizers is also included in the model calculations. The fertilizers replaced are ammonium nitrate and single superphosphate. P-content in the digestate is assumed to replace P-mineral fertilizer with a 100% efficiency (Thyø and Wenzel, 2007). The solid fraction replaces or in this case does not replace N-mineral fertilizer with a 65% efficiency (FARM-N).
- The steps, ‘biogas plant reactor’, ‘biogas upgrade’, ‘transport of digestate to farm’, ‘long term storage of digestate’, ‘transport to and at field’, and ‘field application of digestate’ are the same for all biogas scenarios, refer to section 4.1.2.

The chemical characterization of the slurry which yields the solid fraction is the same as the reference slurry’s characterization calculated in FARM-N. From this starting point, a new chemical characterization was calculated based on the efficiency of a screw press to transfer different components (OM, TAN, water, etc) to the solid fraction based on (Hjorth et al., 2010).

Table 8 **Chemical characterization, solid fraction**

Chemical characterization of solid fraction amounts per 1000 kg slurry	
Total mass, kg	305.85
Total Solids, kg	53.47
Organic matter, kg	33.22
Total nitrogen, kg	1.73
TAN, kg	0.92
Organic nitrogen, kg	0.81
Ash, kg	19.84
P, kg	0.41
H2O, kg	251.46

Changes to the reference slurry and co-substrate through the manure management continuum that includes biogas production can be seen in Table 9. As in the other biogas scenarios, the point of addition of the co-substrate can be seen from 'Slurry ex-prestorage' to 'Digestate ex-reactor'. In the same manner as for the OFMSW scenario, the OM does not decrease to a level below 'Slurry ex-prestorage'. The degradation rate of this co-substrate is the lowest in the biogas reactor out of all biogas scenarios.

Table 9 Separation scenario. Mass balance through life cycle, amounts per 1000 kg slurry

	Slurry ex-animal	Slurry ex-house	Slurry ex-prestorage	Digestate ex-reactor	Digestate ex-storage	Digestate Field
Total mass, kg	1000.00	939.35	875.39	1203.82	1196.06	
Total Solids, kg	78.70	68.97	67.74	89.46	83.34	
Organic matter, kg	52.58	42.86	41.63	43.10	36.98	
Total nitrogen, kg	6.94	5.61	5.55	7.26	7.19	
TAN, kg	4.96	3.99	3.98	5.94	6.06	
Organic nitrogen, kg	1.98	1.61	1.57	1.32	1.14	
Ash, kg	24.97	24.97	24.97	44.81	44.81	
P, kg	1.15	1.15	1.15	1.56	1.56	
H2O, kg	916.34	866.39	803.67	1108.42	1106.67	
OM degradation rate, kg kg^{-1}	n/a	0.19	0.03	0.39	0.14	1.08

4.2 Assumptions

Below, the main assumptions applying to each of the scenarios are presented. A comprehensive list of assumptions can be found in APPENDIX B. Also, a complete description of how the calculations were made can be found in APPENDIX A.

Table 10 List of main assumptions used to construct the four scenarios of manure management.

Scenario	Process	Value	Unit	Description	Reference
All	Avoided application of N-mineral fertilizer	170	kg N ha^{-1}yr^{-1}	Application limit for manure N application	Hutchings et al., 2012c
	Avoided application of N-mineral fertilizer			Mineral fertilizer replaced is ammonium nitrate, 'DE: Ammonium nitrate (AN, solid)	PE International
	Avoided application of N-mineral fertilizer	0.0265	kg NH$_3$-N kg^{-1}N	Proportion of total nitrogen emitted as NH$_3$-N after application of mineral fertilizer	FARM-N
	Avoided application of N-mineral fertilizer	0.024	kg N$_2$-N kg^{-1} N	Proportion of total nitrogen emitted as N$_2$-N after application of mineral fertilizer	FARM-N
	Avoided application of N-mineral fertilizer	0.0195	kg N$_2$O-N kg^{-1} N	Proportion of total nitrogen emitted as N$_2$O-N after application of mineral fertilizer	FARM-N
	Avoided application of N-mineral fertilizer	0.407	kg NO$_3$-N kg^{-1} N	Proportion of total nitrogen emitted as NO$_3$-N after application of mineral fertilizer	FARM-N
	Avoided application of N-mineral fertilizer	0.59	kg N kg^{-1} N	Proportion of total nitrogen taken up by crop after application of mineral fertilizer	FARM-N
	Field	10	kg C kg^{-1}N	C:N ratio of the soil humus	Petersen et al., 2005
	Field	21.5	kg P ha^{-1}yr^{-1}	Average yearly uptake of phosphorus by crop	Hamelin et al., 2011
	Field	0.05	kg P kg^{-1}	Proportion of P surplus lost to the aquatic	Nielsen and Wenzel,

			surplus P	environment	2007
		0.56	kg C kg^{-1} OM	Carbon content of soil organic matter	Hutchings et al., 2012c
	Field			Fields near the animal production farm are assumed to have P build up in soils	Whalen et al., 2001
	Field			Fields 100 km away are assumed to have low P status	Whalen et al., 2001
	Field			Emissions after field application are included for 10 years, as FARM-N calculates emissions for that period	FARM-N
	Field			Surplus of P is N regulated	FARM-N
	General			NOx emissions are not taken into account in this study, except for incineration of biopulp	Limitation
	General			Potassium content of manure is not tracked through this study	Limitation
	General	0.46	kg C kg^{-1} OM	Carbon content of organic matter in slurry and solid fraction	Hutchings et al., 2012c
	General			All electricity produced, is included in the model with the process 'DK: Electricity from hard coal'	PE International
	Storage, Pre-storage	0.01	kg kg^{-1} TAN-N	Proportion of TAN-N in slurry entering storage that is emitted as NH_3-N	Hansen et al., 2008
Baseline	Field			Slurry is spread by trailing hose	FARM-N
	Field	75	%	Efficiency of slurry to replace mineral N fertilizer	FARM-N
	Field			P-mineral fertilizer is not replaced, as it is assumed there is P build up in soils	Whalen et al., 2001
	Field	0.514	kg kg^{-1} N	Proportion of N in slurry that is taken up by crops (after NH_3 emission)	FARM-N
	Field	0.041	kg kg^{-1} N	Proportion of N in slurry that is emitted as N_2 (after NH_3 emission)	FARM-N
	Field	0.02	kg kg^{-1} N	Proportion of N in slurry that is emitted as N_2O-N (after NH_3 emission)	IPCC, 2006
	Field	0.16	kg kg^{-1} TAN-N	Proportion of TAN-N in slurry applied to field that is emitted as NH_3-N	Hansen et al., 2008
	Field	0.395	kg kg^{-1} N	Proportion of N in slurry that is lost as NO_3^- (after NH_3 emission)	FARM-N
	Storage	0.5	years	Average storage period for manure. Single application period per year	Hutchings et al., 2012c
	Storage	0.185	kg kg^{-1} OM	Degradation rate of OM during storage of slurry in the baseline	Hutchings et al., 2012c
	Storage	0.23	kg kg^{-1} OM deg.	kg CH_4-C emitted per kg OM in slurry decomposed (46% C in OM, 50% emitted as CH_4-C)	Hutchings et al., 2012c
All Biogas	Digestate Storage	10	%	Methane yield potential that remains in the effluent under Danish conditions for centralized biogas plants	Paavola and Rintala, 2008 Angelidaki et al., 2006 Sommer et al., 2000
	Digestate Storage	354	L kg^{-1} OM	Average actual methane yield from literature of which 10% potential is left in digestate	Angelidaki and Ellegaard, 2003; Burton and Turner, 2003; Møller et al., 2004; Jørgensen, 2009
	Digestate storage	38	%	Reduction of methane emission by storage	Sommer et al., 2000

	Pre-storage			cover	
	Digestate storage Pre-storage	0.2	CH_4-C:CH_4-C+CO_2	Ratio of CH_4-C to CH_4-C+CO_2 formed during storage	Sommer et al., 2007
	Field			There is no consensus on the effects of digestion on emissions of ammonia in the field therefore FARM-N estimates were used	Pain et al., 1989; Rubæk et al., 1996; Amon et al., 2006; Sommer, Jensen, Clausen, et al., 2006
	Field			There is no consensus on the effects of digestion on emissions of nitrous oxide in the field therefore FARM-N estimates were used	Petersen et al., 1996; Petersen, 1999; Amon et al., 2006; Clemens et al., 2006; Bhandral et al., 2009; Thomsen et al., 2010; Chadwick et al., 2011; Mikkelsen et al., 2011
	Field			Methane emissions are assumed to be negligible	Sommer et al., 1996; Wulf et al., 2002
	Field	80	%	Efficiency of digestate at replacing mineral nitrogen fertilizer	Chantigny et al., 2007
	Field	0.489	kg kg^{-1} N	Proportion of N in digestate that is taken up by crops (after NH_3 emission)	FARM-N
	Field	0.035	kg kg^{-1} N	Proportion of N in digestate that is emitted as N_2 (after NH_3 emission)	FARM-N
	Field	0.02	kg kg^{-1} N	Proportion of N in digestate that is emitted as N_2O-N (after NH_3 emission)	IPCC, 2006
	Field	0.16	kg kg^{-1} TAN-N	Proportion of TAN-N in digestate applied to field that is emitted as NH_3-N	Hansen et al., 2008
	Field	0.484	kg kg^{-1} N	Proportion of N in digestate that is lost as NO_3^- (after NH_3 emission)	FARM-N
	Reactor	53.47	kg TS	Amount of total solids of co-substrate added to reactor in each digester based on kg straw added to straw scenario	Calculation
	Reactor	15	days	Hydraulic retention time for Danish centralized biogas plant	Hansen et al., 2006
	Reactor	61.75	%	Methane content of biogas produced	Burton and Turner, 2003
	Reactor			The biogas production and degradation rates are calculated separately for the slurry and the co-substrates	Møller, 2012
	Reactor	297.82	L kg^{-1}OM	Methane yield of reference pig slurry	Wang et al., 2009
	Reactor			Heat consumption of reactor calculated by procedure in publication	Hamelin et al., 2010
	Reactor			Methane leaks occur from the reactor tank and from upgrading facilities, CO_2 that might escape at the same time is not taken into account as there is no data	Holmgren et al. 2012
	Reactor	1.6	%	Percent of methane leaking from the reactor	Holmgren et al. 2012
	Upgrade	2.7	%	Percent of methane leaking from upgrading facilities	Holmgren et al. 2012
	Pre-storage	0.011	g C h^{-1} kg^{-1} OM	Hourly CH_4-C emission during storage	Sommer et al., 2007
	Pre-storage	10	days	Time duration of pre-storage	Møller, 2012
Straw	Avoided straw left on field			Modeled with Daisy for 100 year period	Abrahamsen and Hansen, 2000

	Avoided straw left on field	0.028	kg N_2-N kg $^{-1}$ N	Proportion of total nitrogen emitted as N_2-N	Abrahamsen and Hansen, 2000
	Avoided straw left on field	0.025	kg N_2O-N kg^{-1} N	Proportion of total nitrogen emitted as N_2O-N	Abrahamsen and Hansen, 2000
	Avoided straw left on field	0.417	kg NO_3-N kg^{-1} N	Proportion of total nitrogen emitted as NO_3-N	Abrahamsen and Hansen, 2000
	Avoided straw left on field	0.529	kg N kg^{-1} N	Proportion of total nitrogen taken up by crop	Abrahamsen and Hansen, 2000
	Avoided straw left on field	0.973	kg CO_2-C kg^{-1} C	Proportion of total carbon emitted as CO_2-N	Abrahamsen and Hansen, 2000
	Co-substrate			Chemical characterization of wheat straw	Wang et al., 2009
	Co-substrate	6.79 x10^{-3}	kg N kg^{-1} TS	Total nitrogen per kg of wheat straw	Wang et al., 2009
	Co-substrate	8.2 x10^{-4}	kg TAN-N kg^{-1} TS	Total ammoniacal nitrogen per kg of wheat straw	Wang et al., 2009
	Co-substrate	2.97x10^{-4}	kg P kg^{-1} straw	Phosphorus per kg of wheat straw	Ontario Ministry of Agriculture, 2012
	Co-substrate	279.2	L CH_4 kg^{-1} OM	Methane yield for wheat straw with 15 day HRT	Wang et al., 2009
	Digestate storage	220	L CH_4 kg^{-1} OM	Average actual methane yield of wheat straw from literature of which 10% potential is left in digestate	Hashimoto, 1983; Burton and Turner, 2003; Jørgensen, 2009
	Pre-treatment	70	%	Increase in potential methane yield for barley straw, it is assumed to be the same for wheat straw	Hjorth et al., 2011
	Reactor	5	% per w/w slurry	Mass of extruded straw added to reactor	Møller, 2012
OFMSW	Avoided incineration biopulp			Emissions from biopulp are assumed to be in the same rate as for household waste	Møller et al., 2008
	Avoided incineration biopulp	0.86	kg NO_x t^{-1} waste	Avoided NO_x emission from biopulp	Møller et al., 2008
	Avoided incineration biopulp			All carbon in biopulp is released as CO_2 during incineration (avoided)	Own assumption
	Co-substrate			Chemical characterization of organic fraction of municipal solid waste (biopulp) is from KomTek's chemical analysis	Lorentzen, 2012
	Co-substrate	2.52	g N kg^{-1} w/w	Total nitrogen in biopulp per kg wet waste	Lorentzen, 2012
	Co-substrate	0.465	g TAN-N kg^{-1} w/w	Total ammoniacal nitrogen in biopulp per kg wet waste	Lorentzen, 2012
	Co-substrate	0.333	g P kg^{-1} w/w	Phosphorus per kg wet waste	Lorentzen, 2012
	Co-substrate	340.2	L CH_4 kg^{-1} OM	Methane yield for organic fraction of municipal solid waste	Davidsson et al., 2007
	Digestate storage	466	L CH_4 kg^{-1} OM	Average actual methane yield of the OFMSW from literature of which 10% potential is left in digestate	Hashimoto, 1983; Torres-Castillo et al., 1995; Davidsson et al., 2007; Jørgensen, 2009
	Pre-treatment	25.5	kWh t^{-1} waste	Electricity consumption of the biopulping process	Lorentzen, 2012
	Production electricity due to avoided incineration	0.1	MJ kg^{-1}	Electricity to be produced in a conventional way per kg biopulp	Møller et al., 2008
	Production electricity due to avoided incineration	1.29	MJ kg^{-1}	Heat to be produced in a conventional way per kg biopulp	Møller et al., 2008

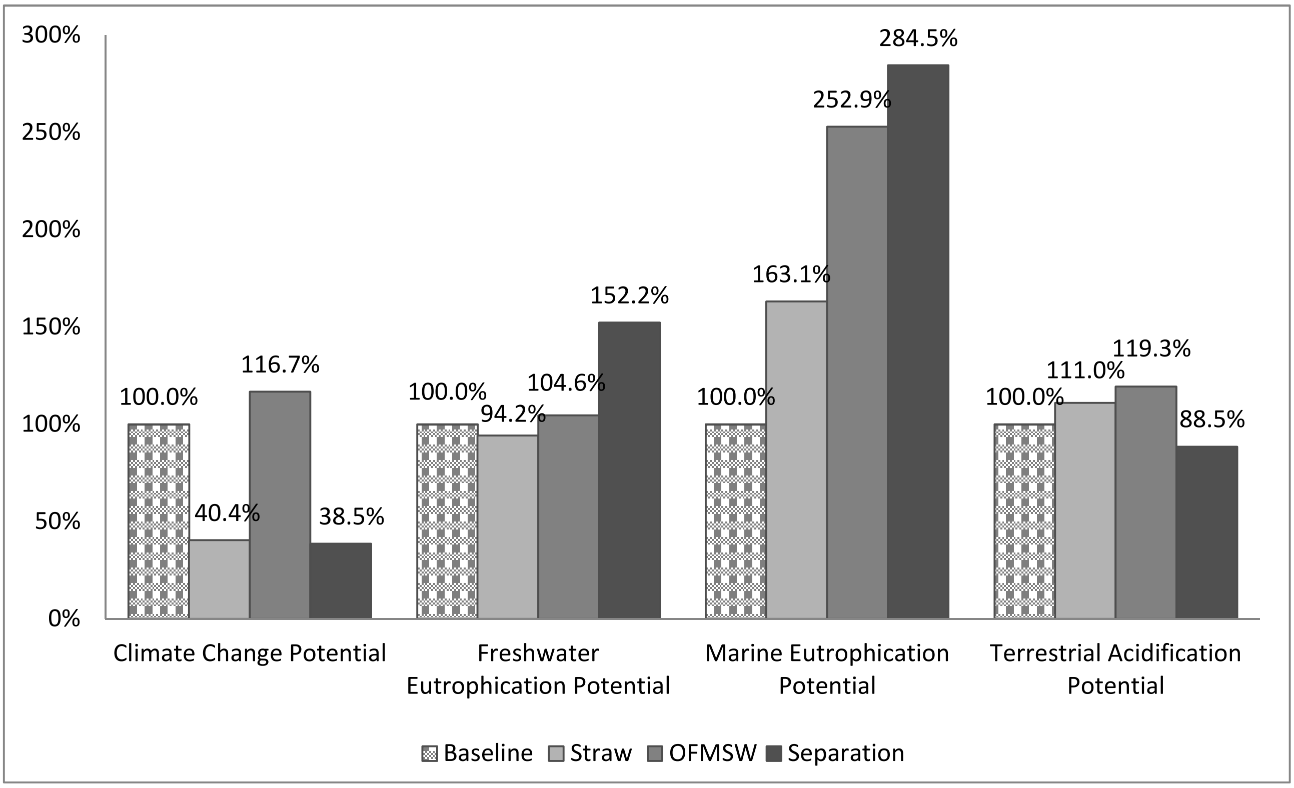
Separation	Avoided application solid fraction on field			It is assumed that a field far away in need of phosphorus does not receive the solid fraction for fertilization	Own assumption Whalen et al., 2001
	Avoided application solid fraction on field	0.39	kg NH_3-N kg^{-1} TAN	Proportion of total ammoniacal nitrogen emitted as NH_3-N after application of solid fraction	Hansen et al., 2008
	Avoided application solid fraction on field	0.038	kg N_2-N kg $^{-1}$ N	Proportion of total nitrogen emitted as N_2-N after application of solid fraction	FARM-N
	Avoided application solid fraction on field	0.02	kg N_2O-N kg^{-1} N	Proportion of total nitrogen emitted as N_2O-N after application of solid fraction	IPCC, 2006
	Avoided application solid fraction on field	0.332	kg NO_3-N kg^{-1} N	Proportion of total nitrogen emitted as NO_3-N after application of solid fraction	FARM-N
	Avoided application solid fraction on field	0.435	kg N kg^{-1} N	Proportion of total nitrogen taken up by crop after application of solid fraction	FARM-N
	Avoided application solid fraction on field			Phosphorus mineral fertilizer replaced is single superphosphate	Thyø and Wenzel, 2007
	Avoided application solid fraction on field	100	%	P fertilizer replacement efficiency in all scenarios	Thyø and Wenzel, 2007
	Avoided application solid fraction on field			When both N and P mineral fertilizer are applied, they are mixed, so only 1 time spreading	Own assumption
	Avoided application solid fraction on field	65	%	N fertilizer replacement efficiency in solid fraction after screw press separation compared to mineral N fertilizer	FARM-N
	Avoided application solid fraction on field	6	hours	Time between spreading and ploughing for the solid fraction	Hansen et al., 2008
	Avoided long term storage of solid fraction			It is assumed that the degradation of solid fraction is equal to slurry due to a limitation	Own assumption, limitation
	Avoided long term storage of solid fraction			Avoided long term storage of solid fraction is calculated in the same way as pre-storage of slurry	Own assumption
	Co-substrate	0.0323	kg N kg^{-1} TS	Total nitrogen per kg of solid fraction TS	FARM-N
	Co-substrate	0.0172	kg TAN-N kg^{-1} TS	Total ammoniacal nitrogen per kg of solid fraction TS	FARM-N
	Co-substrate	0.0076	kg P kg^{-1} TS	Phosphorus per kg of solid fraction TS	FARM-N
	Co-substrate	170	L CH_4 kg^{-1} OM	Methane yield of the solid fraction	Hamelin et al., 2010
	Digestate storage	186.25	L CH_4 kg^{-1} OM	Average actual methane yield of the solid fraction from literature of which 10% potential is left in digestate	Andara and Esteban, 1999; Møller et al., 2004, 2007; Luostarinen et al., 2011
	Pre-storage solid fraction			Calculated in the same way as pre-storage of slurry	Own assumption
	Pre-treatment	0.9	kWh t^{-1} slurry	The energy for separation with a screw press (slurry is 2 weeks old)	Møller et al., 2002
	Pre-treatment	0.24	kg kg^{-1} slurry	Separation efficiency for Organic N (share of ash in solid fraction)	Hjorth et al., 2010
	Pre-treatment	0.17	kg kg^{-1} slurry	Separation efficiency for P (share of ash in solid fraction)	Hjorth et al., 2010
	Pre-treatment	0.11	kg kg^{-1} slurry	Separation efficiency for TAN (share of ash in solid fraction)	Hjorth et al., 2010
	Production and application of P-mineral fertilizer	0.05	kg P kg^{-1} surplus P	Proportion of P surplus lost to the aquatic environment	Nielsen and Wenzel, 2007

