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LCA of biogas/biofuel production from organic waste substrates under Danish and Polish conditions

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MASTER THESIS

for

Student Stefan Danielson

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LCA of biogas/biofuel production from organic waste substrates under Danish and Polish conditions

LCA for biogass/biofuel produksjon fra organiske avfallssubstrat for danske og polske forhold

Background and objective

The waste management sector is at present facing growing attention regarding environmental impacts and resource recovery and efficiency. For organic wastes the EU landfilling directive now bans the disposal of organic wastes, in order to minimise groundwater pollution and greenhouse gas emissions from landfills. As alternative, different types of organic waste have to undergo treatment, and the selection of such treatment technologies increasingly focuses on resource recovery and efficiency.

Biogas production is one of the highly recommended technologies of today, and it can be applied for different organic waste feedstock substrates, such as sewage sludge, the organic fraction of Municipal Solid Waste (MSW), industrial organic wastes, organic fats, and manure from agriculture. An end projects from biogas production is the biogas itself (incl. methane), which can be used for generation of electricity and/or heat, or it can be upgraded to biofuel, in order to substitute other energy carriers for instance in district heating or in bus transport. Another end product from biogas production is the bioresidual, which can be used without dewatering or with dewatering and composting, in agriculture as substitute for mineral fertilizer or for soil amendment or reclamation purposes in other kinds of land use. Such downstream use of biogas and bio residual has the benefit of avoided emissions and avoided environmental impacts from the production of the products they substitute.

The objective of this MSc thesis is to examine the life cycle assessment (LCA) impacts of biogas/biofuel production within a system that may treat different types of organic waste substrates. The purpose is to understand how life cycle environmental impacts of biogas/biofuel production is influenced by given mixes of organic waste substrate and technology choices, and how given critical variables and assumptions in the given systems may influence performance results, with particular focus on applications in a Danish and Polish situation.

The work is considered part of the BIOTENMARE research project at NTNU, where different research components and student projects contribute to joint new knowledge and model development.

The following tasks are to be considered:

1) Carry out a literature study relevant to the topic of this project.

1) Provide a systems definition (incl. goal and scope, system boundaries, processes and flows) of the system you are analysing, aiming at studying different cases or situations of biogas/biofuel production, compared to alternative waste treatment methods.
2) Take an LCA model, developed for the same overall system in SimaPro under a Norwegian application situation and given to you as input to your work, and decide what input data and assumptions need to be collected and provided in order to run the LCA for a typical Danish and Polish situation. Refine the model if needed, in order to fit to your work.
3) Collect information and data needed to define and describe the given technological configurations (solutions) of the system, on the basis of chosen case studies. Populate these into the model, so that it can be run to examine the environmental impacts of typical biogas production. Document own assumptions and sources for your input variables and choices,
4) Calculate the potential life cycle environmental impacts of the system, and perform a sensitivity analysis of your system.
5) Discuss the overall findings of your work, agreement with literature, what are critical variables and assumption, strengths and weaknesses of your methods, and recommendations for further work.

Within 14 days of receiving the written text on the master thesis, the candidate shall submit a research plan for his project to the department.

When the thesis is evaluated, emphasis is put on processing of the results, that they are presented in tabular and/or graphic form in a clear manner, and that they are analysed carefully.

The thesis should be formulated as a research report with summary both in English and Norwegian, conclusion, literature references, table of contents etc. During the preparation of the text, the candidate should make an effort to produce a well-structured and easily readable report. In order to ease the evaluation of the thesis, it is important that the cross-references are correct. In the making of the report, strong emphasis should be placed on both a thorough discussion of the results and an orderly presentation.

The candidate is requested to initiate and keep close contact with his/her academic supervisor(s) throughout the working period. The candidate must follow the rules and regulations of NTNU as well as passive directions given by the Department of Energy and Process Engineering.