5 Results and Discussion

5.1 General Overview

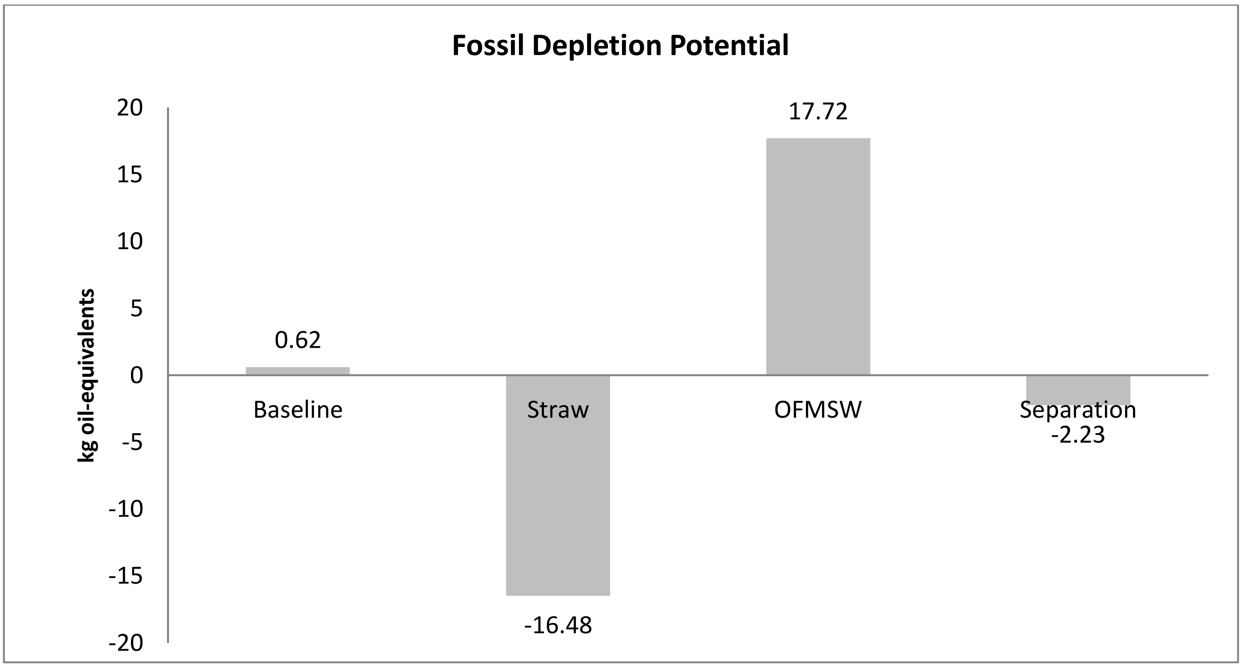
The three biogas scenarios showed varying potentials to impact the environment. Their performance against the baseline scenario, in the five impact categories assessed for this study, is discussed with the figures below.

Figure 8 **Relative impact of scenarios in four impact categories. Baseline scenario is shown as 100%.**



As can be seen in Figure 8 the baseline scenario shows considerably lower environmental impacts for the marine water eutrophication impact category, while the results for the remaining impact categories are mixed (Note: Fossil Depletion category is shown in separate graph). With regards to climate change potential two scenarios, straw and separation, have a clear advantage in comparison to the baseline scenario, potentially contributing 59.6% and 61.5% less to climate change than the baseline respectively. On the contrary, producing biogas from the organic fraction of municipal solid waste results in a higher contribution, by 16.7%, to climate change than if the slurry is treated in the conventional way. The OFMSW scenario consistently performs worse than the baseline for all impact categories, including fossil fuel depletion shown in Figure 9. For terrestrial acidification potential, the separation scenario is the only one that fares better than the baseline. Both straw and OFMSW scenarios contribute more to this impact category, with an added 11% and 19.3% respectively. With regards to the freshwater eutrophication category, only the straw scenario produces savings in comparison to the baseline, while both OFMSW and the separation scenario cause burdens of 4.6% and 52.2% larger than the baseline, respectively.

Figure 9 **Fossil depletion potential of all scenarios in kilogram of oil equivalents.**



As the three scenarios modeled against the baseline produce biogas, which is then upgraded and injected into the natural gas grid, it is no surprise that they all show considerably different results in comparison to the baseline, under the fossil fuel depletion category. Straw in particular shows the most improvement, producing a saving of 15.86 kg oil-equivalents in comparison to the baseline, which in turn represents 0.62 kg oil-equivalents depletion. Following straw, the separation scenario also represents a saving, though not as significant as for the straw scenario, saving 1.61 kg oil equivalents in comparison to the baseline. Lastly, the OFMSW scenario produces the largest fossil fuel depletion, 17.1 kg oil-equivalents more than the baseline, regardless of the production of biogas. This surprising result is mostly due to the system expansion needed to balance this scenario, i.e. what would otherwise be done with the organic fraction of municipal solid waste if it was not taken to the biogas plant, discussed further in section 5.2 and 5.2.1.

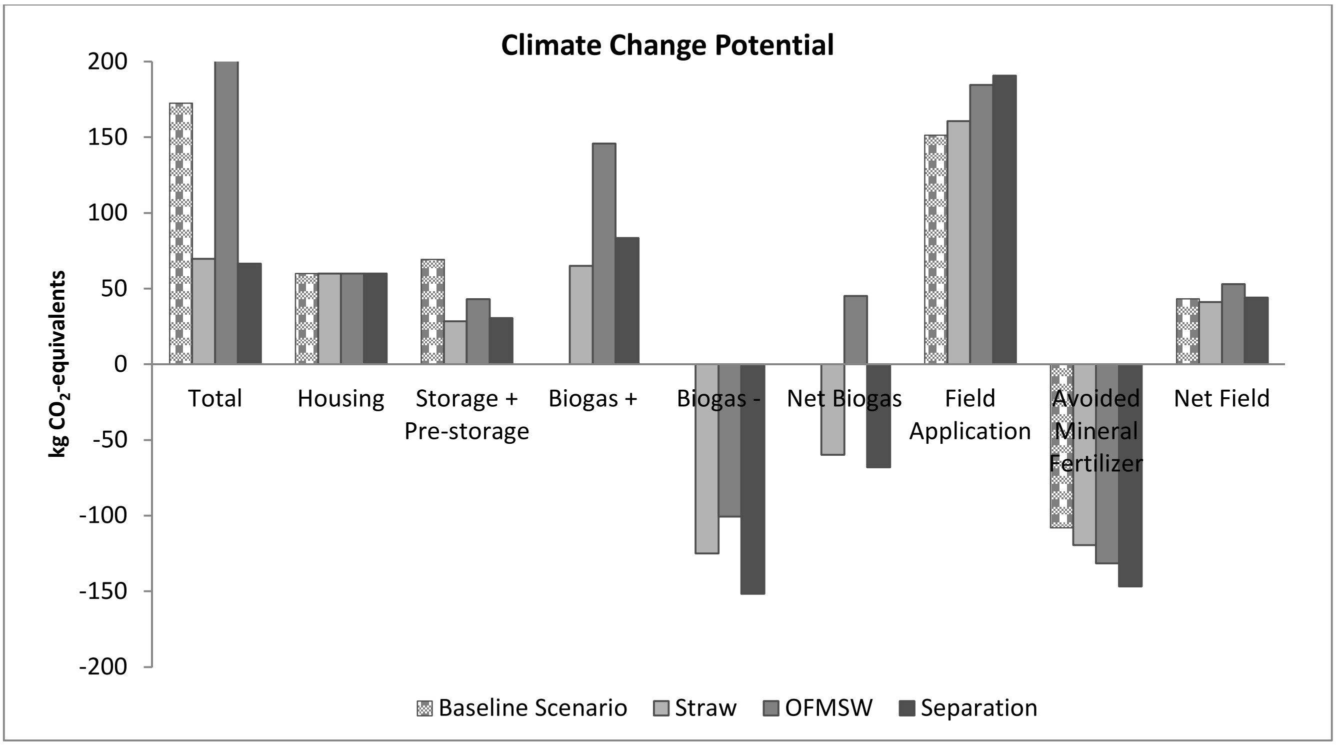
Below the findings of this study for the five impact categories are discussed in more detail. In addition, a small section describes uncertainties that apply to each impact category. These are areas that have the potential to change study results or are merely areas of the modeling that would benefit from improvement.

5.2 Climate Change Potential

The total contribution of the baseline, straw, OFMSW, and separation scenarios to climate change potential in kilograms of CO_2 equivalents is 172.5, 69.7, 201.3, and 66.5 respectively. Figure 10 also shows the individual contributions of the major steps in the four manure management scenarios. Here, system expansion activities, referred to as ‘co-substrate specific activities’, are grouped in the Biogas columns.

The largest contributions to climate change potential happen as a result of emissions after field application. This is reasonable since almost all carbon in organic matter added to the field will be released as a CO_2 emission in the 10 year period modeled. The contribution from this emission varies according to the amount of organic matter added to the field and the degradation rate of the organic matter in the field (see mass balances chapter 4). Thus, the straw scenario, which has the least organic matter added to the field of the biogas scenarios, contributes the least in field emissions. The separation scenario, which has the most organic matter added to the field of the biogas scenarios, has the largest contribution in field emissions. Also, the biogas scenarios have a higher degradation rate in the field, which means that they release more of the carbon locked in the organic matter added. This partly explains why they fare worse against the baseline scenario. The higher degradation rate for digestate is a FARM-N calculation, explained in APPENDIX A section 8.1.2. The baseline scenario, which has the lowest degradation rate in the field out of all scenarios, adds a larger amount of organic matter to the field than the straw scenario, but has a lower contribution from field emissions, due to the lower degradation rate.

Figure 10 **Contribution of all scenarios to climate change potential throughout each step of the manure management continuum. Biogas + are burdens from biogas production, Biogas – are savings from biogas production, Net biogas is the net contribution from biogas production.**



In a similar manner, the amount of organic matter added to the field, which is proportional to the total nitrogen added to the field, accounts for the differences in the avoided emissions from mineral fertilizer. As the separation scenario has the highest total nitrogen content, it follows that it provides the highest savings from avoided mineral fertilizer application. The baseline scenario provides the least savings in avoided mineral fertilizer, as it adds the lowest amount of total nitrogen to the field out of all scenarios. The Net impact arising from field application of the different biomasses can be seen in the last column of Figure 10. Straw has the lowest net field impact out of all scenarios and separation is only slightly higher than the baseline, while OFMSW is consistently worse.

Both CH_4 emissions and CO_2 make large contributions to the climate change potential during the housing phase of manure management. This is perhaps the largest climate change contribution if one compares the net field emissions, instead of the field application emissions alone. However, as the housing emissions are the same for all scenarios it does not affect the results of the comparison. On the other hand, the storage phase signifies the largest net contribution to climate change for the baseline scenario, unlike the biogas scenarios, which have a modest contribution coming from storage. The results shown in Figure 10 for the storage of the biogas scenarios correlate well with the amount of organic matter going into the storage and the co-substrates' specific degradation rates, which arise from each of the co-substrate's methane yield potentials. For example, OFMSW has the highest input amount of organic matter to the storage out of all biogas scenarios and has the highest methane yield potential, leading to the highest degradation rate during storage. As more organic matter is available for degradation in OFMSW storage, this results in the highest methane and carbon dioxide emissions during storage out of the biogas scenarios. A point of interest is the fact that two of the scenarios, separation and OFMSW, have a higher input of organic matter into the storage and very similar degradation rates to the baseline. This should have resulted in higher impacts from these two scenarios' storage phase, but in fact they show a lower contribution from storage emissions than the baseline. It has been determined that this is due to the differences in calculating baseline storage and digestate storage (see APPENDIX A). Among others, possibly because the effects of covering the storage tank are not included in the baseline storage calculations. Therein the storage has been included in the sensitivity analysis and is discussed further in said section.

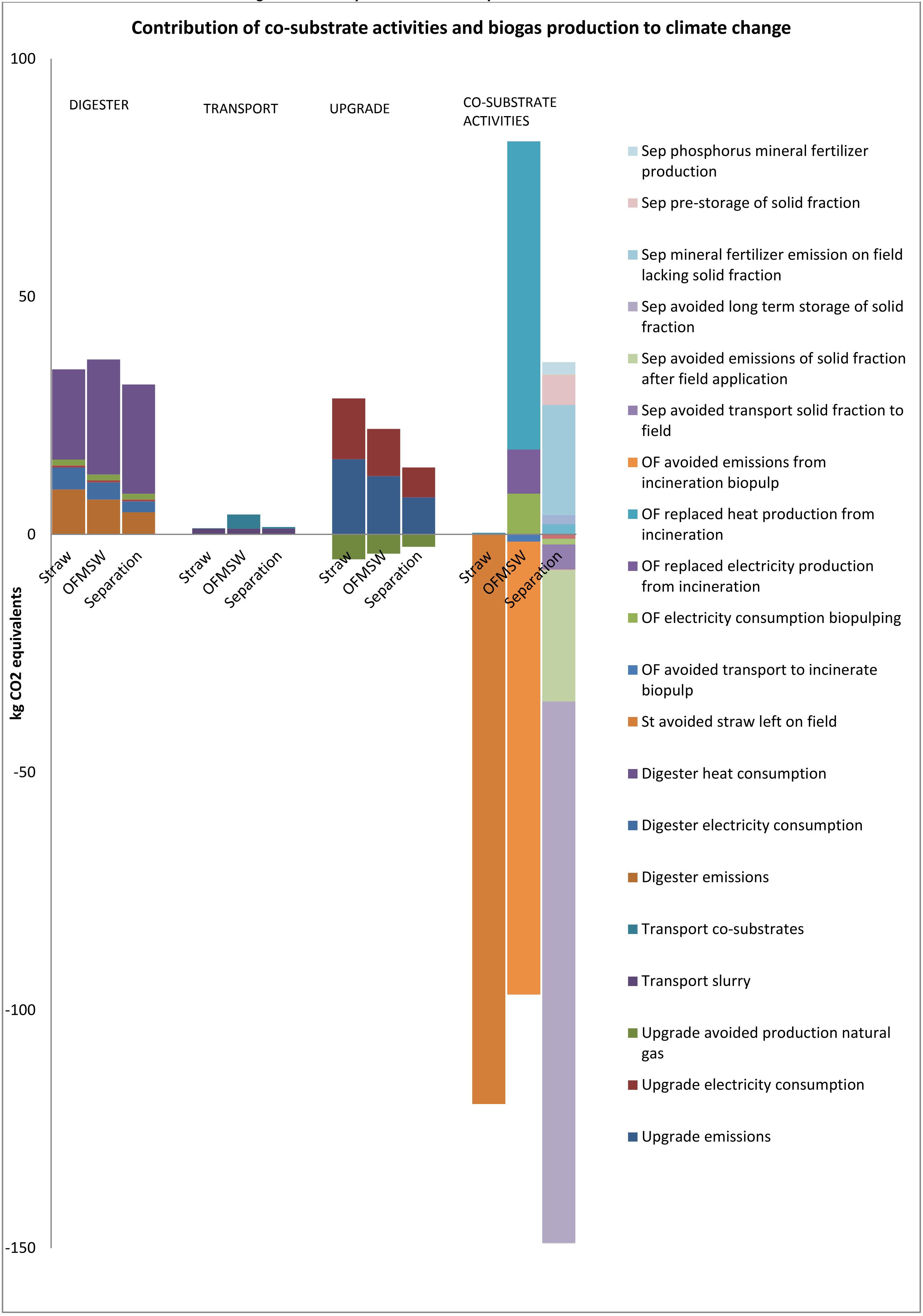
Activities grouped under biogas in Figure 10 include, biogas production activities, such as running a biogas reactor or upgrading the biogas to natural gas quality, and also all activities that arise from the choice to use the co-substrates for biogas production instead of for their conventional use, e.g. avoided emissions from straw that would have been left on the field if it had not been taken to a biogas plant. For the straw and separation scenarios said activities create savings of 59.87 and 68.09 kg-CO_2 equivalents correspondingly; one of the main reasons why these two scenarios are an improvement for climate change in comparison to the baseline scenario. It is also the reason why the separation scenario is the one causing the lowest impact to climate change. A similar result was expected for the OFMSW scenario, but this is not the case. The explanation for this is clearly shown in Figure 11, which shows each biogas related activity for the three biogas scenarios.

The OFMSW differs from the straw and separation scenario in that if the organic fraction had not been taken to the biogas plant, it would have been used to produce energy, but from incineration. As can be seen in Figure 11, GHG emissions are avoided by not incinerating the organic fraction, but the model must still produce the electricity and heat that one would normally produce from this co-substrate. It is the production of said electricity and heat that gives rise to high emissions for the OFMSW scenario (labeled 'OF replaced heat production from incineration' and 'OF replaced electricity production from incineration' in the legend). In addition, the OFMSW scenario has additional consumption of electricity by the biopulping process, higher emissions from the digester's heat consumption because of the waste's high water content and higher emissions from transporting such a liquid prominent co-substrate. All of these factors combined result in that the OFMSW scenario contributes 45.13 kg CO_2-equivalents more to climate change than the baseline scenario from net biogas contribution.

Unlike the OFMSW scenario, the system expansion choices result in savings for the separation scenario and the straw scenario. For the latter scenario these savings come from not leaving straw on the field to degrade. One has to ask the question if this would have been different if the alternative use of the straw had instead been to produce energy with it from incineration. In that case, it is likely that the results would have been different. However, this falls outside of the scope of this study. For the separation scenario, the choices made for system expansion result in both added emissions and ultimately in net savings arising from co-substrate specific activities. Avoided transport, long-term storage and emissions from field application of the solid fraction cause the greatest savings to climate change for the separation scenario. On the other hand, there is a significant contribution from field emissions after application of P-mineral fertilizer to the field that will not receive the solid fraction for fertilization. In this way modeling choices become extremely important for the results of the scenarios.

Figure 11 is also instrumental in showing that the model results agree well with inventory data. The co-substrates' methane yield potential are from highest to lowest for extruded straw, OFMSW and the solid fraction of manure. Under upgrade it is possible to see that the straw scenario replaces the most natural gas, as it produces the most biogas, OFMSW replaces the second highest and separation replaces the least, reflecting their methane yield potentials. Straw also has the highest CH_4 emissions in the digester, the highest electricity consumption in the digester (kWh/L biogas produced), the highest emissions from upgrading to natural gas (% of CH_4 produced) and again the highest electricity consumption for upgrading. These are all a result of the straw scenario producing the most biogas. The other scenarios follow this pattern; the OFMSW having the second highest emissions and electricity consumption and separation the least, all according to the methane yield potential. It is then evident that methane yields are a very important parameter for the model and can influence the results of the model, which is why this parameter is also included in the sensitivity analysis.

Figure 11 **Contribution to climate change potential of biogas production processes; Digester, Transport, Upgrade,as well as activities arising from the use of each co-substrate. Reduced legend shows mayor contributions only.**



In summary, the straw and separation scenario produce the most savings in terms of climate change potential, while the OFMSW scenario contributes more to climate change than the baseline. By looking at the different stages of the life cycle it is possible to see that the highest emissions occur during field application, but the net field impact on climate change is fairly similar between all scenarios. The housing step is exactly the same for all scenarios. In contrast, significant savings occur during the long-term storage and pre-storage of all biogas scenarios in comparison to the baseline. Even larger savings occur from the net impact of biogas production and co-substrate related activities for two of the biogas scenarios. The straw and separation scenarios are clearly a more environmentally friendly choice in terms of climate change potential than both the baseline and OFMSW. The OFMSW scenario contributes more to climate change than the baseline and this is a result of the heat and electricity that must be produced to replace the heat and electricity that would have otherwise been produced from incineration of the OFMSW, had it not been used for biogas.

5.2.1 Climate Change Uncertainties

It is important to stress that the choices made for system expansion are capable of changing the results of the study. Through system expansion the question of what is the environmental consequence of taking these co-substrates for biogas production is explored and an uncertainty enters the model (Baumann and Tillman, 2009). In this study, it was assumed that the organic fraction of municipal solid waste used for biogas production would have been used for incineration if it had not been taken to the biogas plant. As a result, it was necessary to model the avoided emissions from incinerating the organic fraction which resulted in climate change savings, but also the production of heat and electricity that the organic fraction would have produced if incinerated, which resulted in a climate change burden. If the assumption had been that the organic fraction of municipal solid waste used for biogas production would have been used for something else, such as composting, if it had not been taken to the biogas plant, we can expect the results would be different. The same principle applies to the straw and separation scenarios. It was assumed that if the straw was not used for biogas production it would then be left in the field to degrade and it was assumed that if the farmer does not sell the solid fraction to a biogas plant he would then store it and eventually apply it to a field in need of phosphorus. If instead it had been assumed that the straw would be incinerated for energy generation or that the solid fraction would be made into pellets and then combusted, this would yield different results. The choices taken are assumed to mimic what would happen in reality as much as possible. It appears logical to expect that the organic fraction would have otherwise been incinerated since incineration is the number one method of waste disposal in Denmark (Miljøministeriet og Miljøstyrelsen, 2011). Likewise, it is logical to expect that straw used for biogas would otherwise have been left in the field as there are 2.1 million tons of surplus straw annually in Denmark (Skøtt, 2011). With this amount of surplus straw, it is not logical to expect that it would be preferable for the biogas straw to compete with other established uses, such as forage, bedding or incineration. It is also logical to expect that the farmer will apply the solid fraction to his field, for fertilization, if he does not sell it to the biogas plant, as field application is the most conventional use for animal manure in Denmark (Hjorth et al., 2010). Thus, the choices made seem reasonable, nevertheless the fact that different choices for system expansion would lead to different results is very important. In order to be more confident of the results, alternative uses of these co-substrates should be thoroughly explored. Unfortunately, time limitations do not allow for such an exploration here.

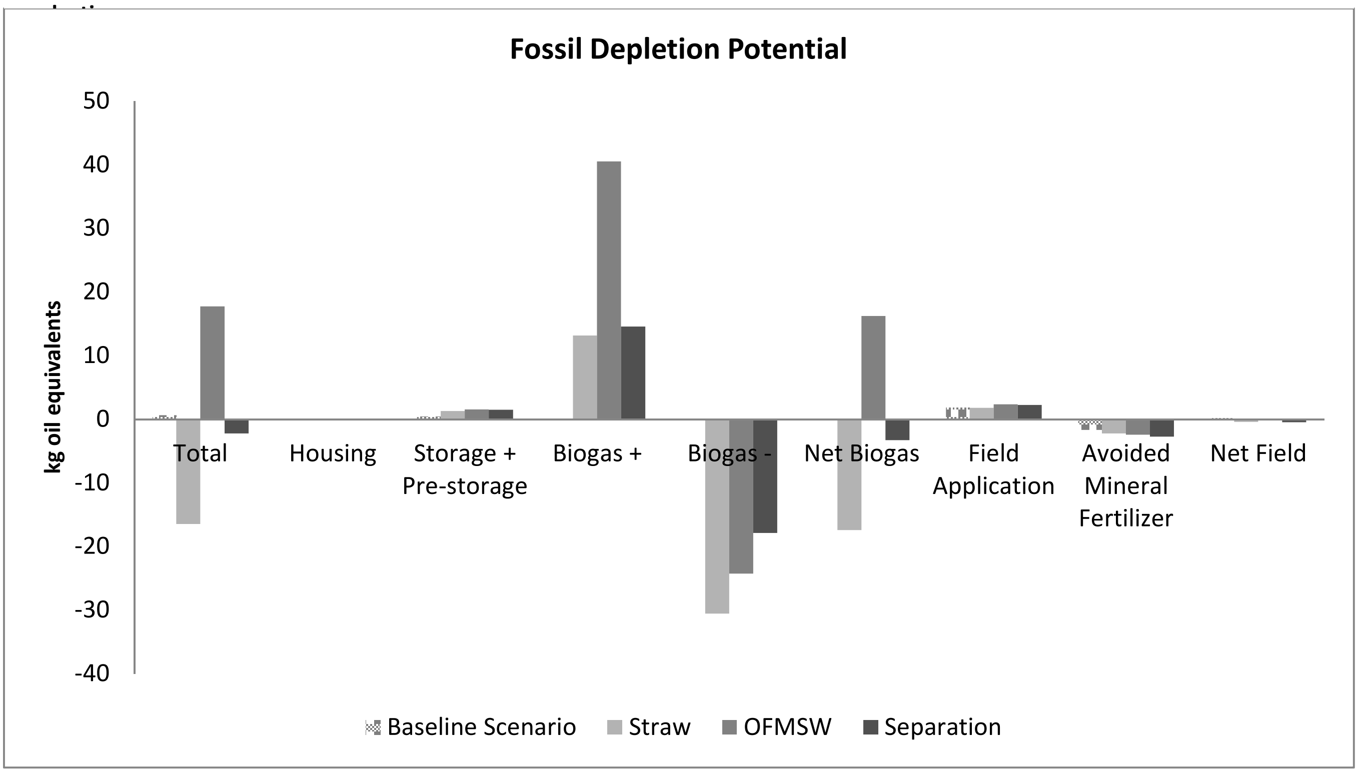
The model results show to be largely influenced by the co-substrates' potential methane yield, for which a wide range is found in literature. The methane yields of the co-substrates were chosen where possible, to represent a 15 day hydraulic retention time, the most common retention time used in Danish biogas plants (Hansen et al., 2006). However, since this parameter is highly influential of the scenarios' results in all impact categories, it is thereby included in the sensitivity analysis for this project (see section 5.7).

A last area of uncertainty was identified for climate change potential of the straw scenario. It arises from a choice to model the straw left on the field for 100 year period, instead of a 10 year period as all other field processes are modeled. However, after a brief investigation it was observed that the degradation rate of the straw left on the field is circa 97% for the 100 year period. In comparison to the rate of degradation of undigested slurry in the field, this value is slightly higher by 5% and in comparison to the rate of degradation of the digestate in the field this value is lower by 16% for a 10 year period. As the value for the degradation of the straw is similar to the others, this seems to indicate that the initial period is the most important for degradation of organic matter. Thus, the differences in time period are not deemed to significantly change the results.

5.3 Fossil Depletion Potential

The mechanisms creating savings/burdens on fossil depletion potential are very similar to those observed for climate change potential. However, it is evident from Figure 12 that the largest deviation from the baseline happens during the biogas phase for the biogas scenarios.

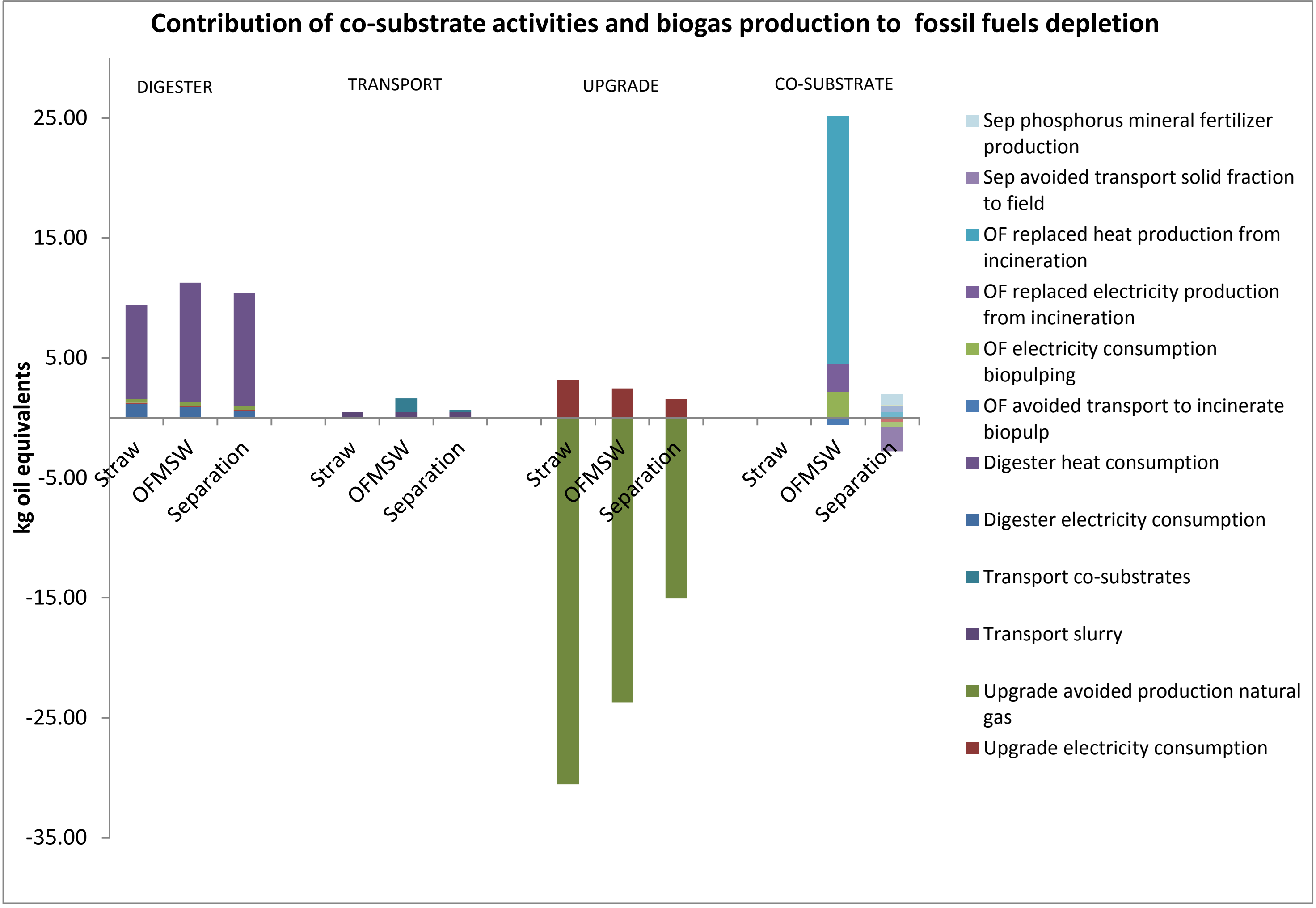
Figure 12 **Contribution of all scenarios to fossil depletion throughout each step of the manure management continuum. Biogas + are burdens from biogas production, Biogas – are savings from biogas production, Net biogas is the net contribution from biogas**



As has been stated before, the straw scenario yields the highest amount of biogas and replaces natural gas production to the largest extent. Avoided natural gas production saves 30.55, 23.71 and 15.05 kg oil-equivalents for the straw, OFMSW, and separation scenarios respectively. Again, these results correlate well with the co-substrates' methane yield potential. For the OFMSW scenario, the production of heat that will not be produced by incinerating the organic fraction negates most of the benefit gained by the avoided natural gas production. This, in combination with the other activities that use more energy for the OFMSW scenario, such as electricity needed for biopulping, extra heat needed to warm the biogas reactor and transport of such a heavy liquid load, result in a net burden for the fossil depletion category. A net consumption of 0.62 kg oil-equivalents is observed for the baseline scenario. In comparison, both the straw and separation scenarios represent a net savings of fossil depletion, though the straw's saving is much higher; 16.48 compared to 2.23 kg oil-equivalents. Here, the main reason for the straw scenario's advantage over separation is that the separation scenario replaces a much lower amount of natural gas, about half of what the straw scenario replaces. This fact can clearly be seen in Figure 13 and it can also be seen that fossil depletion during the digester phase is higher for the separation scenario. The straw scenario does not have any significant contributions to fossil depletion from co-substrate specific activities.

To summarize, the straw scenario produces the most savings of all other scenarios in the fossil depletion category. This is due to the high methane yield of extruded straw. The separation scenario produces modest savings in comparison to the baseline and this is due to the solid fraction's low potential methane yield and various activities that deplete fossil fuels. The OFMSW scenario produces a high burden explained by the need to produce heat and electricity in a conventional way that would otherwise have been produced by incineration of the organic fraction.

Figure 13 **Contribution to fossil depletion potential of biogas production activities; Upgrade, Transport, Digester, as well as activities arising from the use of each co-substrate. Reduced legend shows most important contributions**



5.3.1 Fossil Depletion Uncertainties

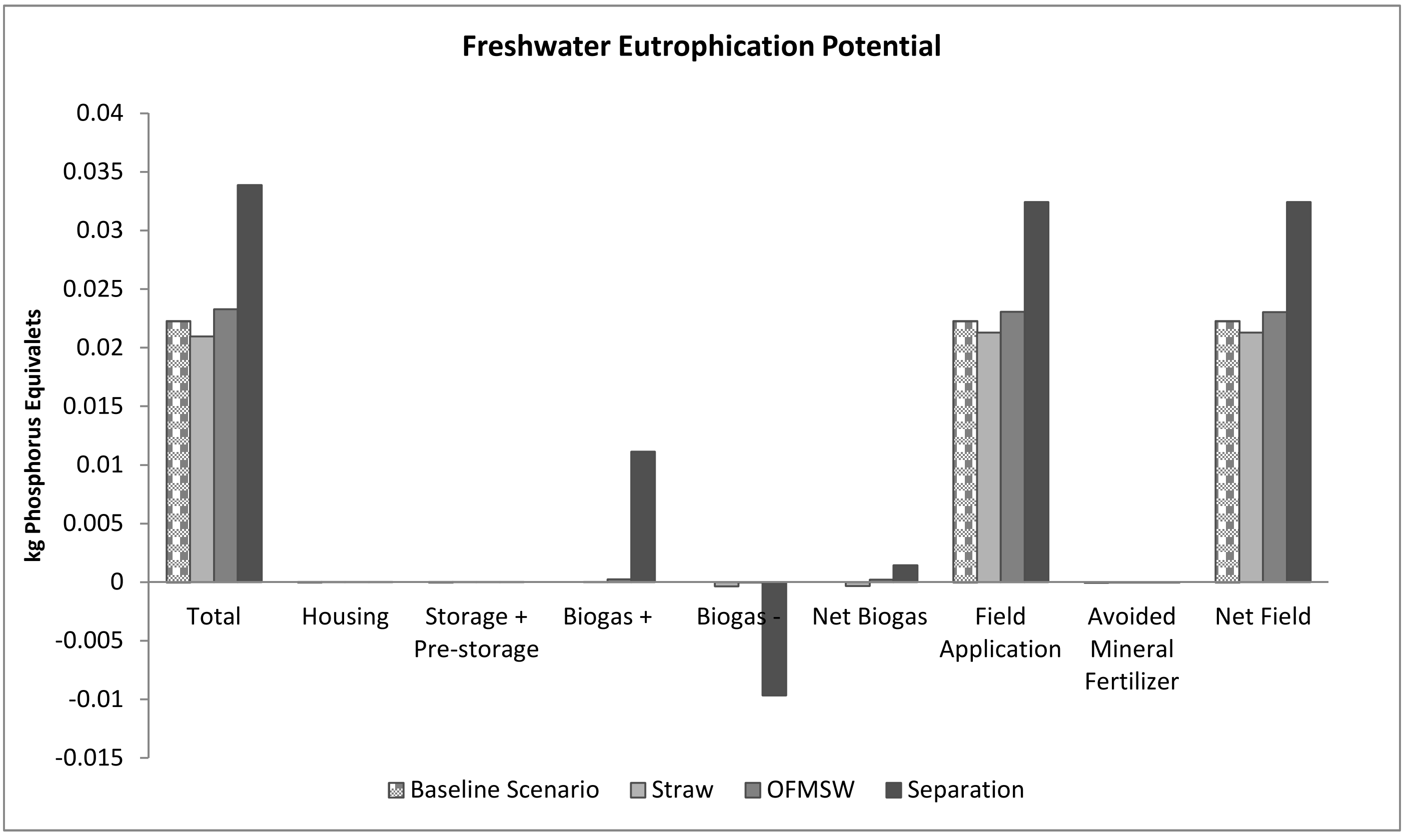
As mentioned before, under section 5.2.1, system expansion introduces uncertainties which are also valid for fossil depletion. If the choices for avoided activities for any of the three biogas scenario had been different, the results for fossil depletion would also change. However, the choices taken here are considered reasonable for the scope of this study.

In the same way as for climate change, the potential methane yield of the co-substrates is important, as here it was observed that the straw scenario had the most savings because of its high methane yield potential. Changes to this parameter are thus investigated thoroughly in the sensitivity analysis.

5.4 Fresh Water Eutrophication Potential

By far, the biggest environmental burden for this impact category comes from emissions of phosphorus leaching in the field, as can be seen in Figure 14. Additionally, the separation scenario shows a somewhat significant burden from co-substrate specific activities in the Net Biogas column i.e. choices for system expansion, shown in detail in Figure 15.

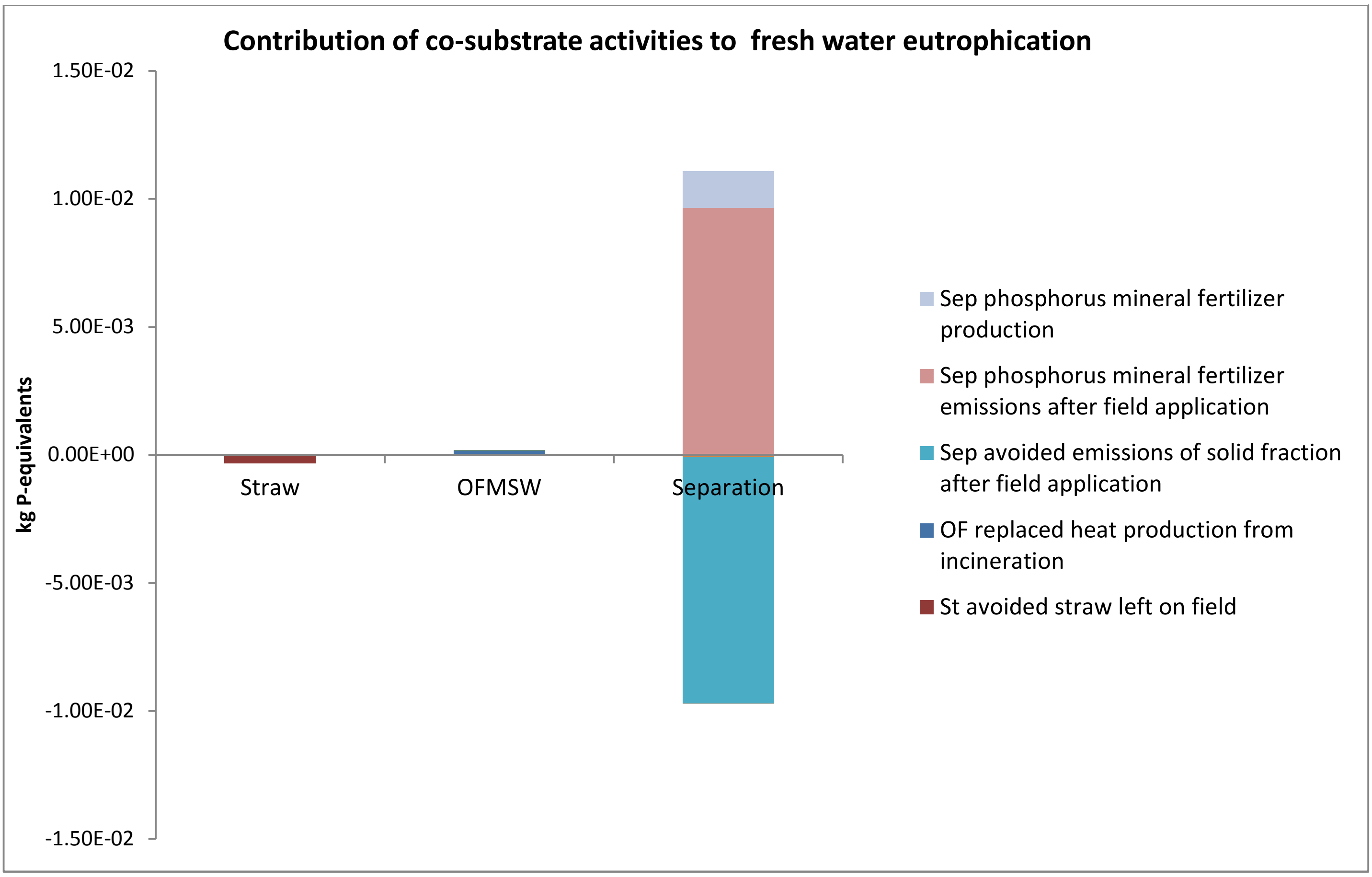
Figure 14 **Contribution of all scenarios to freshwater eutrophication potential throughout each step of the manure management continuum. Biogas + are burdens from biogas production, Biogas – are savings from biogas production, Net biogas is the net contribution from biogas production.**



The straw scenario shows a lower burden for the freshwater eutrophication category than the baseline, even though the phosphorus content applied to the field in this scenario is slightly higher. For the baseline, the input of phosphorus comes solely from the 1000 kg of slurry, the functional unit, whereas the biogas scenarios have an additional input of phosphorus coming from each co-substrate. Phosphorus content of the co-substrates increases starting with straw, followed by the OFMSW and lastly the solid fraction.