Risk assessment of the candidate’s work shall be carried out according to the department's procedures. The risk assessment must be documented and included as part of the final report. Events related to the candidate's work adversely affecting the health, safety or security, must be documented and included as part of the final report. If the documentation on risk assessment represents a large number of pages, the full version is to be submitted electronically to the supervisor and an excerpt is included in the report.

Pursuant to “Regulations concerning the supplementary provisions to the technology study program/Master of Science” at NTNU §20, the Department reserves the permission to utilize all the results and data for teaching and research purposes as well as in future publications.

The final report is to be submitted digitally in DAIM. An executive summary of the thesis including title, student’s name, supervisor's name, year, department name, and NTNU's logo and name, shall be submitted to the department as a separate pdf file. Based on an agreement with the supervisor, the final report and other material and documents may be given to the supervisor in digital format.

— Work to be done in lab (Water power lab, Fluids engineering lab, Thermal engineering lab)
— Field work

Department of Energy and Process Engineering, 06. February 2015

Olav Bolland
Department Head

Helge Brattebo
Academic Supervisor

Abstract

Anaerobic digestion (AD) of organic waste and manure in Denmark (DK) and Poland (PL) is in expansion and it is believed to contribute considerably to reaching the EU goals on reducing CO_2 and establishing a sustainable energy system. This study analyzes the national potentials and performs lifecycle assessment (LCA) on an AD system compared to the current practice of incinerating waste and on-farm manure spreading. This mainly concerns the interconnection of energy and nutrient flows. Among 11 scenario variants for DK and for PL the highest climate impact saving was caused by PL (-1729 kg $CO_{2\ eq}$/t DM) from incineration (INC) with on-farm manure spreading compared to DK (-856 kg $CO_{2\ eq}$/t DM) mainly due to CHP replacement of more fossil rich energy. This is the general picture for PL having more CO_2 intensive energy resources and thus the choice of energy marginal to replace can be decisive. The only variants yielding net GHG for both countries is from producing biofuel for diesel substitution but without nutrient recovery (compost), and from energy intensive LBG production despite of CO_2 capture and substitution. The only scenario with opposite impacts for DK (-47 kg $CO_{2\ eq}$/t DM) and PL (119 kg $CO_{2\ eq}$/t DM) is found when 40% DM sludge undergoes AD and the VS/DM ratio for DK sludge is 50% higher than for PL. The difference is also caused by the use of energy type for biogas upgrading. Storability of biofuel is considered key for flexibility in a sustainable energy system, unlike CHP utilization. In all cases there is a trade-off between LBG and CBG benefits in terms of transportation. Utilization of bioresidual from nutrient rich organic substrates can significantly reduce emissions from producing mineral fertilizer. DK scenarios showed that dry matter content and volatile solids are decisive for maximized CH_4 production and thereby fossil replacement. Optimal substrate mixtures of manure and waste for INC combined with replacement of CO_2 intensive energy can yield high GHG savings but is limited by the on-farm GHG emissions which are considerable. CHP for both AD and INC systems can be decisive for system performance and depend on the efficiency and energy marginal replaced. Also climate change and fossil depletion is very sensitive to fugitive CH_4 emissions, while terrestrial acidification and marine ecotoxicity can be sensitive towards dry matter of biofertilizer and spreading practices. Excluding use of the liquid fraction of separated bioresidual lowers nutrient leaching potential but also the mineral fertilizer replacing potential. For instance this trade-off will rely on decision makers and the choice of weighting impact categories can contribute to choosing the environmentally most sound waste management options.

Sammendrag (Danish)

Anaerobisk nedbrydning (AD) af organisk affald og gylle i Danmark (DK) og Polen (PL) er i ekspansion og forventes at bidrage betydeligt til opnåelse af EU målsætningen om CO_2 reduktion og etablering af et bæredygtigt energisystem. Dette studie analyserer de nationale potentialer og udfører livscyklusvurdering (LCA) af et AD system sammenlignet med nuværende praksis af affaldsforbrænding (INC) og gyllespredning. Dette drejer sig hovedsageligt om forbindelsen mellem energi- og næringsstofstrømme. Blandt 11 scenario varianter for DK og for PL blev den største besparelse i klimapåvirkning fundet for PL (-1729 kg $CO_{2\ eq}$/t TS) fra INC med gyllespredning, sammenlignet med DK (-856 kg $CO_{2\ eq}$/t TS) hovedsageligt takke været fordrivelse af fossilrige brændsler med CHP. Dette er det generelle billede for PL som har mere CO_2 intensive energiressourcer og dermed kan valget af energimarginal til fordrivelse være afgørende. De eneste varianter som forårsagede netto drivhusgaspåvirkning for begge lande stemte fra produktion af biobrændstof til fordrivelse af diesel men uden genvinding af næringsstoffer (kompostering), og fra energiintensiv LBG produktion på trods af CO_2 oparbejdning. Det eneste scenario med modsatte miljøpåvirkninger for DK (-47 kg $CO_{2\ eq}$/t TS) og PL (119 kg $CO_{2\ eq}$/t TS) skyldes at 40% tørstof DK spildevandsslam til AD har et 50% større glødetab/tørstof forhold end PL slammet. Forskellen skyldes også bruget af energitype i biogas opgraderingen. Muligheden for lagring af biobrændstof betragtes som afgørende for fleksibilitet i et bæredygtigt energisystem til forskel for CHP nyttiggørelse. I alle tilfælde er der en opvejning af LBG og CBG fordele i forhold til transport. Nyttiggørelse af biorest fra næringsstofholdige substrater kan betydeligt nedsætte udledningerne fra kunstgødningsproduktion. DK scenarier viste at tørstofindhold og glødetab er afgørende for maksimal CH_4 produktion og dermed fordrivelse af fossile brændsler. Optimal substratblanding af husdyrgødning og affald for INC kombineret med fordrivelse af CO_2 intensiv energi kan resultere i store drivhusgasbesparelser men dette er begrænset af betydelige gasudledninger på farmen. CHP for både AD og INC systemer kan være afgørende for systemernes miljøprofil og afhænger af effektiviteten og energitypen som fordrives. Også påvirkninger af klimaforandring og fossil udtømning er yderst følsomme overfor CH_4 udslip, mens jordforsuring og marin økotoksicitet kan være følsomme overfor tørstofindhold og spredningspraksis. Ved at udelukke anvendelsen af vådfraktionen af separeret biorest opnås et mindre udvaskningspotentiale for næringsstoffer men også mindre potentiale til fordrivelse af kunstgødning. For eksempel denne afvejning beror på beslutningstagere og valget af vægtning af miljøpåvirkningskategorierne kan bidrage til valget af de mest miljøvenlige affaldshåndteringsmetoder.

Preface

The Nordic5Tech (Enviro5Tech) double degree master program comprises a collaboration between The Technical University of Denmark (DTU) and Norwegian University of Science and Technology (NTNU). As a part of the program study track “Residual Resources” the author is completing the final year of Environmental Engineering Master studies at NTNU, Norway. The present thesis counts for 30 ECTS and has been written at the Department of Energy and Process Engineering, NTNU, in the period ultimo January to ultimo June. Being a part of the BIOTENMARE research program ”Innovation in recycling technologies of sewage sludge and other biowaste-energy and matter recovery” the report has performed an extensive literature research too on related LCA topics which can also contribute as input for further work.

It was planned to modify the given Simapro model to include few additional process variants considered an interesting issue in an LCA perspective, at least in Denmark. This would also have required a re-definition of the functional unit. Due to limited time once the model was fully ready, unfortunately it has not been possible to do so, but the possibilities are presented and described in the Methodology chapter where the framework is set for future research work. Also, in the very end of the project period few inconsistencies in the model regarding crucial CHP electricity substitution and nutrient substitution were identified. Fortunately those were corrected in order to obtain logical and reliable results. This project work provided a good opportunity to get familiar with the SimaPro software tool and apply it for LCA modelling.