Furthermore, since phosphorus does not have a gaseous form, it is assumed that there are no losses of it throughout the life cycle of manure management, until it reaches the field where it can leach. Thus, digesting the co-substrates along with the pig slurry does not diminish the amount of phosphorus that eventually reaches the field. The relationship between nitrogen and phosphorus applied to the field seen in equation (18) in section 8.1.2, partly explains why the straw scenario fares better against the baseline and why the separation scenario fares much worse. The total amount of phosphorus taken up by the crop is directly correlated to the total nitrogen applied to the field (the total nitrogen of co-substrate plus slurry). The more nitrogen applied to the field the higher the total uptake of phosphorus by the crop (equation 18). In this case, the nitrogen added to the field increases in this order, baseline > straw > OFMSW > separation and so does the total uptake of phosphorus by the crop. At the same time, the biogas scenarios add a higher amount of phosphorus to the field. In the case of the straw scenario the extra phosphorus added to the field is not much higher than that of the baseline and with a higher uptake of phosphorus by the crop the outcome means the straw scenario causes a lower environmental impact. In contrast, the extra addition of phosphorus to the field by the separation scenario is significantly higher than that of added by the baseline and although there is a higher uptake by the crop, it is not enough to counteract the extra addition of phosphorus. The result is a lower rate of uptake by the crop and more surplus phosphorus available for leaching by the separation scenario. The same occurs for the OFMSW scenario, but as the co-substrate's phosphorus is not nearly as high as the solid fraction's then this scenario is only 4.6% higher than the baseline.

Figure 15 **Contribution to fresh water eutrophication potential of co-substrate specific activities. Reduced legend shows most important contributions.**



Aside from having the highest phosphorus content, the separation scenario is also affected by the system expansion activities, which ultimately signify a net burden from the biogas process for this scenario. A closer look at Figure 15 reveals that this is due to emissions from mineral phosphorus fertilizer applied to the field that does not receive the solid fraction and emissions related to the production of said mineral fertilizer. These two combined, are higher than the field emissions saved by not applying the solid fraction to the field. Here, the assumption that the fields nearby have a surplus of phosphorus and thus the solid fraction is transported to a far-away field plays an important role, discussed further under 3.1.3. For the straw scenario, there are additional savings, albeit small, of phosphorus leaching from not leaving the straw on the field. For the OFMSW scenario, there is additional eutrophication caused by the production of heat in a conventional way.

To sum up, the straw scenario demonstrates the potential to reduce phosphorus leaching into the environment in comparison to the baseline. This is due to higher nitrogen content in the straw-digestate applied to the field, which causes a higher phosphorus uptake by the crop and a phosphorus content that is only slightly higher than the baseline. The other two biogas scenarios fare worse because the added phosphorus content is significantly higher than the baseline and the higher crop uptake rate cannot compensate enough to produce a saving.

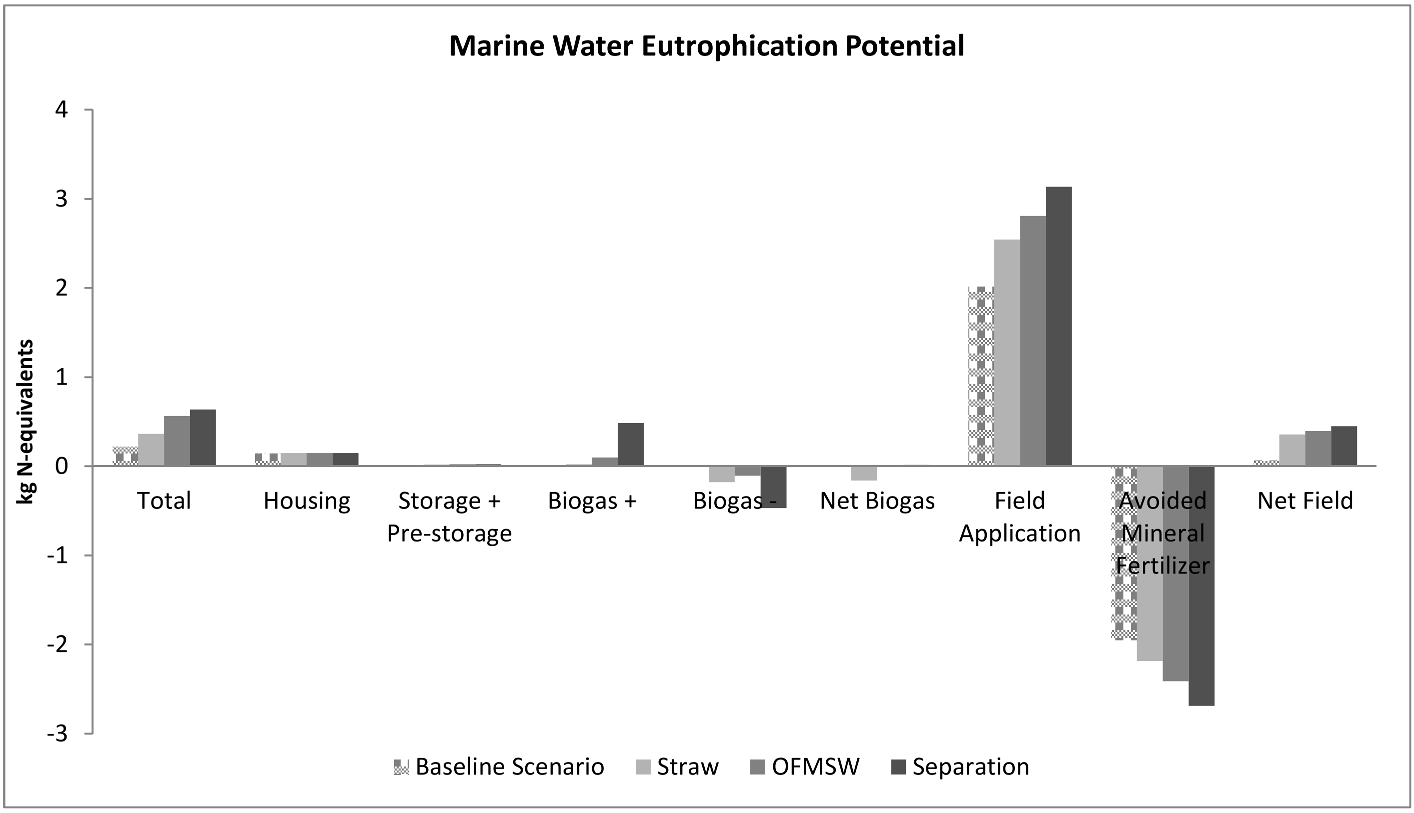
5.4.1 Freshwater Eutrophication Uncertainties

The assumption that the fields where digestate is applied has a surplus of phosphorus built up in the soil has consequences for all scenarios. It is assumed that mineral phosphorus fertilizer will not be used on these fields, and therefore there is no avoided production or application of phosphorus mineral fertilizer. If instead it was assumed that there is replacement of phosphorus mineral fertilizer, then the results might be different for this impact category, though this assumption would not be realistic as the farms that have animal production have a surplus of phosphorus built up in the soil (Whalen et al., 2001). Also, for the separation scenario it would mean that the emissions from production and application of phosphorus mineral fertilizer in the field that does not receive the solid fraction would be negated to some degree by the avoided production and application of phosphorus mineral fertilizer in the field receiving the digestate.

5.5 Marine water Eutrophication Potential

As the marine environment is nitrogen limited, the nitrogen content of the co-substrates as well as the emission factors of the nitrogen pool throughout the life cycle are of great importance for this impact category. This fact is evident in Figure 16, which shows that all biogas scenarios contribute more to marine water eutrophication than the baseline scenario. A reason for this is the fact that for the baseline the nitrogen that is available for eutrophication is solely that which is found in the 1000 kg of slurry, while for the other scenarios there is the nitrogen content of 1000 kg of slurry plus the nitrogen content of the co-substrate. In decreasing order, the co-substrate with the highest nitrogen content is the solid fraction, followed by the OFMSW and lastly the straw. Thus, the impact caused follows this order, with the separation having the highest impact, followed by OFMSW as second highest and straw as the one causing the lowest impact of the biogas scenarios.

Figure 16 Contribution of all scenarios to marine water eutrophication potential throughout each step of the manure management continuum. Biogas + are burdens from biogas production, Biogas – are savings from biogas production, Net biogas is the net contribution from biogas production.



In Figure 16 it can also be seen that the housing step causes a significant contribution to marine water eutrophication. On closer inspection it was revealed that this contribution comes from ammonia emissions to air, which are quite high and at the same level for all scenarios. But by far the largest contributions to marine eutrophication happens after field application, though avoided emissions from application of mineral fertilizer abate the positive contributions by a large margin. In column Net Field, it can be seen that there is a stark difference between the baseline and the biogas scenarios, where the baseline almost comes out even when avoided emissions from mineral fertilizer application are subtracted from emissions after field application of slurry. The sharp contrast between the baseline and biogas scenarios is largely due to NO_3^- leaching into the aquatic environment, which happens at a rate that is 8.9% higher for the biogas scenarios in comparison to the baseline (FARM-N). Smaller contributions that make the biogas scenarios worse off, occur from higher ammonia emissions in the field due to a higher amount of nitrogen being present in TAN form after anaerobic digestion. Also, the rate of nitrogen uptake by the crop is higher for the baseline scenario than the biogas scenarios; a FARM-N calculation which could be looked into in more detail, since it has been stated in the literature that TAN is readily available for plants facilitating uptake (Sommer, Jensen, Clausen, et al., 2006).

In this graph, the Net Biogas column, shows considerable savings for straw, while the separation and OFMSW scenarios' savings/burdens from biogas production almost break even. For the straw scenario, these savings come from avoided nitrogen emissions from straw left on the field for 100 year period.

In summary, the biogas scenarios are able to cause more marine water eutrophication than the baseline, which is largely due to the digestates' higher nitrogen content and higher NO_3^- leaching rate. In addition, the digestates' higher TAN content allow more ammonia emissions to happen.

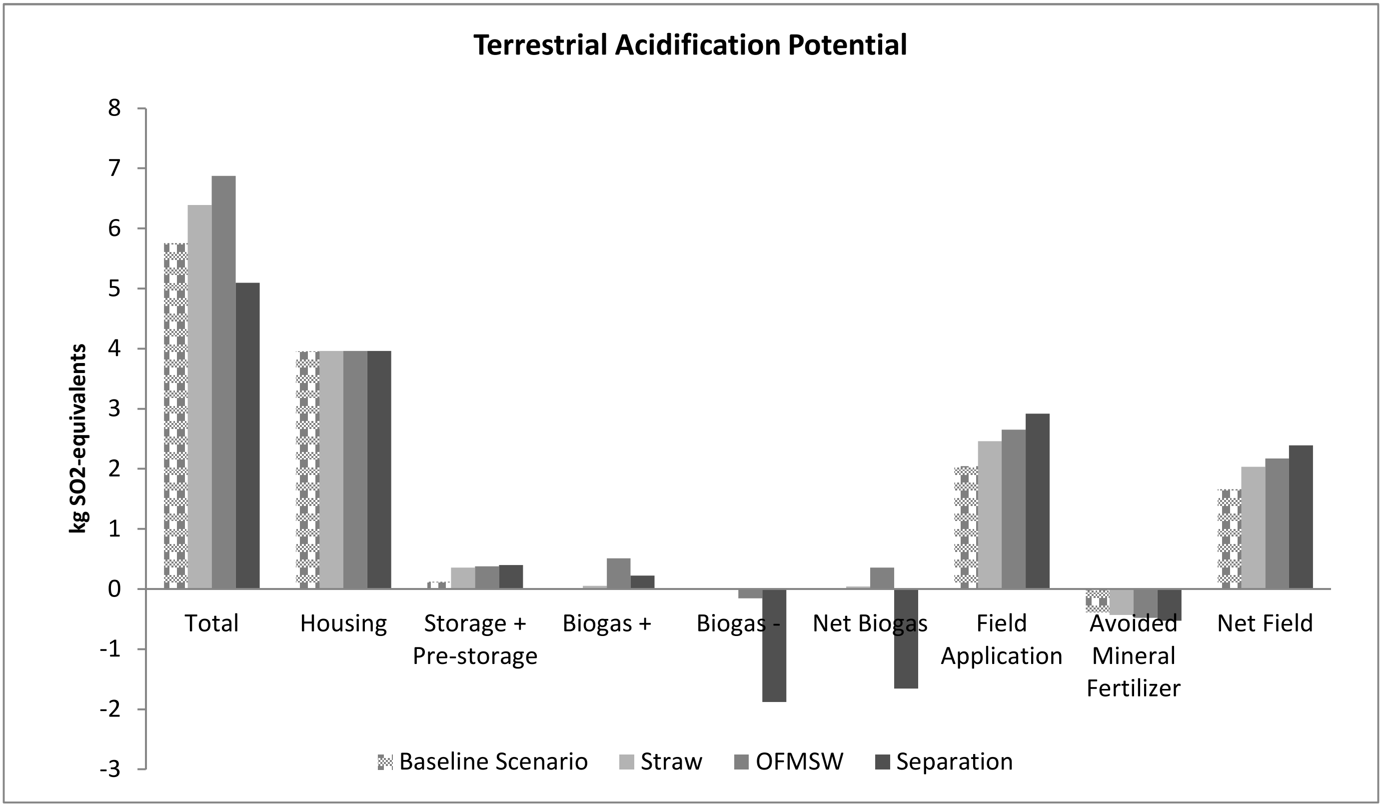
5.5.1 Marine Water Eutrophication Uncertainties

Two areas of improvement are identified here. To begin with, as has been stated throughout the results, the avoided emissions from straw left in the field would be more in line with the rest of the model if calculated for a 10 year period. It is not expected that this change will alter the order of the study's results, but it would indeed be a preferable choice. Secondly, the higher nitrate leaching rate and lower crop nitrogen uptake calculated by FARM-N is a gray area, which could be clarified for improvement in the future.

5.6 Terrestrial Acidification Potential

Emission of ammonia is the main contributor from all scenarios to this impact category. These emissions occur during housing, storage, pre-storage, and after field application. Some co-substrate specific activities also contribute to this impact potential with ammonia and other species such as NO, NO_x, and SO_2.

Figure 17 **Contribution of all scenarios to terrestrial acidification potential throughout each step of the manure management continuum. Biogas + are burdens from biogas production, Biogas – are savings from biogas production, Net biogas is the net contribution from biogas production.**

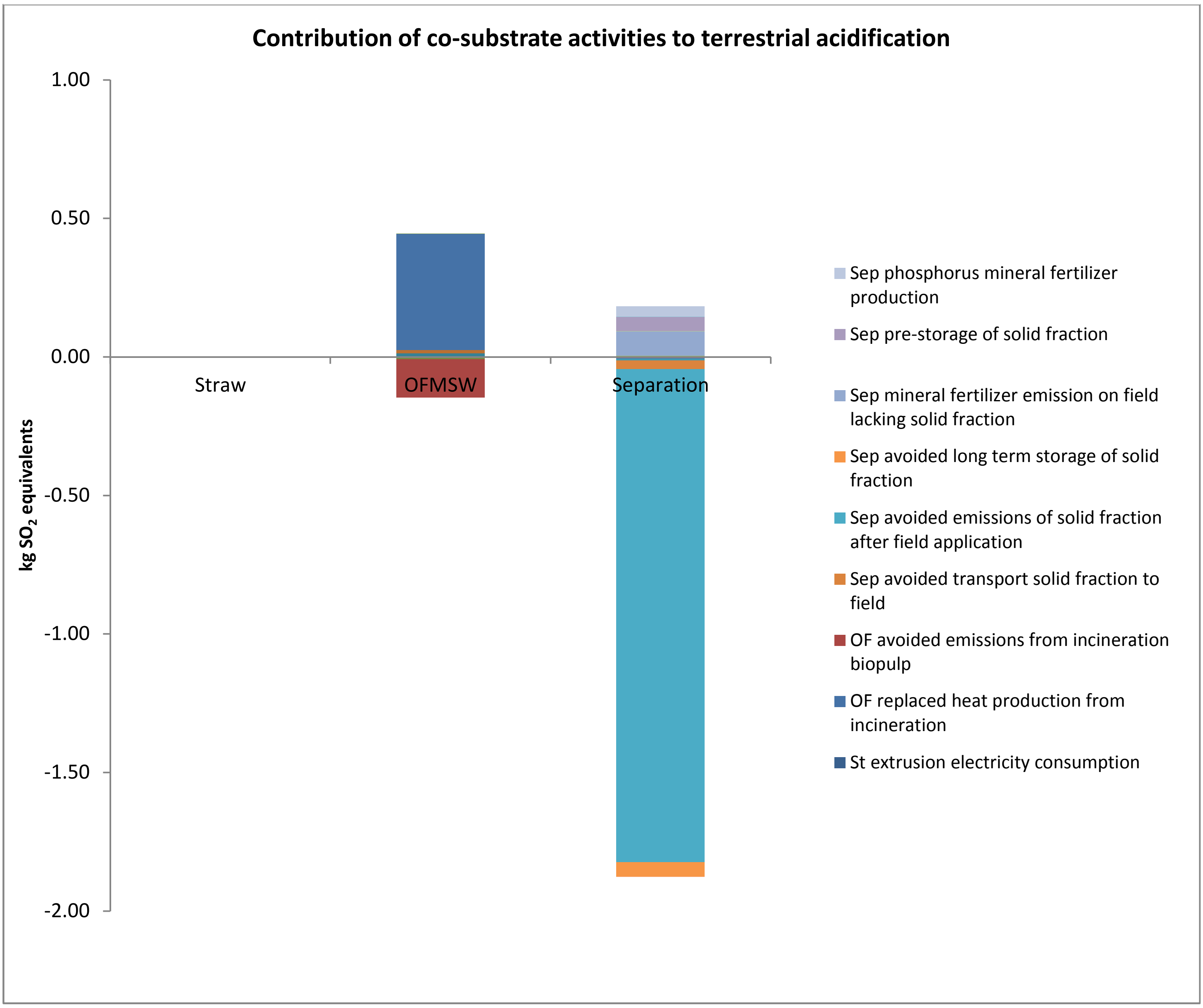


The biggest contribution to acidification is observed during the housing stage of the life cycle, which can be explained by the rather high ammonia emission rate of 0.25 kg NH_3-N per kg TAN (FARM-N; (Hutchings et al., 2012c)). This rate compares rather poorly with the emission factor of 16% of TAN presented in the latest Danish emission inventory for agriculture; based on normative Danish standards for slurry in housing with fully slated floors (Mikkelsen et al., 2011). Revision of this value should be considered, though the change will not alter the overall results, as the housing stage contributes the same for all modeled scenarios. Additionally, the biogas scenarios have an extra pre-storage step which contributes ammonia emissions that the baseline does not have; explaining why the impact during Storage + Pre-storage is higher for the biogas scenarios.

Field emissions of ammonia reflect well the nitrogen content of the digestate in the biogas scenarios, as previously discussed in section 5.5. More specifically, the higher ammonia emissions for the biogas scenario are a direct consequence of the higher amount of nitrogen available in TAN form, which volatizes into ammonia emissions. Thus, the baseline has the lowest ammonia emissions and this increases in the following order baseline > straw > OFMSW > separation according to TAN added to the field. Moderate savings of acidification are observed from avoided mineral fertilizer, again in order of nitrogen content in the slurry or digestate applied to the field.

For this impact category, the activities making the separation scenario less acidifying than the baseline are attributed to biogas production. A closer look at Figure 18 makes it evident that the co-substrate specific activities create big savings for the separation scenario. The savings are produced mainly by avoided emissions of the solid fraction after field application and also by avoided transport and long-term storage of the solid fraction for the separation scenario. The OFMSW shows a loading from having to produce the heat that would have been produced by incineration of the organic fraction. Heat production emissions contribute more to the impact category than the avoided emissions from not incinerating the organic fraction, resulting in a net loading from biogas activities for the OFMSW. Note also that the straw scenario shows no savings or loading from co-substrate specific activities. As has been mentioned in section 3.1.2, modeling for the straw left on the field was performed by the DAISY model, which does not include NH_3 emissions, so a possible savings was excluded due to modeling limitations. But, as straw contains very little TAN, it is not thought to cause different results.