I want to thank Prof. Helge Brattebø for offering the possibility to be a part of BIOTENMARE exchanging knowledge and for good guidance during the project work. LCA results would not have been possible to generate without the contribution from colleague Simon A. Saxegård who developed the generic Simapro model as an input to the project. I also want to thank all externals who contributed with advices and material inputs during the research, including Lars Kristensen (CEO Lemvig Biogas), Lorie Hamelin (Postdoc at Syddansk Universitet), and Bodil Harder (Biogas Taskforce, Energiministeriet).

Stefan Emil Danielsson
Trondheim, June 22

Nomenclature

AD	Anaerobic digestion
ALT	Alternative scenario (Anaerobic digestion)
AS	Amine scrubbing
BG	Biogas
CBG	Compressed biogas (methane)
CH	Switzerland / Swiss
CHP	Combined heat and power
COMP	Composting
CS	Cryogenic scrubber
CSTR	Continuously stirred tank reactor
DK	Denmark / Danish
DM	Dry matter (particles of wet weight after water is evaporation, consists of VS and ash)
GHG	Greenhouse gases
GWP	Global warming potential
Hm^3	Cubic hectometre (million cubic meters)
INC	Incineration
ISO	International Standard Organization
LBG	Liquefied biogas (methane)
LCA	Life cycle assessment
LCI	Lifecycle inventory
LCIA	Lifecycle impact assessment
LCT	Lifecycle thinking
MFA	Material flow analysis
MS	Membrane separation
Nm^3	Normal cubic meter (gas volume at 0°C and 1 atm)
PL	Poland / Polish
PSA	Pressure swing absorber
RECY	Recycling
REF	Reference scenario (Incineration + conventional manure management)
TJ	Tera joule (1 million mega joule, MJ)
VS	Volatile solids (mass share of organic particles in DM)
WS	Water Scrubber
ww	Wet weight

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1 Introduction

1.1 Background

The EU Renewable Energy Directive (2009/28/EC) set a goal of supplying 20% of the European energy demand from renewables to replace fossil fuels and reduce CO_2 emissions by 2020. Denmark has committed to achieve 30% renewables while Poland must meet a 15% share in the energy system (Ministry of Economy, 2010a; Klima- og Energiministeriet, 2010). Currently Denmark is leading in fulfilling the goals while Poland benefits from having a long term higher renewable energy potential (Baum, et al., 2013). A major part of bioenergy is anticipated to be exploited from European farming of which more than 25% can originate from biogas (Holm-Nielsen, et al., 2009) and best available technologies (BAT) are to contribute to this.

Rapid biogas plant expansion creates an alternative to incineration and landfilling being the most employed treatment option for organic waste in Denmark and Poland, respectively. In comparison, biogas production has good opportunities for recovery of nutrient resources in addition to energy. One end product is biogas (including methane) which can be utilized as fuel for heat and power generation and transportation and is additionally an excellent energy storage asset (Hamelin, et al., 2014; Fruergaard & Astrup, 2011). The co-product is bioresidual which can be treated and applied as organic fertilizer in agriculture or simply as composted soil amendment. Such use of the waste treatment products has the benefit of avoiding impacts from equivalent fossil energy and commercial fertilizer substitution, which would otherwise have been produced (Lukehurst, et al., 2010).

In the light of this future waste management transitions, lifecycle assessment (LCA) can contribute to determining the environmentally most sound option in the Waste Hierarchy for decision making in a present and future context. However, the outcome can differ significantly depending on the context and region of case-specific LCA application (Cherubini & Strømman, 2011).

1.2 Objective

The objective of the present Master thesis is to understand how the environmental performance of different waste management and energy systems is affected by the choices in LCA modelling. More specifically to comparing a reference system of manure management and waste incineration with an alternative integrated anaerobic digestion system treating certain mixtures of organic wastes in the context of Denmark and Poland. For that purpose the following research questions are formulated:

- What is the environmental impact from two given organic waste treatment systems in a Danish and Polish context, and how do they cross-compare?
- Which parameters have a key influence on the environmental impacts of chosen scenarios?
- How does the choice of LCA methodology influence the results?

1.3 Scope of work

The tasks considered in order to fulfil the objective of this study are the following:

- A topic relevant literature study is carried out
- Reference and alternative waste treatment system definitions are provided aiming at studying different scenarios and variants of biogas/biofuel production in comparison
- A generic LCA model in Simapro is pre-developed and upgraded as to include country-specific elements
- Data is collected and populated to the degree allowed by the model constraints
- The lifecycle environmental impacts are calculated and tested with sensitivity analysis
- Overall findings are discussed and compared internally and externally with findings in literature including method and data quality considerations
- The report overall excludes economical considerations and the input considerations are mainly harmonized with the needs of the BIOTENMARE research project.

1.4 Report outline

This research oriented thesis combines background analysis and elements of a classical LCA report. It is formulated as a product of engineering consultancy with thorough analysis and realistic aspects. The chapters below are constituted in accordance:

Chapter 2: Presents the theoretical understanding of biodegradation and methane potentials in relations to relevant waste treatment options and its various technological configurations. The background serves for LCA modelling considerations and for the broader perspective

Chapter 3: Reviews the findings of other LCA studies similar to the topic of the present study, serving mainly as basis for comparison of the results according to the research questions, but also as a source of data. The chapter points out findings that may be interesting in future work

Chapter 4: Outlines the biogas potentials of Denmark and Poland from residuals and waste types relevant to the scope of study

Chapter 5: The legislative background based on EU law is described along with specific national legislation of Denmark and Poland in the categories of waste, energy, and agriculture. This is complemented with descriptions of current practice within organic waste and manure management.

Chapter 6: Includes the methodological approach and project scoping with model description, scenario setup, main data and assumptions in a LCA modelling perspective

Chapter 7: Includes the results of LCA modelling with uncertainty and sensitivity analysis

Chapter 8: Includes main result findings according to research questions, agreement with literature, methodology robustness considerations and outlook describing how this study can be used in future research

Chapter 9: Includes summarization of main findings and conclusive remarks

NB: The apparent length of the present report is mainly caused by presence of numerous tables and figures and because it analyzes characteristics of two countries.

2 Theory and literature study

This chapter introduces the processes of biological degradation of organic compounds in anaerobic and aerobic environments, followed by a description of feedstock and their process-wise potentials. In extension, different waste treatment technologies in engineered systems are presented, with an overview of end-product utilization. Finally, the chapter summarizes findings from scientific studies on different aspects of anaerobic treatment of organic waste.

2.1 Biodegradation

Organic matter is composed of substances containing carbon (C), originating from the remains of organisms such as plants and animals. This matter can also be regarded as organic waste. The process of biodegradation can occur in the nature or in engineered facilities. Mainly two types occur: anaerobic digestion (AD) and aerobic oxidation (composting) (Christensen, 2011).

2.1.1 Anaerobic

AD occurs when microorganism species specialized in anaerobic metabolism utilize the inherent energy sources (primarily C) and other substrate elements (vitamins, trace metals, inorganics as electron acceptors) in biomass for functioning and growing, in total absence of oxygen (O_2) as the external electron acceptor (Christensen, 2011; Schnürer & Jarvis, 2010). The two end products are bioresidual and biogas, containing mainly carbon dioxide (CO_2) and methane (CH_4) and most of the energy is bound to CH_4 (Schnürer & Jarvis, 2010). This anaerobic respiration follows a so called "structured process" displayed in Figure 1.