Figure 18 **Contribution to terrestrial acidification potential of co-substrate specific activities. Reduced legend shows main contributions.**



5.6.1 Terrestrial Acidification Uncertainties

Here, the OFMSW scenario is at a disadvantage because of the heat that must be produced as a system expansion. As discussed before, a system expansion implies the introduction of an uncertainty. When this choice is made, several questions arise, such as is this the right choice for system expansion? or should the heat be produced from 100% coal or a representative energy mix for Denmark? It is considered that the right choices have been made, taking into consideration the scope of this study and time limitations. More time for careful consideration of these questions would surely improve the scenario. For the straw scenario, missing NH_3 calculations for the avoided straw left on field do not benefit the overall impact of the straw scenario. Saved ammonia emissions make the model more consistent.

5.7 Sensitivity Analysis

The results discussed reveal many areas that could be explored further in order to test the model's robustness. Testing the model is important to see if the results hold, even when critical data is changed (Baumann and Tillman, 2009). Thus, the methane yield potential of the co-substrates was identified as a critical parameter and the choice in calculating the baseline scenario's long term storage as a critical choice to be investigated in the sensitivity analysis.

5.7.1 Effect of Methane Yield Range

In the literature, it is possible to find a wide range of methane yields for various substrates. This is often due to the operational conditions under which the substrate is fermented. As discussed in section 2, the hydraulic retention time, organic loading rate, and C:N ratio of substrates are just a few of the parameters affecting the ability of microbes to degrade a substrate into CO_2 and CH_4. It is reasonable to expect that methane yields in real plants will vary according to the operational parameters just mentioned. Also, the methane yields during digestion of the co-substrates have important consequences for the results. Methane yields affect the degree of degradation of organic matter in the co-substrates and in turn this affects the amount of organic nitrogen that mineralizes into TAN, thereby affecting all nitrogen emissions and also the amount of organic matter that is available for degradation in the field. This highly regulatory parameter is worth investigating.

For this sensitivity analysis a value representing the low end and a value representing the high end of the methane potential range found in literature was chosen for each co-substrate. Thus, the previous results are for methane potential yields that lie in the middle of these ranges. The values chosen are presented in Table 11.

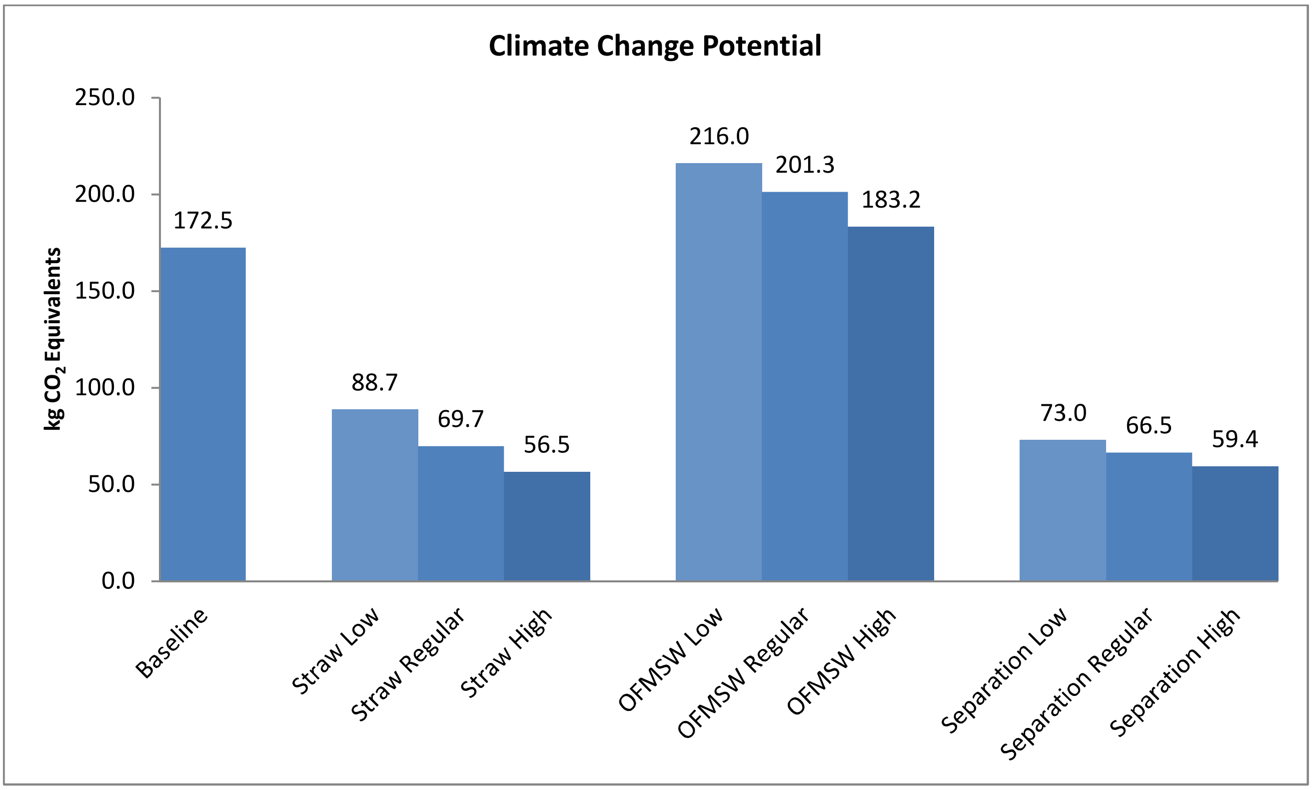
Table 11 **Range of potential methane yields found in literature for each co-substrate in the three biogas scenarios.**

	Methane yield potential, L CH_4/kg VS			Reference
	Low	Regular	High	
Straw*	150	279	370	(Wang et al., 2009; Angelidaki and Ellegaard, 2003; Torres-Castillo et al., 1995)
OFMSW	210	340	500	(Davidsson et al., 2007; Davidsson et al., 2007; Luostarinen et al., 2011)
Separation	78.7	170	270	(Menardo et al., 2011; Hamelin et al., 2010; Luostarinen et al., 2011;)

* Straw methane yields are shown without 70% increase from extrusion

For all impact categories, the results for the sensitivity analysis behave in a logical way. Changes in methane yield potential do not reverse the results of the model or change them significantly. This shows that the model is able to make robust conclusions regarding biogas potential yields. This can be better appreciated by looking at the figures in this section.

Figure 19 Sensitivity analysis results for climate change potential impact category. Scenarios were tested with a low and high methane yield potential.



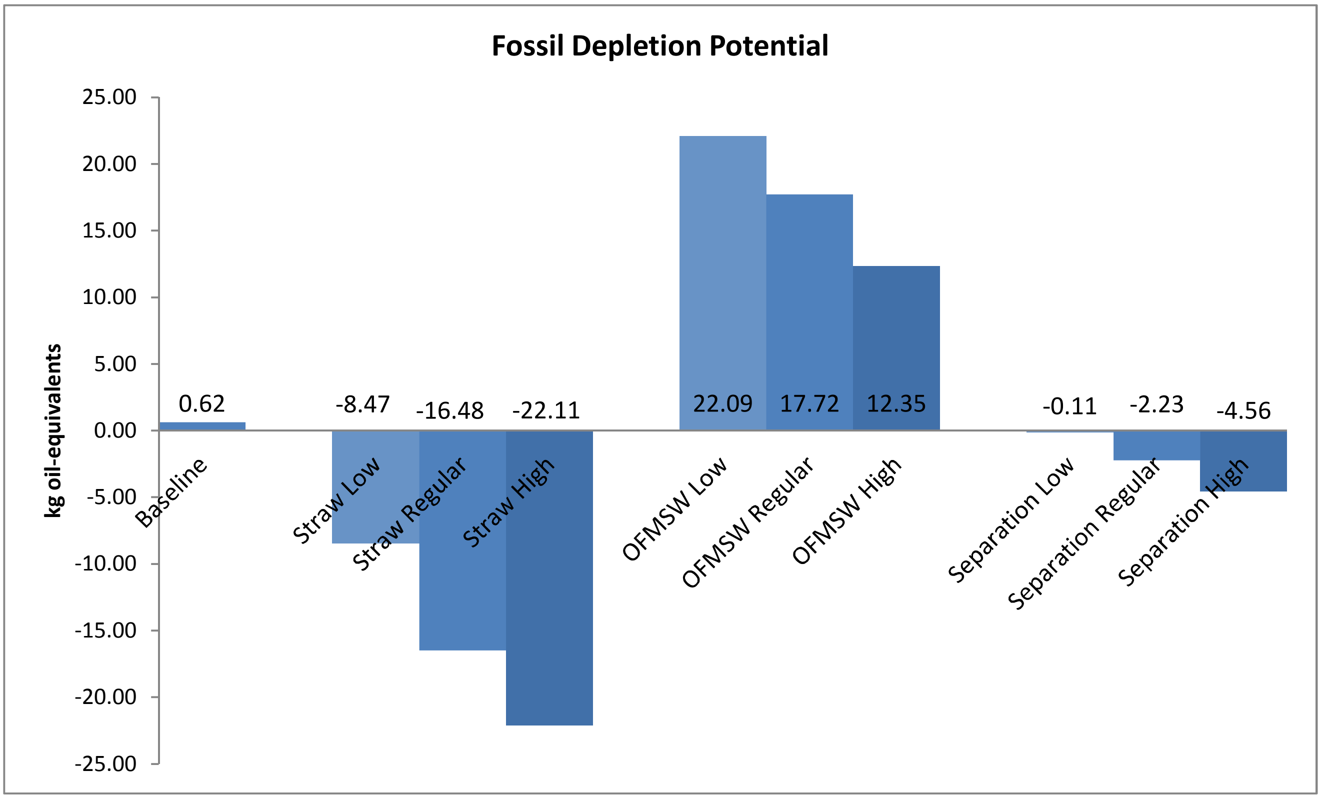
With regards to climate change potential, the biogas scenarios perform better against the baseline, if a high methane yield is applied to the co-substrate and perform worse if a low methane yield is applied. This is logical, since a high methane yield means a higher rate of degradation of the co-substrate in the biogas reactor and less organic matter left to degrade into CO_2 emissions, both in the field and during digestate storage. On the contrary, if the methane yields are low, more organic matter is left, thus a higher carbon pool is available for release into the atmosphere. As stated in section 5.2, the biggest contribution to this impact category happens after field application, so if emissions are sought to be mitigated, using substrates with a high methane potential is beneficial. After degradation with the high end of the methane yield potential, the scenario performing the best is the straw scenario, which adds the least organic matter to the field stage. This is slightly different to the initial results where the separation scenario performed the best. It seems to indicate that there is a threshold at which the straw scenario is able to surpass the separation scenario, if the straw scenario's methane yield is high enough, producing more savings for climate change. The scenario adding the second highest amount of organic matter to the field is the OFMSW, but as explained previously high burdens to climate change for this scenario come from the production of electricity and heat, resulting in this scenario's poor performance for all impact categories in comparison to the baseline scenario. The scenario adding the most organic matter to the field is the separation scenario, but same as before, savings from the avoided application and long-term storage of the solid fraction are able to compensate for high emissions after field application. It is important to note that whether in the high end or low end of the methane yield potential range, two of the biogas scenarios always perform better than the baseline, causing overall savings. The initial results are never reversed.

Similarly for the fossil depletion category, if the methane yields are low, the biogas scenarios are able to save less depletion and if the methane yields are high the biogas scenarios save more depletion. The relative order of the scenarios is not changed in comparison to the baseline and initial results. Weather high or low methane yields are applied, the straw scenario performs best, followed by the separation scenario and finally the OFMSW performs worse in comparison to the baseline scenario.

Figure 20 is instrumental in showing the ability of the co-substrates to produce methane. Here, it is evident that extruded straw is very effective at producing methane and also has a wide range of methane production. The methane potential yield range is also wide for the OFMSW scenario and generally higher than solid fraction’s methane yield potentials. The latter are generally low and of a narrow range. Thus, the possibilities to increase methane yields and therein biogas profits for the solid fraction are not as good as for the other substrates. Yet, this observation does not take into account the benefits the separation scenario shows in a few of the impact categories. Also, synergies in methane production, that is, unaccounted for increases in methane productions because a co-substrate has characteristics that are a good complement to the slurry, are not modeled in this study and would require a different set of data to make the calculations. But, they are a very real possibility in actual biogas plant operation.

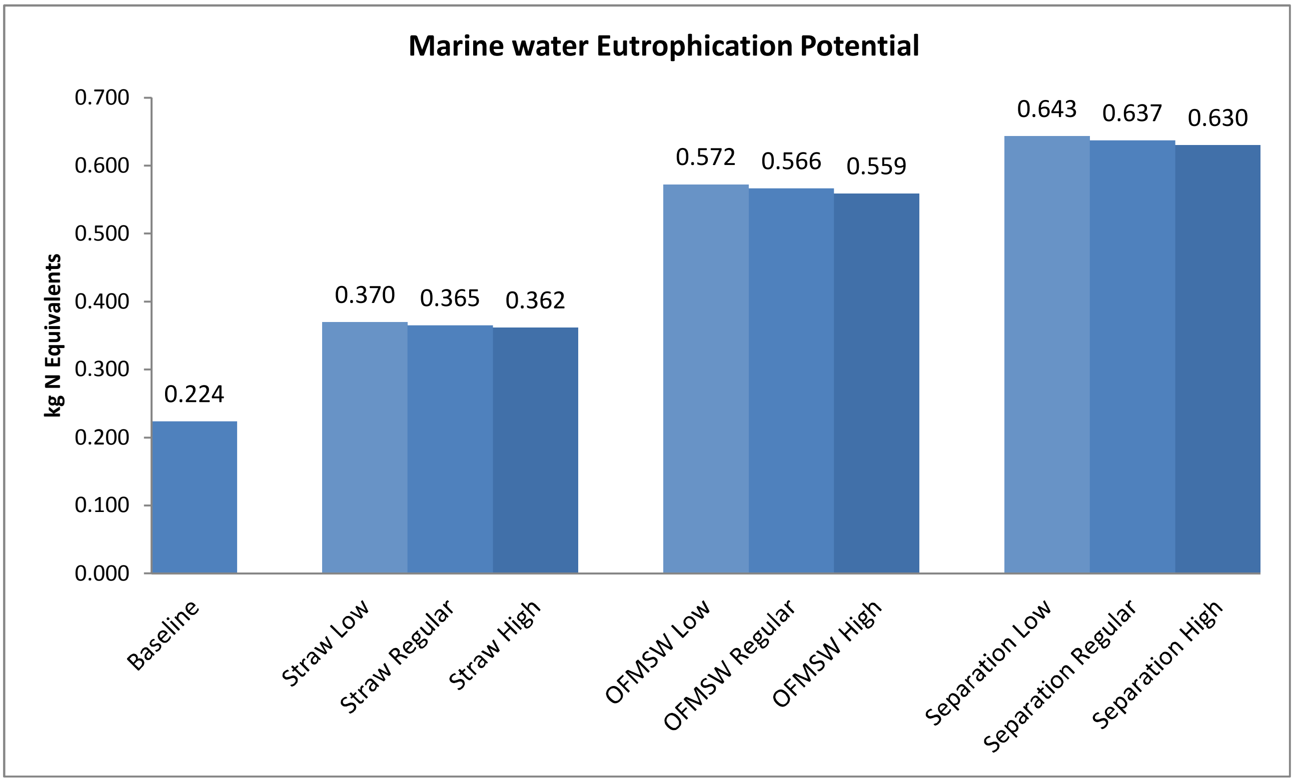
The results for marine water eutrophication and terrestrial acidification shown in Figure 21 and Figure 22 vary only slightly depending on whether a low or high methane yield is applied. Again, the initial results are not reversed and the general order of the scenarios is not changed.

Figure 20 **Sensitivity analysis results for the fossil depletion potential impact category for high and low methane yield potentials.**



For marine water eutrophication potential, it turns out that the higher the methane yield and degradation of the co-substrates in the reactor, the lower the overall environmental impact of the scenario. This has a logical explanation. As more organic matter is degraded so is more organic nitrogen released into TAN form, which means that there is a higher emission of ammonia, but also that there is less overall nitrogen available for the other emissions of N_2, N_2O, and NO_3^-. Earlier it was discovered that the biggest contribution to marine water eutrophication came from nitrate leaching. It is then reasonable that as there is less nitrogen available to leach, when a high methane yield is applied, then there will be less marine water eutrophication with high methane yields and more eutrophication with low methane yields and low organic matter degradation.

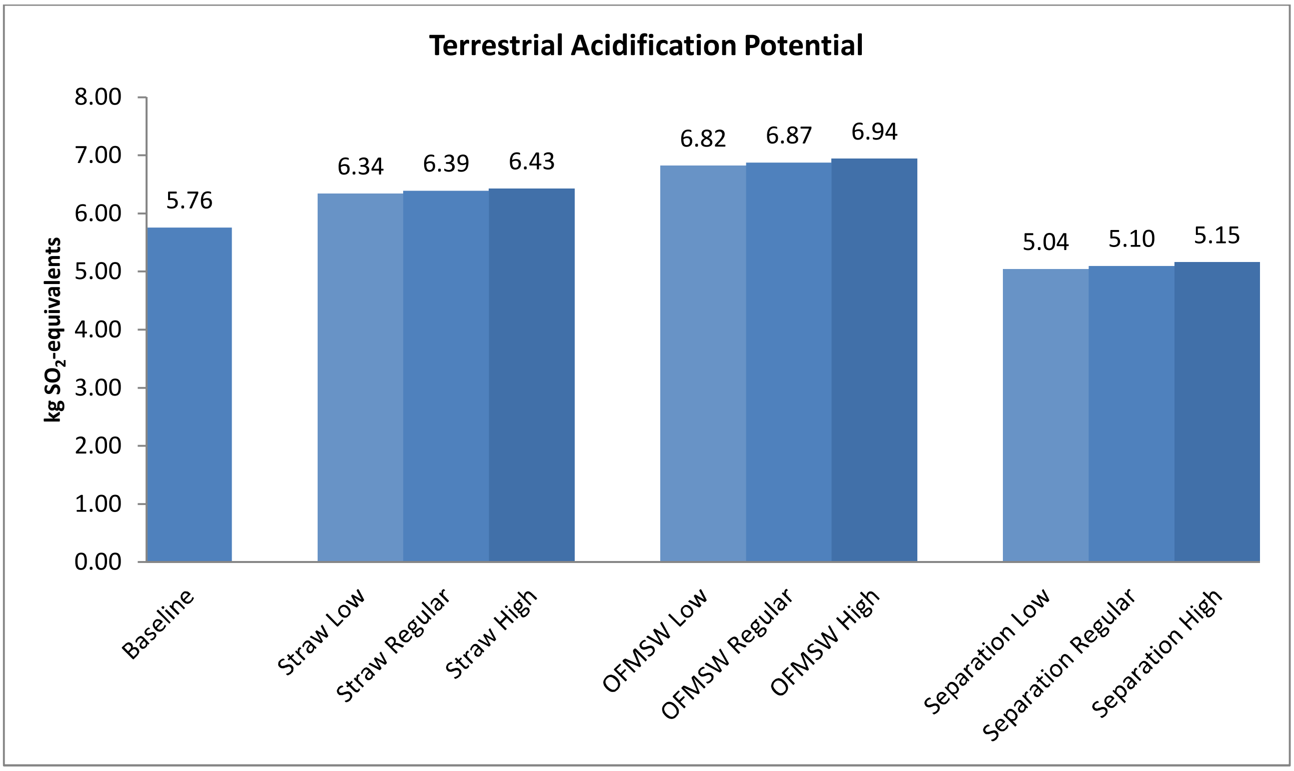
Figure 21 Sensitivity analysis results for the marine water eutrophication impact category for high and low methane yield potentials.



By the same process, the terrestrial acidification impact category shows the exact opposite result as the marine water eutrophication category. For acidification, ammonia emissions are extremely important and as previously stated these emissions increase when high methane yields are applied because more TAN becomes available. So for low methane yields, less organic matter degrades, less organic nitrogen mineralizes into TAN and less ammonia is ultimately emitted.

Various studies have explored the effects of digestion on slurry ammonia emissions and though some have found that digested slurry emits more ammonia (Amon et al., 2006; Sommer, Jensen, Clausen, et al., 2006), others have found that improved infiltration rates of digested slurry counteract higher ammonia emissions (Pain et al., 1989; Rubæk et al., 1996). In this study, higher infiltration rates of digested slurry are not included in the calculations. Therefore, it is noted that this is an area where the model could benefit from more detailed calculations of all factors affecting ammonia emissions.