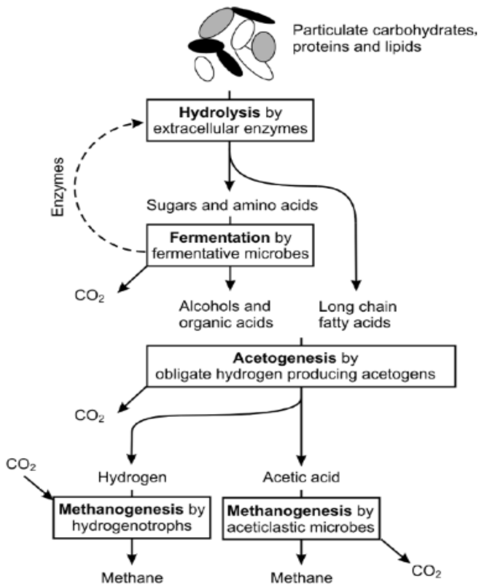

Figure 1. Pathways of stepwise anaerobic degradation of organic compounds into biogas: left (Christensen, 2011), right (Loustarinen, et al., 2011)

Stepwise metabolism happens through hydrolysis, fermentation (acidogenesis), acetogenesis, and methanogenesis. AD initiates when simple substrates in the organic waste convert to alcohols, various organic acids (VFA), hydrogen gas, and CO_2 by fermentative microbes. During fermentation amino acids release ammonium (NH_4^+). In this phase there is a considerable energy yield when producing e.g. acetate. Those microbes then produce enzymes to hydrolyse the more complex polymeric compounds (mainly carbohydrates, proteins, and lipids), enabling further uptake by other specific microbes. In an oxidative pathway acetogenic organisms convert the acids to acetate while the electrons are wasted to hydrogen ions, forming dissolved H_2.

The two end products are converted into CH_4 by respectively aceticlastic and hydrogenotrophic methanogens, typically accounting for respectively 60-70% and 30-40% CH_4 (Christensen, 2011). As in Figure 1, CO_2 is successively generated and some is reused as a (low) energy electron acceptor in the methanogenesis. In this stage the microbes require essential micro nutrients, such as Ba, Fe, Ca, Co, Mg, Mo, and Ni, as building blocks (Schnürer & Jarvis, 2010).

Sulphate (SO_4^{2-}) is a larger energy source than CO_2 (where O_2 would be the absolute largest). In case SO_4^{2-} enters the methanogenesis process, oxidation of acetate by SO_4^{2-} would be thermodynamically most preferable, enabling reduction of the SO_4^{2-} electron acceptor to form sulphide (H_2S). Large amounts of sulphate would thus result in sulphate-reducing microbes out-competing the methane producing microbes, naturally decreasing the CH_4 concentration in favour of H_2S. This compound is odorous and corrodes iron (Schnürer & Jarvis, 2010; Christensen, 2011). Therefore, avoidance of SO_4^{2-} in the AD is a two-fold advantage.

Different governing factors determine the fermentation rates and the end-products, where aceticlastic methanogenesis is one of the processes most sensitive. Under certain conditions, other microbes are decisive for the end-product distribution. Namely, at high temperatures acetate may be alternatively converted to H_2 and CO_2, while at lower, the opposite conversion path may take place. This is worth mentioning, as acetate is the crucial precursor to CH_4, and as mentioned 60-70% of the organic matter passes through acetate, while the remainder is through hydrogen and CO_2 (Christensen, 2011).

In case of excessive production of H_2 or acetate, or pH extremes, an overload can occur. Fermentation would direct to pathways forming less oxidised compounds, and proteins may form higher organic acids that would need oxidation by organic acid oxidising microbes, which are now subject to pH and hydrogen inhibition. Thereby a positive feedback (vicious circle) is created.

Another advantage during anaerobic digestion implies that certain bacterial flora is capable of transforming several cancerous xenobiotics such as PAH and LAS into harmless by-products such as CH_4, CO_2, H_2O, NH_4^+/NH_3 (Miljøstyrelsen, 1999).