Figure 22 **Sensitivity analysis results for the terrestrial acidification impact category for high and low methane yield potentials.**



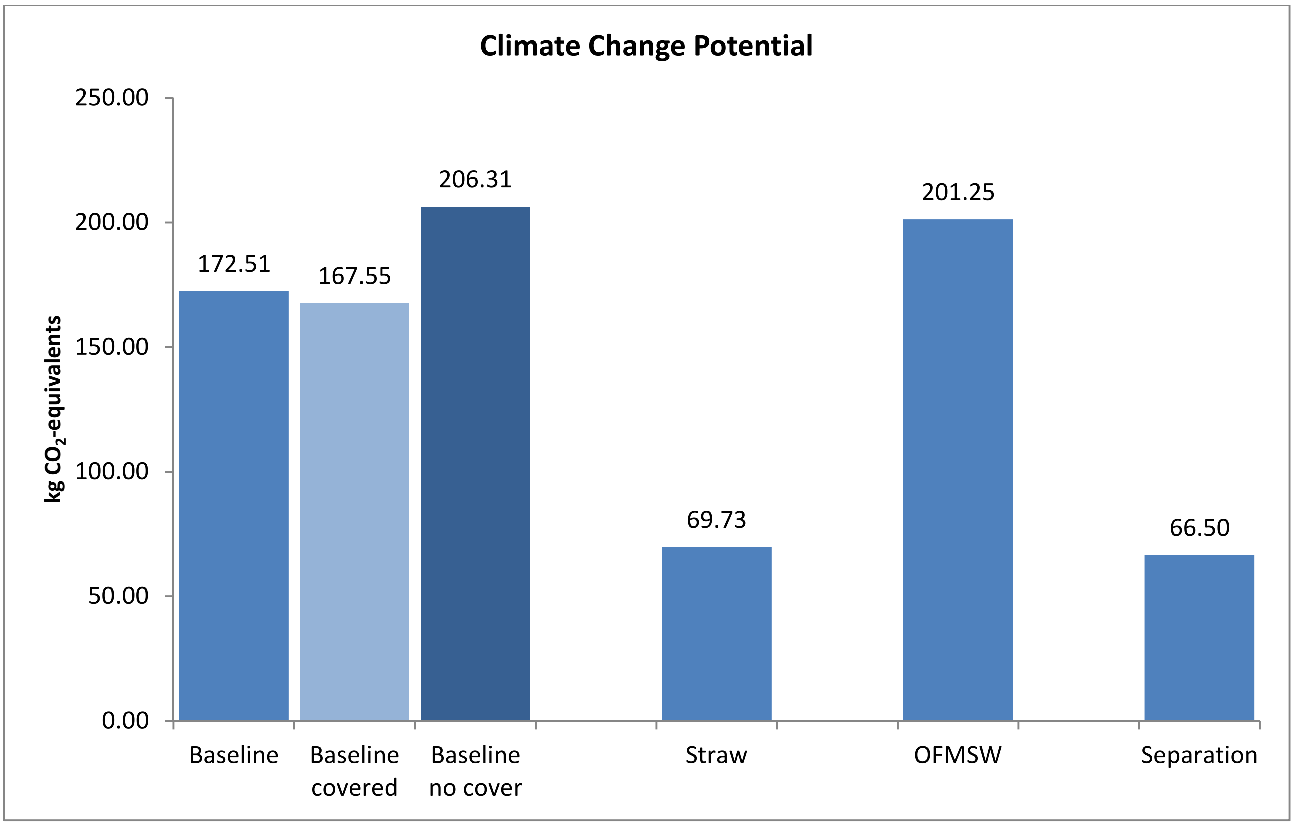
Results for the fresh water eutrophication category showed no changes, when either low or high methane yields were applied. This is also to be expected, as phosphorus is highly inert and is not affected by how much organic matter degrades in the reactor. Thus, a graph is not shown since there are no changes to the results.

From this sensitivity analysis it is possible to say that the model's results are robust, as the general order of the scenarios' performance against the baseline, in the various impact categories, is not changed. Furthermore, the results exhibited by each impact category can all be explained logically and offer insight into the processes governing the overall impacts. It has been useful to do this sensitivity analysis, not only to see that the model's results hold, but also to be able to make predictions of what impacts can be expected to get worse or better depending on the capability of a co-substrate to produce methane.

5.7.2 Effect of Storage Calculation

A second sensitivity analysis was performed after a suspicion arose about calculations done for the baseline storage, which are different than for the biogas scenarios digestate storage and pre-storage. The effects of a cover, which are to reduce methane emissions by 38% on average (Sommer et al., 2000), are not included in the baseline's storage calculation, but are included for digestate storages and pre-storage (refer to section 8.1.1, 8.2.1 and 8.2.4 for full explanation of how calculations were done). The results of how storage is calculated affect the climate change impact category and to a very small degree the fossil depletion category. Thus, a graph is only shown for the climate change impact category.

Figure 23 **Sensitivity analysis results show the climate change impact category tested by using different storage calculation for the baseline. Includes the effects of covering and not covering the storage.**



The sensitivity analysis was performed to see if the overall results would change, if the method for calculating CO_2 and CH_4 emissions during long-term storage of the baseline was changed to that used to calculate pre-storage. A second difference is that baseline storage was calculated for a period of 6 months. The new storage calculations were done for a period of 270 days, around 9 months, based on Hansen et al., 2006, which says slurry storage is emptied in April and is from then on allowed to fill again until the next year.

Despite the longer time period for baseline storage, 'Baseline covered' showed a lower contribution to climate change than the initial Baseline results for this impact category, see Figure 23. As can be seen, when the new storage calculation is performed, the 'Baseline covered' becomes better than 'Baseline' only by 3%, not enough to alter the overall results of the study.

A second part of this sensitivity analysis shows the overall impact of the baseline if the storage is not covered, but calculating the emissions as calculated in pre-storage (section 8.2.1). Here, it can be seen that when storage is not covered the baseline has a much higher negative impact on climate change, around 20% more for 'Baseline no cover' in comparison to 'Baseline'. In this case, all biogas scenarios are less burdensome to climate change than the new baseline.

The conclusion here is that the initial storage calculation is acceptable, as changing it did not change the overall results significantly. Furthermore, it was possible to see that covering the storage is of great importance and can alter the results of the study, giving the OFMSW scenario a greater advantage than that previously experienced.

6 Conclusion

The present study has modeled the conventional manure management strategies with the goal to compare it to manure management with biogas production and more specifically different co-substrates to slurry. Now it is possible to address the initial aims of the project: to identify hot spots of biogas production and manure management, to discover environmental implications of using different co-substrates and to offer recommendations for manure management and biogas production.

The most important hotspot identified throughout the manure management continuum, occurs from emissions after field application of organic matter. These emissions contribute significantly to all impact categories and it was determined that to degrade the organic matter prior to field application lowers greenhouse emissions. The most important hotspots identified during biogas production happen due to heat consumption of the biogas reactor and during upgrade of biogas due to fugitive emissions of methane. Substrates with high water content produce more emissions as they require higher heat consumption. Other important hotspots during the biogas process are co-substrate specific.

A visual representation, summary for the results of this study, is offered in Table 12, which shows the performance of the three biogas scenarios against the baseline for all impact categories.

Table 12 **Performance of the three biogas scenarios against the baseline scenario. Numbers 1,2,3 signify the rating of the scenario against the baseline; 1 being the best, 3 the worse. Cell color red signifies that the scenario performs worse than the baseline while green means the scenario is better than the baseline.**

	Straw	OFMSW	Separation
Climate change	🟩 2	🟥 3	🟩 1
Fossil depletion	🟩 1	🟥 3	🟩 2
Freshwater eutrophication	🟩 1	🟥 2	🟥 3
Marine water eutrophication	🟥 1	🟥 2	🟥 3
Terrestrial acidification	🟥 2	🟥 3	🟩 1

From Table 12 it is possible to see that the benefits of using biogas production as a way to manage manure are co-substrate dependent. In two of the scenarios modeled there are lower environmental impacts for several of the impact categories, while one of the scenarios, the OFMSW scenario, is not preferable in comparison to conventional manure management. The differences between the co-substrates were found to be a consequence of two factors; the co-substrates chemical characteristics and the alternatives for that co-substrate if it were not used for biogas production. In this regard, the OFMSW produced the most environmental loading because its alternative use is to produce energy from incineration. The production of said energy in a conventional way is responsible for the emissions that make this scenario a bad choice. The separation scenario showed advantages in three impact categories, but its high nutrient content and low methane yield make it a less desirable substrate. More importantly, the high nutrient content of the solid fraction is a problem for eutrophication. In contrast, the straw scenario showed the lowest environmental loading overall, out of the three biogas scenarios. Extruded straw's low nutrient content, low water content, and high methane yield cause several benefits. It means there are less nutrients that can leach in the field, lower heating needs in the biogas reactor, high replacement of natural gas and less organic matter available for degradation in the field.

Due to the factors just mentioned, extruded straw is recommended as a co-substrate to slurry for biogas production. Extruded straw, as a resource, could significantly contribute to the Danish goals of using 50% of manure for biogas production and ultimately become a fossil free society by 2050. But, caution should be exercised, as the alternative use of straw, e.g. incineration of straw, could change the results of this study.

More co-substrates should be investigated through LCA in order to better guide future production of biogas, so that the health of ecosystems and humans is ensured. In addition, more research is needed to identify synergies of co-substrates that complement each other well and result in higher methane yields. Lastly, an expansion of this model with more scenarios to address the concerns under the uncertainties section would allow for more possibilities to be explored. The future of biogas has not yet been decided. Future scientific contributions will play an important role in shaping the energy future of our societies.

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8 APPENDIX A CALCULATIONS

In this section, choices for the models are thoroughly justified and specific assumptions applying directly to the calculations are presented. Calculations for the major steps of the life cycle are shown. A point of departure for the calculations in this project originates in the CLEANWASTE project. Thus, calculations for the baseline come from the mentioned larger project. Also, the method of calculating field processes comes from the same source.

8.1 Baseline Scenario

Calculations for the baseline scenario where done by the CLEANWASTE group. Much of the data used originates in the FARM-N model, which can be referred to for a deeper explanation of how processes are calculated. Calculations for the chemical composition of the reference slurry and housing will not be explained here, because they are the same for all scenarios and do not influence the final results of this study. The interested reader can refer to the FARM-N scientific explanation for more information on these processes. Calculations shown here include, emissions calculations and transformations of four variables tracked for all substrates in this study: OM, TAN, ON, H_2O. Ash and P are considered inert and stay the same throughout the steps of the life cycles, with the exception of the field process.

8.1.1 Long-term storage of slurry

Emissions calculated during this stage are methane, carbon dioxide, and ammonia. Nitrous oxide emissions during storage are assumed to be negligible based on Sommer et al. 2000, and so are N_2 emissions (Hutchings et al., 2012c). This applies to all storage processes calculated. Changes in organic matter and carbon during this stage happen as follows:

(1) $$OM_{lost} = OM_{input} * \ OM_{degrade-rate}$$

Where OM_{lost} (kg) is the organic matter that degrades during storage, OM_{input} (kg) is the organic matter ex-animal in housing, and $OM_{degrade\text{-}rate}$ (kg kg^{-1}) is the degradation rate of organic matter during storage for a period of 6 months. The degradation of organic matter, $OM_{degrade\text{-}rate}$ (kg kg^{-1}) is 0.185 kg per kg OM, based on (Hutchings et al., 2012c). The change in OM_{input} during storage can then be calculated, where OM_{output} is the remaining organic matter that goes to the next step in the process.

(2) $$OM_{output} = OM_{input} - OM_{lost}$$

From the organic matter lost during storage, a methane emission can be calculated where:

(3) $$Methane_{storage} = \ OM_{lost} * \ CH_4perOM_{rate}$$

$Methane_{storage}$ is the total amount of carbon released as CH_4-C in kg and CH_4perOM_{rate} (kg kg^{-1}) is the rate at which degrading OM releases carbon as CH_4-C. Here, CH_4perOM_{rate} is 0.23 kg C per kg OM, as it is assumed there is 46% carbon in the organic matter and that 50% of the emission is released as CH_4-C (Hutchings et al., 2012c). Carbon dioxide released is assumed to be the other 50%, thus CO_2-C = CH_4-C emitted.

As organic matter degrades, a part of the organic nitrogen mineralizes. It becomes part of TAN, which is the sum of NH_3-N and NH_4^+-N, though it is often used as a synonym for NH_4^+ ; the form readily available for crop uptake (Hamelin et al., 2010). This process occurs as follows:

(4) $$ON_{mineralize} = ON_{input} * OM_{degrade-rate}$$

Where ON_{input} (kg) is the organic nitrogen in the slurry left after housing and coming into storage. Thus, the portion of organic nitrogen to mineralize, $ON_{mineralize}$ (kg) is directly correlated to the organic matter degradation during storage by, $OM_{degrade\text{-}rate}$. Changes to TAN in the substrate can be calculated as:

(5) $$TAN_{output} = TAN_{input} + ON_{mineralize}$$

Where TAN_{input} (kg) is the input of TAN into the process, after the previous process. In this case, it is the TAN after housing, coming into storage and TAN_{output} (kg), is the output of TAN after storage into the next process. The ammonia emission, dependent on TAN amounts, is then calculated by:

(6) $$NH_3 emission = TAN_{output} * NH_3Store_{rate}$$

The new quantity of TAN is multiplied by an emission factor, NH_3Store_{rate} (kg NH_3-N kg^{-1} TAN) in order to get the NH_3 emission (kg NH_3-N). The emission factor used for ammonia is 0.01 kg kg^{-1}, for storage covered by a tent and is based on (Hansen et al., 2008).

The changes to ON during storage can be calculated by subtracting $ON_{mineralize}$ from the initial input of ON into the storage. The calculation is as follows:

(7) $$ON_{output} = ON_{input} - ON_{mineralize}$$

This gives ON after storage in kg N. Lastly, degradation of organic matter during storage is an anaerobic process consuming water. How much water is degraded per amount of organic matter is substrate specific and can be calculated by using Buswell's formula for methane fermentation of carbohydrates (Symons and Buswell, 1933). The formula is as follows:

(8) $$C_nH_aO_b + \left(n - \frac{a}{4} - \frac{b}{2}\right) H_2O \rightarrow \left(\frac{n}{2} - \frac{a}{8} + \frac{b}{4}\right) CO_2 + \left(\frac{n}{2} + \frac{a}{8} - \frac{b}{4}\right) CH_4$$

Here, C,H,O have their usual chemical meaning representing the elements and n, a, b are the number of respective atoms. By knowing the average carbohydrate composition of a substrate and modifying this formula, the share of water that degrades per slurry organic matter, H_2O_{rate}, in kg H_2O per kg OM can be calculated. For slurry the result is 0.286 kg kg^{-1}, which can be used to calculate the total water lost H_2O_{lost} for slurry, under any anaerobic condition, not just storage. From there, the total amount of water that degrades in the storage is calculated as:

(9) $$H_2O_{lost} = OM_{lost} * H_2O_{lost}$$

8.1.2 Field application of slurry

Important gaseous emissions of CO_2, N_2O, NH_3, and N_2 happen after application of organic matter to fields and are calculated for this study. In addition, nitrate and phosphorus leaching, which ends up in the aquatic ecosystem, are also calculated. In this study, all carbon in organic matter is assumed to degrade into CO_2 emissions in the field. Methane emissions after field application are considered negligible base on (Sommer et al., 1996; Wulf et al., 2002).

Degradation of organic matter, $OM_{degrade}$ (kg), in the field is derived from $N_{immobilized}$ (kg), the C:N ratio (kg kg^{-1}) of the soil humus and a constant assumed for the amount of C in OM of the soil, $CperOM_{soil}$ (kg C kg^{-1} OM).

(10) $$OM_{degrade} = \frac{N_{immobilized}}{CperOM_{soil}}$$

Where the C:N ratio is assumed to be 10 kg kg^{-1} and $CperOM_{soil}$ is 0.56 kg C per kg OM (FARM-N; Hutchings et al., 2012b). $N_{immobilized}$ is the amount of nitrogen staying in the field after all N emissions and uptake by the crop has taken place, calculated as follows:

(11) $$N_{immobilized} = Total\_N_{after}NH_3 - (CropNuptake + N_2O_{emission} + N_{2\ emission} + NO_3^-{}_{loss})$$

Where $Total\_N_{after}NH_3$ is the slurry nitrogen left in kg after ammonia emission. In turn CropNuptake is calculated from the total nitrogen available after ammonia emissions, which is multiplied by a nitrogen uptake rate generated in FARM-N, in this case 0.514 kg N kg^{-1} N.

(12) $$CropNuptake = Total\_N_{after}NH_3 * CropNuptake_{rate}$$

Ammonia emissions after field application are calculated with an emission factor and depend on the quantity of TAN in the slurry applied to the field. The rate, $rateNH_3$ is in kg NH_3-N kg^{-1} TAN and TAN_{input} (kg) is the TAN remaining in the slurry after storage. Here, $rateNH_3$ is 0.16 kg NH_3-N kg^{-1} TAN, according to (Hansen et al., 2008). The calculation is as follows:

(13) $$NH_3emission = TAN_{input} * rateNH_3$$

After the ammonia emission has been calculated, the nitrogen remaining $Total\_N_{after}NH_3$ (kg) from which all subsequent emissions of nitrogen arise can be calculated.

(14) $$Total\_N_{after}NH_3 = TAN_{input} - NH_3emission$$

The subsequent gaseous emissions of nitrogen, N_2 and N_2O and NO_3^- leaching are calculated according to the following equation:

(15) $$N_2emission\ or\ N_2Oemission\ or\ NO_3^-loss = \\ Total\_N_{after}NH_3 * rateN_2\ or\ rateN_2O\ or\ rateNO_3^-$$

For N_2 emissions, $rateN_2$ (kg N_2-N kg^{-1} total N) and $rateNO_3^-$ come from calculations in FARM-N, while $rateN_2O$ is based on 2006 IPCC values. The rates are 0.041 (kg N_2-N kg^{-1} total N), 0.395 (kg NO_3^--N kg^{-1} total N), and 0.02 (kg N_2O-N kg^{-1} total N) respectively.

The area for spreading the slurry is based on regulation by the European Comission, in Council Directive 91/676/EEC of 12 December 1991. A limit of 170 kg $ha^{-1}yr^{-1}$, N_{limit}, of manure-N application was established in this Directive. Thus, the area (ha) for application of the slurry, which depends on the slurry's total N content after storage, Total_N (kg), is calculated by:

(16) $$Area\ application = \frac{Total\_N}{N_{limit}}$$

Also, avoided application and production of mineral fertilizer was calculated. The amount of nitrogen mineral fertilizer that slurry may replace is calculated as follows, where $N_{efficiency}$ is the efficiency at which slurry N can replace mineral N. Based on FARM-N;Hutchings et al., 2012b, $N_{efficiency}$ is 75%.

(17) $$N_{replace} = Total\_N \, * \, N_{efficiency}$$

Lastly, losses of phosphorus in the field are regulated by nitrogen applied to the field. Phosphorus leached, P_{loss} (kg), is calculated as a share, P_{rate} (kg P kg^{-1} P surplus), of the surplus remaining after P has been taken up by the crop. Uptake of P by the crop is calculated as:

(18) $$CropPuptake = UptakeP_{yearly} * \left(\frac{Total\_N}{N_{limit}}\right)$$

Here CropPuptake in kg P, varies with Total_N (kg N), the total N in the slurry applied to the field. $UptakeP_{yearly}$ is a constant, 21.5 kg P $ha^{-1}yr^{-1}$, assumed to be the yearly crop requirement based on national guidelines for fertilization (Hamelin et al., 2011). It is then possible to calculate the surplus of P after uptake, $P_{surplus}$, and finally the P_{loss} by:

(19) $$P_{surplus} = P_{input} - CropPuptake$$

It is assumed that 0.05 kg P per kg of surplus P, P_{rate}, leaches to the aquatic environment, based on Nielsen and Wenzel, 2007. P_{input} (kg) is the total amount of phosphorus in the slurry. The calculation is as follows:

(20) $$P_{loss} = \; P_{surplus} * P_{rate}$$

Ash and water are assumed to incorporate into the soil. Because the environment is aerobic, there is oxidation of water. However, since the amount of water applied to the field does not affect the study results, calculations for the oxidation of water are not shown.

8.2 Biogas Scenarios

Processes included in the biogas scenario, absent from the baseline, are pre-storage of slurry, degradation in biogas reactor, biogas upgrade and digestate storage. Calculations are shown for each of these processes. Additionally, a small discussion about calculations in the field is presented.

8.2.1 Pre-storage of slurry

Prior to digestion, the slurry is briefly stored in outdoor storage at the farm. The same emissions have been calculated here as in 8.1.1. Calculation of organic matter degradation, methane and carbon dioxide emissions have been done differently than for the baseline long term storage. Calculation of changes in ON, TAN, H_2O and of ammonia emissions are performed in the same way and with the same rates as the baseline (refer to equations 4-9).

Organic matter degradation is derived from methane emissions possible during storage, CH_4-C_{rate} with a value of 0.011 (g C h^{-1} kg^{-1} OM), which comes from laboratory scale experiment that reproduced storage conditions for slurry at different temperatures (Sommer et al., 2007). The same study, estimated a ratio of CH_4-C:CH_4-C+CO_2 which is used to calculate the total carbon emitted during storage and subsequently the CO_2 emission. A range of 0.1 to 0.3 is stated in the study, therefore it was chosen to use 0.2 ratio. The CH_4-C emission (kg) is thereby calculated as:

$$(21) \qquad CH_4C_{emission} = \frac{CH_4C_{rate} * OM_{input} * Days_{storage} * 24}{1000}$$

CH_4-$C_{emission}$ is then given in kg of CH_4-C. Here, OM_{input} (kg) is the organic matter of the slurry after animal housing, $Days_{storage}$ are the number of days the slurry will be stored, 24 is the number of hours in a day. It is assumed that the duration of pre-storage is 10 days, after a personal communication (Møller, 2012). Also, this is the amount of methane produced in storage is not the same as that released to the atmosphere. It is assumed that the storage cover reduces methane emissions by 38% (Sommer et al., 2000) and a calculation is done to account for this reduction. Total carbon lost during storage is:

$$(22) \qquad C_{lost} = \frac{CH_4C_{emission}}{CH_4C{:}CH_4C + CO_2C}$$

CO_2-$C_{emission}$ (kg CO_2-C) can then be calculated by subtracting CH_4-$C_{emission}$ from the total C lost.

$$(23) \qquad CO_2C_{emission} = C_{lost} - CH_4C_{emission}$$

Since the amount of C per OM of the slurry is known, $CperOM_{slurry}$ (0.46 kg C kg^{-1}OM, assumption in 8.1.1), the organic matter lost during storage can be calculated by:

$$(24) \qquad OM_{lost} = \frac{C_{lost}}{CperOM_{slurry}}$$

OM_{lost} is given in kg. By knowing how much organic matter degrades during storage, the degradation rate $OM_{degrade\text{-}rate}$ (kg kg^{-1}) can be calculated as follows:

$$(25) \qquad OM_{degrade-rate} = \frac{OM_{lost}}{OM_{input}}$$

Where OM_{input} is the initial amount of organic matter that enters the storage after housing (kg). With the $OM_{degrade\text{-}rate}$ it is then possible to calculate the ON that mineralizes as describes in section 8.1.1. The rest of the calculations for TAN, ON, H2O outputs also follow the method of the mentioned section.

8.2.2 Biogas reactor

In the biogas reactor, the reference slurry meets the co-substrate where the materials are fermented at thermophilic temperature. Methane yield is calculated from methane yield potential found in literature corresponding to each substrate. Whenever possible, the methane yield potential was chosen to represent a 15 HRT, which is most typical for centralized biogas plants in Denmark (Hansen et al., 2006). The biogas composition, that is the percentages of methane, carbon dioxide, and other gases in the biogas was determined by taking an average of the values found in literature (Burton and Turner, 2003; Jørgensen, 2009). These values are 61.75% methane, 32.75% carbon dioxide, and 1.5% other (composed of a mix of H_2, H_2O, NH_3…). The materials are immediately fermented upon arrival to the biogas plant. In reality they might spent up to 7 days in receptor tank (Jørgensen, 2009), but for simplicity this step is excluded from the model. An explicit calculation of water vapor produced during biogas production was also excluded from this study, as the calculation could not be reconciled with the biogas composition just stated. The biogas production and degradation rates occurring in the reactor are calculated separately for the slurry and the co-substrates. They are then added together to get the total biogas production from the mixture. After degradation, the substrates' composition is also added together to get the chemical composition of the digestate e.g. OM of reference slurry left after degradation is added to OM of straw left after degradation to give OM of digestate. The methane yield of the reference slurry, $CH_4slurry_{rate}$, for an HRT of 15 is 297.82 L kg^{-1}OM. The methane yields for the three co-substrates investigated, $CH_4co\text{-}sub_{rate}$, vary and are given in L kg^{-1}OM. A total volume of methane, Total_CH_4Vol in (L), for the slurry fermented is calculated by:

(26) $$Total\_CH_4Vol_{slurry} = CH_4slurry_{rate} * OM_{input,slurry}$$

Where $OM_{input,slurry}$ is the organic matter in the slurry after pre-storage. By the same formula, a total volume of methane produced is calculated for the co-substrate, with substitutions of the appropriate variables for $OM_{input,co\text{-}sub}$ and $CH_4co\text{-}sub_{rate}$ giving Total_$CH_4Vol_{co\text{-}sub}$. The volume of CO_2 (L) produced can be derived from the volume of methane produced and the percentages described above by the following equation:

(27) $$Total\_CO_2Vol_{slurry} = \left(\frac{Total\_CH_4Vol_{slurry}*36.75}{61.75}\right)$$

The same procedure is followed to calculate the CO_2 yield of the co-substrates, substituting Total_CH_4Vol_{slurry} for Total_$CH_4Vol_{co\text{-}sub}$ which gives Total_$CO_2Vol_{co\text{-}sub}$. The CO_2 and CH_4 volumes are then converted into CO_2-C and CH_4-C masses through the ideal gas law, so that the total amount of carbon that is made into biogas of the material can be calculated. The ideal gas law is:

(28) $$PV = nRT$$

Where P is pressure (atm), V is volume (l), n is moles, R is the ideal gas constant, and T temperature (K^o) at standard conditions. With this equation, it is possible to convert the volumes of CO_2 and CH_4 into moles of CO_2 and CH_4 and subsequently masses TotalCH_4_$mass_{slurry}$ (kg CH_4-C) and Total_CO_2mass_{slurry} (kg CO_2-C). The mass of total carbon that is lost to biogas, $C_{lost,biogas}$ (kg C) is then given by:

(29) $$C_{lost,biogas} = Total\_CO_2mass_{slurry} + Total\_CH_4mass_{slurry}$$

To obtain the amount of organic matter that is lost during production of biogas, equation (24) is used. This requires that the C in OM of the material is known. The value $CperOM_{slurry}$ has been defined in section 8.2.1. $CperOM_{co\text{-}sub}$ has been gathered or calculated for each co-substrate. For the solid fraction this value is assume to be the same as for slurry and for the biopulp and straw the value was calculated through the known carbon content and organic matter of the substrate. The values are 0.535 kg C kg^{-1} OM straw and 0.565 kg C kg^{-1} OM biopulp. OM degradation rates for the slurry and the co-substrates are calculated using equation (25). The rest of the components of slurry and co-substrates (TAN, ON, H_2O) can be calculated in the same way as in section 8.1.1. The amount of H_2O degrading per co-substrate OM has to be calculated in the same way as before, by using Buswell's formula (10). For straw this value is O.232 kg H_2O kg^{-1}OM straw, for the biopulp it is 0.365 kg H_2O kg^{-1}OM biopulp, for the solid fraction this value is assumed to be equal as for slurry. From these calculations it is possible to know the final mass, TAN, ON, OM, and H_2O content of the digestate (along with P and Ash, which do not change for the substrates throughout the steps).

Heat and electricity consumption of the reactor are also calculated. Heat consumption was calculated as described in Hamelin et al., 2010, by using the specific heat of dry matter and water to calculate how much heat is needed to raise the temperature of the substrates to 55^{o}C (thermophilic temperature). Electricity consumption was calculated using the value 0.09 kWh per m^3 of biogas produced (Nielsen, 2004).

In addition, fugitive emissions can be found at the biogas producing plant and also at the biogas upgrading facility. The leak coming from the biogas plant reactor was calculated based on results from a Swedish study that tested several biogas plants for fugitive emissions (Holmgren et al., 2012). Efforts were made to gather Danish data on this subject, but it was not possible to find any well documented information on the topic. The same Swedish study investigated leaks in biogas upgrading facilities, which are also used for calculations.

A percentage of the methane produced that leaks from the reactor, $CH_4reactor_{\%}$, was identified to be 1.6% of CH_4 produced per year (Holmgren et al., 2012). The methane leak from the reactor, $CH_4leak_{reactor}$ (l), is then calculated by:

(30) $$CH_4leak_{reactor} = \left(Total\_CH_4Vol_{slurry} + Total\_CH_4Vol_{co-sub}\right) * 1.6\%$$

The total biogas volume (CH_4+CO_2+other) minus the reactor methane leak is the biogas available for upgrade. The biogas gets upgraded into biomethane and injected into the natural gas grid.

8.2.3 Biogas upgrade

Currently, most centralized biogas plants in Denmark produce electricity and heat through a cogeneration unit (Al Seadi, 2000). In this study, it has been chosen to upgrade all biogas so that it may be injected in the natural gas grid, avoiding the production of natural gas. Recent legislation in Denmark will potentially drive biogas producers to upgrade their biogas instead, since a financial incentive of 79-115 DKK per GJ will be paid out for biogas that is injected into the grid (Energipolitik, 2012).

The electricity required to upgrade biogas into biomethane is 0.25 kWh per m^3 of biogas (Petersson and Wellinger, 2009). The electricity requirement can be calculated by multiplying the amount of biogas produced in the reactor, minus the reactor's methane leak, by this value. In addition, methane leaks from upgrading facilities, $CH_4leak_{upgrade}$ (m^3), were measure to be 2.7% of the methane content in the biogas, $CH_4content_{biogas}$ (L CH_4) (Holmgren et al., 2012). $CH_4content_{biogas}$ is the methane left in the biogas after leakage from the reactor, calculated by (30). Thus, total methane available for injection into the grid, CH_4grid (m^3), is:

(31) $$CH_4grid = CH_4content_{biogas} - CH_4leak_{upgrade}$$

Then the energy value of the methane being injected in the grid, $CH_4energy$ (kWh), can be calculated from the heating value of methane, CH_4heat (kWh m^{-3} CH_4), which is 9.94 according to Hamelin et al. 2010. The energy value of methane needs to be calculated as the process 'DK: natural gas, high pressure, at consumer' requires the input as energy (kWh).

(32) $$CH_4energy = CH_4grid * CH_4heat$$

8.2.4 Long-term storage digestate

Digestate is the second product derived from biogas production. It consists of a mixture of the slurry and straw OM, ON, TAN, H_2O, P and ash. It is not expected that digestate degrades in an equal manner as raw slurry, therefore a different calculation has been done for digestate degradation during long-term storage. Methane yield potential of digestate, $CH_4digestate_{potential}$, was found to be between 5-25% of the methane yield potential of the substrate, $CH_4slurryrate$ (L CH_4 kg^{-1} OM) (Angelidaki et al., 2006; Paavola and Rintala, 2008). In a study conducted in Danish centralized biogas plants, Angelidaki et al. 2006 found residual biogas potential to be between 5-15%, thus a value of 10% was chosen. An average of the methane yield potentials stated in literature was done for each co-substrate, $CH_4slurry_{rate}$ and CH_4cosub_{rate} (L kg^{-1} OM). Then, a weighted average of the methane yield potential of the slurry plus that of the co-substrate added was performed; representing the share of slurry OM, $Share_{slurry,OM}$ (kg kg^{-1}) and co-substrate OM, $Share_{cosub,OM}$ (kg kg^{-1}), in the digestate. The rate of methane production during digestate storage, $CH_4digestate_{rate}$ (L kg^{-1} OM) is then given by:

(33) $$CH_4digestate_{rate} = \left(CH_4slurry_{rate} * Share_{slurry,OM} + CH_4cosub_{rate} * Share_{cosub,OM}\right) * CH_4digestate_{potential}$$

A total volume of methane produced during storage, $CH_4Vol_{digestate,storage}$ (l), is calculated by:

(34) $$CH_4Vol_{digestate,storage} = OM_{digestate} * CH_4digestate_{rate}$$

Where $OM_{digestate}$ is the organic matter of digestate after the biogas reactor. To calculate the organic matter lost during storage, methane and carbon dioxide emissions from storage, calculations (22-25) in pre-storage are followed. With the $OM_{degrade\text{-}rate}$ it is then possible to calculate the ON that mineralizes as described in section 8.1.1. The rest of the calculations for TAN, ON, H_2O outputs also follow the method of the mentioned section.

8.2.5 Field application of digestate

Calculations for field application of digestate followed the method delineated under 8.1.2 Field application of slurry. Review of articles dealing with application of slurry vs. application of digestate to the field was done in order to make this decision. In literature, very similar rates were found for all important emissions (NH_3, N_2O, CO_2,CH_4)(Petersen et al., 1996; Sommer et al., 1996; Wulf et al., 2002; Amon et al., 2006; Clemens et al., 2006; Bhandral et al., 2009; Thomsen et al., 2010; Chadwick et al., 2011; Mikkelsen et al., 2011). However, it is important to note that CropNuptake rate, 0.489 kg N kg^{-1}N, for digestate is lower than that for slurry and that nitrate loss rate is higher for digestate, 0.484 kg N kg^{-1}N, than for raw slurry. These values were calculated by FARM-N and are not supported by literature.

8.3 Co-substrate specific activities

System expansion was performed for each scenario to represent the alternative activity that must be taken due to the co-substrates use in the biogas plant. Calculations for each of these alternative activities are presented here.

8.3.1 Straw scenario

An amount of straw is taken to the biogas plant for co-digestion with slurry. It was assumed that this amount of straw would otherwise be left in the fields, if it was not co-digested for biogas. This is based on the large amounts of straw left on the fields in Denmark each year, around 2.1 million tons (Skøtt, 2011).

It is necessary to calculate carbon, nitrogen, and phosphorus emissions from the straw left on the field. These are avoided emissions because the straw will not be left in the field. Due to time constraints and a lack of a better way, these emission were calculated using DAISY.

8.3.2 OFMSW scenario

As a consequence of the OFMSW being taken to a biogas plant for co-digestion with slurry, the same fraction will not be incinerated for energy production. This means the energy that would have been produced by incineration of the organic fraction still must be produced, but in a conventional way. In addition, emissions from the incineration of the organic fraction will be avoided. It was assumed that the organic fraction would otherwise be incinerated because this is the number one method of household waste disposal in Denmark (Miljøministeriet og Miljøstyrelsen, 2011).

Avoided CO_2 and NO_x emissions from the avoided incineration of the OFMSW were calculated as follows. It was assumed that all carbon in the biopulp is released during incineration. Here $Carbon_{emission}$ (kg CO_2-C) is the carbon to be released, $OM_{biopulp}$ (kg OM) is the organic matter of the biopulp to be incinerated and $CperOM_{biopulp}$ (kg C kg^{-1} OM) is the carbon content of the biopulp's organic matter.

(35) $$Carbon_{emission} = OM_{biopulp} * CperOM_{biopulp}$$

NO_x emissions, $NOx_{emission}$ (kg NO_x), are calculated based on a rate of nitrous oxide emissions in Møller et al., 2008 for incinerated Danish waste. The rate, $rateNO_x$ (kg t^{-1} waste), is 0.86. NO_x emission is then:

(36) $$NOx_{emission} = \frac{(Biopulp_{in} - Biopulp_{H_2O}) * rateNOx}{1000}$$

Where $Biopulp_{in}$ (kg biopulp) is the total mass of the organic fraction being incinerated and $Biopulp_{H2O}$ is the total amount of water in the organic fraction.

To calculate how much electricity and heat must be produced in a conventional way, the lower and upper heating values, LHV (MJ kg^{-1}), and UHV (MJ kg^{-1}), and efficiencies, LHV_{ef} and UHV_{ef}, must be known. The calculations performed to arrive at these numbers are not shown here. Table 13 shows the values calculated. The lower heating value and efficiency was used to calculate electricity. The upper heating value and efficiency was used because it is assumed that the latent heat of water will be recovered.

Table 13 Upper and lower heating values plus heat and electricity efficiencies of the upper and lower heating values. Calculated from (Møller et al., 2008).

	Biopulp (MJ/kg)
Lower heating value, MJ/kg	0.45
Upper heating value, MJ/kg	2.39
Electricity Efficiency/Lower heating value	0.22
Heat Efficiency/upper heating value	0.54

The heat, $Heat_{biopulp}$ (MJ kg^{-1}), and electricity, $El_{biopulp}$ (MJ kg^{-1}), produced from the organic fraction can thereby be calculated as:

(37) $$Heat_{biopulp} = UHV_{ef} * UHV$$

(38) $$El_{biopulp} = LHV_{ef} * LHV$$

The heat and electricity produced by the biopulp must instead be produced in a conventional way. In this project it was assumed that the electricity and heat were both produced from coal.

8.3.3 Separation scenario

Activities that must be modeled as a consequence of the solid fraction of separated slurry being taken to a biogas plant for co-digestion with slurry are: avoided long term storage of the solid fraction, avoided field application of the solid fraction, application and production of P-mineral fertilizer. The latter occurs because it is assumed that when the solid fraction is separated, it is transported to fields far-away from the animal far that are in need of phosphorus fertilization. If this field does not receive the solid fraction then it must use conventional phosphorus mineral fertilizer.

Avoided long term storage of the solid fraction is calculated in the same manner as the pre-storage of slurry. Refer to section 8.2.1 for details on the calculation.

Avoided field application of the solid fraction is calculated in the same manner as field application of slurry in the baseline scenario. However, rates of emissions, immobilization and crop uptake are different. These can be seen in Table 14.

Table 14 Emission rates for solid fraction after field application

	Rate	Unit	Reference
N_2 emission	0.038	kg kg^{-1} N	FARM-N
NH_3 emission	0.390	kg kg^{-1} TAN	Hansen et al. 2008
NO_3 loss	0.332	kg kg^{-1} N	FARM-N
Crop N uptake	0.435	kg kg^{-1} N	FARM-N
N immobilized field	0.175	kg kg^{-1} N	FARM-N
Crop P uptake	0.528	kg kg^{-1} P	FARM-N
P immobilized field	0.449	kg kg^{-1} P	FARM-N

Lastly, application and production of phosphorus mineral fertilizer is calculated for single superphosphate (P_2O_5) mineral fertilizer. A ready-made process was used for the production of single superphosphate. The phosphorus applied to the field is assumed to have 100% efficiency in replacing mineral fertilizer. Thus, the only calculation needed is the conversion of phosphorus into single super phosphate, which can be done by using molar masses. Phosphorus emission into the aquatic ecosystem from the applied single superphosphate are calculated in the same manner as phosphorus leaching after field application in the baseline scenario (section 8.1.2).