2.1.2 Aerobic

Composting is a relatively simple process of microbial oxidation of carbon in aerobic conditions, in the presence of O_2, producing CO_2, H_2O, minerals and stabilized organic matter (compost). Most of the energy is released as heat (Schnürer & Jarvis, 2010). Temperatures around 25-45°C yield the highest biodiversity, 45-55°C optimum degradation rate and above this the highest sterilization rate (Christensen, 2011).

Mesophilic (25-35°C) composting initializes decomposition of easily degradable matter. Energy is released and about half of it is utilized for microbial growth while the remaining is lost as heat. The temperature accumulation creates thermophilic conditions (55-60°C), and if not controlled may exceed 70°C, benefitting pathogen sterilization at temperatures above 55°C (Christensen, 2011). Microbes not able to survive the high temperatures cause re-establishment of mesophilic conditions, where microbes start recovering to degrade the long polymers (e.g. lignin and cellulose). In the later phases the number of microbes decreases due to decrease in temperature, pH and moisture content as well as higher O_2 content caused by lower degradability and water content of the organic waste, with simultaneously decreasing nutrient availability (Christensen, 2011). Similar physical governing factors apply in composting as in anaerobic digestion.

The C/N ratio is normally 10-15 (ideally 20-35) for typical substrate mixtures. The moisture content should not be below 35-40% or optimally 55-65% to allow for water loss (Christensen, 2011). O_2 demand follows temperature increasing rapidly in the initial phase as microbes grow. It is maintained somewhat longer time and drops as the most degradable matter has been decomposed. After stabilization the O_2 demand and temperature is lowest and maturation into humus is initialized.

Maturation is mineralization of slowly degradable compounds such as lignocelluloses into humus. During growth of the present microbial consortium (bacteria, fungi, or worms) metabolites are produced which may be toxic in plant use. This phytotoxicity is eliminated during composting representing an indicator on when the process should ideally end to preserve the organic matter quality.

2.2 Methane potentials

The theoretical biomethane potential (BMP) of substrates in AD will rarely be fully utilized in practice as the CH_4 yield depends on substrate origin and composition, and on operational conditions associated to the engineered AD process (Neczaj, et al., 2013; Khalid, et al., 2011). Optimal substrate and process parameters for the AD stages are summarized in Table 3.

2.2.1 Theoretical and Practical yields

Respectively biogas and CH_4 yield from AD of organic waste can be estimated from the general stochiometrical equation in Eq. 1 under standard conditions for temperature and pressure (STP)

(Hansen, et al., 2007; Kiatkittipong, et al., 2009). STP is defined as a condition of 0°C and 1 atm and the volumetric yield can be expressed in normal cube meter [Nm^3] (Christensen, 2011).

Eq. 1. Theoretical methane yield (Hansen, et al., 2003)

$$C_nH_aO_bN_c + \left(n - \frac{a}{4} - \frac{b}{2} + \frac{3c}{4}\right) \longrightarrow \left(\frac{n}{2} + \frac{a}{8} - \frac{b}{4} - \frac{3c}{8}\right) CH_4 + \left(\frac{n}{2} - \frac{a}{8} + \frac{b}{4} + \frac{3c}{8}\right) CO_2 + cNH_4$$

$$B_{O,th} = \frac{22.4 \cdot \left(\frac{n}{2} + \frac{a}{8} - \frac{b}{4} - \frac{3c}{8}\right)}{12n + a + 16b + 14c} \quad \frac{\mathrm{STP\ l\ CH_4}}{\mathrm{g\ VS}}$$

The values in Table 1 are estimated from Eq. 1 for the single substrates fat, protein, and carbohydrate, and vary slightly from those in (Jørgensen, 2009) and (Christensen, 2011) due to use of different molecular formulas and presumably equation (excluding N as formed to NH_3 in biogas). It is observed that lipids yield the highest biogas output per VS followed by carbohydrates and protein. The particular lipids also clearly have the highest CH_4 content in the biogas even though the relative CH_4/CO_2 ratio is lower than of proteins. The ratio in carbohydrates is even lower because of the complexity of degradation.