9 APPENDIX B ASSUMPTIONS

Scenario	Process	Value	Unit	Description	Reference
All				Total nitrogen (ON + TAN) replaces mineral-N fertilizer	Hutchings et al., 2012c
	Avoided application of N-mineral fertilizer	170	kg N $ha^{-1}yr^{-1}$	Application limit for manure N application	Hutchings et al., 2012c
	Avoided application of N-mineral fertilizer	129.7	kg N $ha^{-1}yr^{-1}$	Mineral N fertilizer application for a JB3 soil	FARM-N
	Avoided application of N-mineral fertilizer			N from the atmosphere and sowing are included in emissions from mineral N fertilizer	FARM-N
	Avoided application of N-mineral fertilizer			Mineral fertilizer replaced is ammonium nitrate, ‘DE: Ammonium nitrate (AN, solid)	PE International, 2012
	Avoided application of N-mineral fertilizer			Application of mineral fertilizer is with "GLO: fertilizing; mineral fertilizer", is assumed most representative	PE International, 2012
	Avoided application of N-mineral fertilizer			For fertilizer spreading "EU-27: Diesel mix at refinery" is used as petrol	PE International, 2012
	Avoided application of N-mineral fertilizer	0.0265	kg NH_3-N kg^{-1}N	Proportion of total nitrogen emitted as NH_3-N after application of mineral fertilizer	FARM-N
	Avoided application of N-mineral fertilizer	0.024	kg N_2-N kg^{-1} N	Proportion of total nitrogen emitted as N_2-N after application of mineral fertilizer	FARM-N
	Avoided application of N-mineral fertilizer	0.0195	kg N_2O-N kg^{-1} N	Proportion of total nitrogen emitted as N_2O-N after application of mineral fertilizer	FARM-N
	Avoided application of N-mineral fertilizer	0.407	kg NO_3-N kg^{-1} N	Proportion of total nitrogen emitted as NO_3-N after application of mineral fertilizer	FARM-N
	Avoided application of N-mineral	0.59	kg N kg^{-1} N	Proportion of total nitrogen taken up by crop after application of mineral fertilizer	FARM-N

fertilizer				
Field	10	kg C kg^{-1}N	C:N ratio of the soil humus	Petersen et al., 2005
Field	21.5	kg P ha^{-1}yr^{-1}	Average yearly uptake of phosphorus by crop	Hamelin et al., 2011
Field	0.05	kg P kg^{-1} surplus P	Proportion of P surplus lost to the aquatic environment	Nielsen and Wenzel, 2007
Field	0.705	kg kg^{-1} OM	Water produced during oxidation of organic matter	Hutchings et al., 2012c
	0.56	kg C kg^{-1} OM	Carbon content of soil organic matter	Hutchings et al., 2012c
Field			Fields near the animal production farm are assumed to have P build up in soils	Whalen et al., 2001
Field			Fields 100 km away from animal production farm are assumed to have low P status	Whalen et al., 2001
Field			Emissions after field application are included for 10 years, as Farm-N calculates emissions for that period	FARM-N
Field			Surplus of P is N regulated	FARM-N
General			NOx emissions are not taken into account in this study, except for incineration of biopulp	Own assumption
General			Potassium content of manure is not tracked through this study	Own assumption
General	0.46	kg C kg^{-1} OM	Carbon content in organic matter in slurry and solid fraction	Hutchings et al., 2012c
General	0.286	kg H_2O kg^{-1} OM	Portion of water degrading per slurry OM, calculation	Symons and Buswell, 1933
General			All electricity produced is included in the model with the process 'DK: Electricity from hard coal'	PE International
Housing	0.15	kg kg^{-1}	Share of degraded OM that is emitted as CH_4-C	Petersen and Ambus, 2006
Housing	215.25	kg 1000 kg^{-1} slurry	Drinking water spilt	Hutchings et al., 2012c
Housing	71.75	kg 1000 kg^{-1} slurry	Mass of water used for cleaning	Hutchings et al., 2012c
Housing	60.27	kg 1000 kg^{-1} slurry	Evaporation of water from animal housing	Hutchings et al., 2012c
Housing	0	kg kg^{-1} TAN-N	N_2 emission rate during housing (ignored here, due to assumed short residence time)	Hutchings et al., 2012c
Housing	0	kg kg^{-1} TAN-N	N_2O-N emission rate during housing (ignored here, due to assumed short residence time)	Hutchings et al., 2012c
Housing	0.25	kg kg^{-1} TAN-N	NH_3-N emission rate during housing	Sommer, Jensen, Hutchings, et al., 2006
Housing	0.185	kg kg^{-1} OM	Degradation rate of OM during housing	System Analysis
Housing			Slurry is stirred before storage under slated floor storage in the pig housing	Own assumption

	Housing			Slurry is pumped to a covered outdoor storage facility	Own assumption
	Housing			Cleaning of stables is done with water only, no disinfectant	Own assumption
	Housing			The high pressure cleaner is a Poseidon 4-28 of Nilfisk (Power = 2,9 kW (electricity), 580 L/h)	http://issuu.com/wolterink/docs/nilfisk-alto-2010
	Housing	1.01	MJ	Energy needed for cleaning housing	http://issuu.com/wolterink/docs/nilfisk-alto-2010
	Storage, Pre-storage	0	kg kg^{-1} N	N_2O emissions assumed negligible during storage	Sommer et al., 2000
	Storage, Pre-storage	0	kg kg^{-1} N	Proportion of total N in the slurry entering storage that is lost as N_2 (assumed no crust)	Hutchings et al., 2012c
	Storage, Pre-storage	0	Kg m^{-2}	Added precipitation is 0, as slurry storage is covered by tent	Hutchings et al., 2012c
	Storage, Pre-storage, Digestate storage	0.01	kg kg^{-1} TAN-N	Proportion of TAN-N in slurry entering storage that is emitted as NH_3-N	Hansen et al., 2008
	Storage, Pre-storage	0	kg kg^{-1} H_2O	Evaporation rate for slurry storage	Hutchings et al., 2012c
Baseline	Field			Slurry is spread by trailing hose	FARM-N
	Field			Slurry is stirred before it is pumped into the trailing hose	Own assumption
	Field	75	%	Efficiency of slurry to replace mineral N fertilizer	FARM-N
	Field	8	km	Transport distant to and at field	Own assumption
	Field			P-mineral fertilizer is not replaced, as it is assumed there is P build up in soils	Whalen et al., 2001
	Field	0.514	kg kg^{-1} N	Proportion of N in slurry that is taken up by crops (after NH_3 emission)	FARM-N
	Field	0.041	kg kg^{-1} N	Proportion of N in slurry that is emitted as N_2 (after NH_3 emission)	FARM-N
	Field	0.02	kg kg^{-1} N	Proportion of N in slurry that is emitted as N_2O-N (after NH_3 emission)	IPCC, 2006
	Field	0.16	kg kg^{-1} TAN-N	Proportion of TAN-N in slurry applied to field that is emitted as NH_3-N	Hansen et al., 2008
	Field	0.395	kg kg^{-1} N	Proportion of N in slurry that is lost as NO_3^- (after NH_3 emission)	FARM-N
	Storage	0.5	years	Average storage period for manure. Assuming a single application period per year	Hutchings et al., 2012c
	Storage	0.185	kg kg^{-1} OM	Degradation rate of OM during storage of slurry in the baseline	Hutchings et al., 2012c
	Storage	0.23	kg kg^{-1} OM deg.	kg CH_4-C emitted per kg OM in slurry decomposed (46% C in OM, 50% emitted as CH_4-C)	Hutchings et al., 2012c
All Biogas	Digestate Storage	10	%	Methane yield potential that remains in the effluent under Danish conditions for centralized biogas plants for a period of	Paavola and Rintala, 2008 Angelidaki et al., 2006

			around 9 months	Sommer et al., 2000
Digestate Storage	354	L kg^{-1} OM	Average actual methane yield from literature of which 10% potential is left in digestate	Angelidaki and Ellegaard, 2003; Burton and Turner, 2003; Møller et al., 2004; Jørgensen, 2009
Digestate storage Pre-storage	38	%	Reduction of methane emission by storage cover	Sommer et al., 2000
Digestate storage Pre-storage	0.2	CH_4-C:CH_4-C+CO_2	Ratio of CH_4-C to CH_4-C+CO_2 formed during storage	Sommer et al., 2007
Field			There is no consensus on the effects of digestion on emissions of ammonia in the field therefore FARM-N estimates were used	Pain et al., 1989; Rubæk et al., 1996; Amon et al., 2006; Sommer, Jensen, Clausen, et al., 2006
Field			There is no consensus on the effects of digestion on emissions of nitrous oxide in the field therefore FARM-N estimates were used	Petersen et al., 1996; Petersen, 1999; Amon et al., 2006; Clemens et al., 2006; Bhandral et al., 2009; Thomsen et al., 2010; Chadwick et al., 2011; Mikkelsen et al., 2011
Field			Methane emissions are assumed to be negligible	Sommer et al., 1996; Wulf et al., 2002
Field	80	%	Efficiency of digestate at replacing mineral nitrogen fertilizer	Chantigny et al., 2007
Field	0.489	kg kg^{-1} N	Proportion of N in digestate that is taken up by crops (after NH_3 emission)	FARM-N
Field	0.035	kg kg^{-1} N	Proportion of N in digestate that is emitted as N_2 (after NH_3 emission)	FARM-N
Field	0.02	kg kg^{-1} N	Proportion of N in digestate that is emitted as N_2O-N (after NH_3 emission)	IPCC, 2006
Field	0.16	kg kg^{-1} TAN-N	Proportion of TAN-N in digestate applied to field that is emitted as NH_3-N	Hansen et al., 2008
Field	0.484	kg kg^{-1} N	Proportion of N in digestate that is lost as NO_3^- (after NH_3 emission)	FARM-N
Pre-storage	0.011	g C h^{-1} kg^{-1} OM	Hourly CH_4-C emission during storage	Sommer et al., 2007
Pre-storage	10	days	Time duration of pre-storage	Møller, 2012
Pre-storage	5.6	km	Transport of slurry to biogas plant after pre-storage	Al Seadi, 2000
Reactor	53.47	kg TS	Amount of total solids of co-substrate added to reactor in each digester based on kg straw added to straw scenario	Calculation
Reactor	15	days	Hydraulic retention time for Danish centralized biogas plant	Hansen et al., 2006
Reactor	61.75	%	Methane content of biogas produced	Burton and Turner, 2003
Reactor	32.75	%	Carbon dioxide content of biogas produced	Burton and Turner, 2003
Reactor	1.5	%	Other gases in biogas (H_2, H_2O, NH_3)	Burton and Turner, 2003

	Reactor			The materials are immediately fermented upon arrival to the biogas plant, degradation in receptor tanks ignored due to short retention time	Own assumption
	Reactor			The biogas production and degradation rates are calculated separately for the slurry and the co-substrates	Møller, 2012
	Reactor	297.82	L kg^{-1} OM	Methane yield of reference pig slurry	Wang et al., 2009
	Reactor	0.09	kWh m^{-3}	Electricity consumption of reactor per biogas produced	Nielsen, 2004
	Reactor			Heat consumption of reactor calculated by procedure in publication	Hamelin et al., 2010
	Reactor	3	kj kg^{-1} DM ^{o}C	Specific heat of dry matter	Hamelin et al., 2010
	Reactor	9.94	kWh m^{-3}	Heating value of methane	Hamelin et al., 2010
	Reactor			Methane leaks occur in two sections of the biogas process, from the reactor tank and from upgrading facilities, carbon dioxide that might escape at the same time is not taken into account as there is no data	Holmgren et al. 2012
	Reactor	1.6	%	Percent of methane leaking from biogas plant	Holmgren et al. 2012
	Upgrade	2.7	%	Percent of methane leaking from upgrading facilities	Holmgren et al. 2012
	Upgrade	0.25	kWh m^{-3} biogas	Electricity consumption needed to upgrade biogas by water scrubber or PSA	Petersson and Wellinger, 2009
	Upgrade	96	%	Percent of methane in the biomethane produced after upgrade	Petersson and Wellinger, 2009
Straw	Avoided straw left on field			Modeled with Daisy for 100 year period	Abrahamsen and Hansen, 2000
	Avoided straw left on field	0.028	kg N_2-N kg $^{-1}$ N	Proportion of total nitrogen emitted as N_2-N	Abrahamsen and Hansen, 2000
	Avoided straw left on field	0.025	kg N_2O-N kg^{-1} N	Proportion of total nitrogen emitted as N_2O-N	Abrahamsen and Hansen, 2000
	Avoided straw left on field	0.417	kg NO_3-N kg^{-1} N	Proportion of total nitrogen emitted as NO_3-N	Abrahamsen and Hansen, 2000
	Avoided straw left on field	0.529	kg N kg^{-1} N	Proportion of total nitrogen taken up by crop	Abrahamsen and Hansen, 2000
	Avoided straw left on field	0.973	kg CO_2-C kg^{-1} C	Proportion of total carbon emitted as CO_2-N	Abrahamsen and Hansen, 2000
	Co-substrate			Chemical characterization of wheat straw	Wang et al., 2009
	Co-substrate	0.00679	kg N kg^{-1} TS	Total nitrogen per kg of wheat straw	Wang et al., 2009
	Co-substrate	0.00082	kg TAN-N kg^{-1} TS	Total ammoniacal nitrogen per kg of wheat straw	Wang et al., 2009
	Co-substrate	0.535	kg C kg^{-1} OM	Carbon content per kg organic matter of wheat straw	Calculation

	Co-substrate	0.232	kg H_2O kg^{-1} OM	Water that degrades per kg of organic matter that degrades, calculation	Symons and Buswell, 1933
	Co-substrate	0.000297533	kg P kg^{-1} straw	Phosphorus per kg of wheat straw	Ontario Ministry of Agriculture, 2012
	Co-substrate	150	L CH_4 kg^{-1} OM	Methane yield low end of range	Angelidaki and Ellegaard, 2003
	Co-substrate	279.2	L CH_4 kg^{-1} OM	Methane yield for wheat straw with 15 day HRT	Wang et al., 2009
	Co-substrate	370	L CH_4 kg^{-1} OM	Methane yield high end of range	Torres-Castillo et al., 1995
	Digestate storage	220	L CH_4 kg^{-1} OM	Average actual methane yield of wheat straw from literature of which 10% potential is left in digestate	Hashimoto, 1983; Burton and Turner, 2003; Jørgensen, 2009
	Pre-treatment	0.007	kWh kg^{-1}	Electricity consumption of the extrusion process	Hjorth et al., 2011
	Pre-treatment	70	%	Increase in potential methane yield for barley straw, it is assumed to be the same for wheat straw	Hjorth et al., 2011
	Reactor	5	% per w/w slurry	Mass of extruded straw added to reactor	Møller, 2012
OFMSW	Avoided incineration biopulp			Emissions from biopulp are assumed to be in the same rate as for household waste	Møller et al., 2008
	Avoided incineration biopulp	0.86	kg NO_x t^{-1} waste	Avoided NO_x emission from biopulp	Møller et al., 2008
	Avoided incineration biopulp			All carbon in biopulp is released as CO_2 during incineration (avoided)	Own assumption
	Co-substrate			Chemical characterization of organic fraction of municipal solid waste (biopulp) is from KomTek's chemical analysis	Lorentzen, 2012
	Co-substrate	2.52	g N kg^{-1} w/w	Total nitrogen in biopulp per kg wet waste	Lorentzen, 2012
	Co-substrate	0.465	g TAN-N kg^{-1} w/w	Total ammoniacal nitrogen in biopulp per kg wet waste	Lorentzen, 2012
	Co-substrate	14	%	Dry matter content of biopulp	Lorentzen, 2012b
	Co-substrate	86	%	Easily degradable content per total solids	Lorentzen, 2012b
	Co-substrate	0.01	%	Reject present in the organic fraction after biopulping process	Lorentzen, 2012b
	Co-substrate	0.565	kg C kg^{-1} OM	Carbon content per kg organic matter of biopulp	Calculation
	Co-substrate	0.365	kg H_2O kg^{-1} OM	Water that degrades per kg of organic matter that degrades, calculation	Symons and Buswell, 1933
	Co-substrate	0.333	g P kg^{-1} w/w	Phosphorus per kg wet waste	Lorentzen, 2012
	Co-substrate	210	L CH_4 kg^{-1} OM	Methane yield low end of range	Davidsson et al., 2007
	Co-substrate	340.2	L CH_4 kg^{-1} OM	Methane yield for organic fraction of municipal solid waste	Davidsson et al., 2007
	Co-substrate	500	L CH_4 kg^{-1} OM	Methane yield high end of range	Luostarinen et al., 2011
	Digestate storage	466	L CH_4 kg^{-1} OM	Average actual methane yield of the OFMSW from literature of which 10% potential is left in digestate	Hashimoto, 1983; Torres-Castillo et al., 1995; Davidsson et al., 2007; Jørgensen, 2009

	Pre-treatment	25.5	kWh t^{-1} waste	Electricity consumption of the biopulping process	Lorentzen, 2012
	Pre-treatment	0.1	L t^{-1} waste	Diesel consumption of the biopulping process	Lorentzen, 2012
	Pre-treatment	0.001	L t^{-1} waste	Cleaner consumption of the biopulping process, sodium metasilicate	Lorentzen, 2012
	Pre-treatment	0.25	m^3 t^{-1} waste	Clean water consumption of the biopulping process	Lorentzen, 2012
	Production electricity due to avoided incineration	0.45	MJ kg^{-1}	Estimated lower heating value of biopulp	Møller et al., 2008
	Production electricity due to avoided incineration	2.39	MJ kg^{-1}	Estimated upper heating value of biopulp	Møller et al., 2008
	Production electricity due to avoided incineration	0.22		Lower heating efficiency value of biopulp	Møller et al., 2008
	Production electricity due to avoided incineration	0.54		Estimated lower heating value of biopulp	Møller et al., 2008
	Production electricity due to avoided incineration	0.1	MJ kg^{-1}	Electricity needed to be produced in a conventional way per kg biopulp	Møller et al., 2008
	Production electricity due to avoided incineration	1.29	MJ kg^{-1}	Heat needed to be produced in a conventional way per kg biopulp	Møller et al., 2008
	Reactor	53.47	kg TS	Same amount of TS added to reactor as straw scenario	Own assumption
Separation	Avoided application solid fraction on field			It is assumed that a field far away in need of phosphorus does not receive the solid fraction for fertilization	Own assumption Whalen et al., 2001
	Avoided application solid fraction on field	0.39	kg NH_3-N kg^{-1} TAN	Proportion of total ammoniacal nitrogen emitted as NH_3-N after application of solid fraction	Hansen et al., 2008
	Avoided application solid fraction on field	0.038	kg N_2-N kg $^{-1}$ N	Proportion of total nitrogen emitted as N_2-N after application of solid fraction	FARM-N
	Avoided application solid fraction on field	0.02	kg N_2O-N kg^{-1} N	Proportion of total nitrogen emitted as N_2O-N after application of solid fraction	IPCC, 2006
	Avoided application solid fraction on field	0.332	kg NO_3-N kg^{-1} N	Proportion of total nitrogen emitted as NO_3-N after application of solid fraction	FARM-N
	Avoided application solid fraction on field	0.435	kg N kg^{-1} N	Proportion of total nitrogen taken up by crop after application of solid fraction	FARM-N
	Avoided application			Phosphorus mineral fertilizer replaced is single	Thyø and Wenzel, 2007

	solid fraction on field			superphosphate	
	Avoided application solid fraction on field	100	%	P fertilizer replacement efficiency in all scenarios	Thyø and Wenzel, 2007
	Avoided application solid fraction on field	44	%	P content of P2O5	Calculation
	Avoided application solid fraction on field			When both N and P mineral fertilizer are applied, they are mixed, so only 1 time spreading	Own assumption
	Avoided application solid fraction on field	65	%	N fertilizer replacement efficiency in solid fraction after screw press separation compared to mineral N fertilizer	FARM-N
	Avoided application solid fraction on field	6	hours	Time between spreading and ploughing for the solid fraction	Hansen et al., 2008
	Avoided application solid fraction on field			The solid fraction is spread out by a solid manure spreader	FARM-N
	Avoided application solid fraction on field	100	km	Transportation distance solid fraction	Own assumption
	Avoided long term storage of solid fraction			It is assumed that the degradation of solid fraction is equal to slurry due to a limitation	Own assumption, limitation
	Avoided long term storage of solid fraction			Avoided long term storage of solid fraction is calculated in the same way as pre-storage of slurry	Own assumption
	Co-substrate	0.0323	kg N kg^{-1} TS	Total nitrogen per kg of solid fraction TS	FARM-N
	Co-substrate	0.0172	kg TAN-N kg^{-1} TS	Total ammoniacal nitrogen per kg of solid fraction TS	FARM-N
	Co-substrate	0.0076	kg P kg^{-1} TS	Phosphorus per kg of solid fraction TS	FARM-N
	Co-substrate	78.7	L CH_4 kg^{-1} OM	Methane yield low end of range	Menardo et al., 2011
	Co-substrate	170	L CH_4 kg^{-1} OM	Methane yield of the solid fraction	Hamelin et al., 2010
	Co-substrate	270	L CH_4 kg^{-1} OM	Methane yield low end of range	Luostarinen et al., 2011
	Digestate storage	186.25	L CH_4 kg^{-1} OM	Average actual methane yield of the solid fraction from literature of which 10% potential is left in digestate	Andara and Esteban, 1999; Møller et al., 2004, 2007; Luostarinen et al., 2011
	Pre-storage solid fraction			Calculated in the same way as pre-storage of slurry	Own assumption
	Pre-storage solid fraction	10	days	Same as slurry's pre-storage	Own assumption
	Pre-treatment	0.9	kWh/ton slurry	The energy for separation with a screw press (value is for slurry that is 2 weeks old)	Møller et al., 2002
	Pre-treatment	0.37	kg/kg slurry	Separation efficiency for ash (share of ash in solid fraction)	Hjorth et al., 2010
	Pre-treatment	0.11	kg/kg slurry	Separation efficiency for water (share of ash in solid fraction)	Hjorth et al., 2010

	Pre-treatment	0.37	kg/kg slurry	Separation efficiency for OM (share of ash in solid fraction)	Hjorth et al., 2010
	Pre-treatment	0.24	kg/kg slurry	Separation efficiency for Organic N (share of ash in solid fraction)	Hjorth et al., 2010
	Pre-treatment	0.17	kg/kg slurry	Separation efficiency for P (share of ash in solid fraction)	Hjorth et al., 2010
	Pre-treatment	0.11	kg/kg slurry	Separation efficiency for TAN (share of ash in solid fraction)	Hjorth et al., 2010
	Pre-treatment			After screw press separation, all fractions are pumped to storage facilities	Own assumption
	Production and application of P-mineral fertilizer	100	%	Efficiency of phosphorus in substrate to replace mineral P-fertilizer	Thyø and Wenzel, 2007
	Production and application of P-mineral fertilizer	0.05	kg P kg^{-1} surplus P	Proportion of P surplus lost to the aquatic environment	Nielsen and Wenzel, 2007
	Reactor	53.47	kg TS	Same amount of TS added to reactor as straw scenario but in this case it is pre-stored for a short period before reactor	Own assumption