Table 1. Theoretical biogas and methane potentials from three substrate components (Carlsson & Uldal, 2009)

Substrate	Biogas (Nm^3/kg VS)	Methane (Nm^3/kg VS)	Methane rate (%)
Lipid	1.37	0.96	70
Protein	0.64	0.51	80
Carbohydrate (cellulose)	0.84	0.42	50

Laboratory batch experiments are necessary to determine practical biogas yields expected in engineered systems (Kumar, 2011; Angelidaki & Ellegaard, 2003; Christensen, 2011). The practical biogas yield will always be lower than the theoretical ones, usually achieving up to 85-95% (30-60% in highly particulate matter) due to a range of factors (Christensen, 2011):

- 5-10% of substrates used for bacterial growth
- 5-10% of organic mass lost in the effluent (bioresidual)
- Lignin is not degraded anaerobically
- Organic matter inaccessible as in bound structure

Khalid, et al. (2011) has reported a list of methane yields for different combinations of organic co-substrates, suggesting that highly lipid substrates can increase the overall efficiency. Sole manure treatment can provide a methane yield of 10-20 m^3 CH_4/t manure treated and AD is profitable when the biogas yield is higher than 30 m^3/m^3 biomass (about 20 m^3 CH_4/m^3 biomass), and can be realised when feeding in substrates with relatively higher CH_4 potential (Angelidaki & Ellegaard, 2003). As industrial organic waste in Denmark is limited, the organic fraction of MSW (e.g. sludge, food and garden waste) has become an attractive co-substrate (Hjort-Gregersen & Petersen, 2011).

2.2.2 Substrate properties

Feedstock or substrate refers to the organic waste types suitable for AD. Those have different inherent physical-chemical characteristics e.g. with respect to water content, dry matter, organic matter, and degradability of organic matter. Substrate composition is depicted in Figure 2.

The total wet weight is divided in a solid and a liquid phase. The liquid phase consists of dissolved compounds such as trace elements and NH_4-N and, of which the form depends on pH. Apart from an ash content in which heavy metals can be bound, the solid phase (dry matter) includes mainly macro nutrients in the form of bound C, N, and P (Deublein & Steinhauser, 2008).

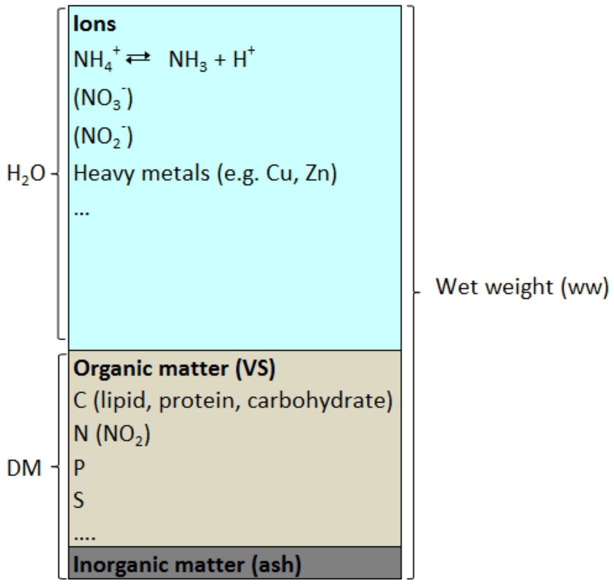

Figure 2. Conceptual illustration of an organic substrate profile (typically livestock manure) with a particulate and a soluble phase. Based on information from (Christensen, 2011), (Hamelin, et al., 2010),

Dry matter: Dry matter (DM) is defined as the remaining compound (solid fraction) after evaporating the water content (liquid fraction) from wet waste at 80°C for 24h. DM consists of organic and inorganic matter which is bound in respectively volatile solids and fixed solids (ash) (Hamelin, et al., 2010). Higher levels of heavy metal and organic contaminants may inhibit degradation (Schnürer & Jarvis, 2010).

Volatile solids: Volatile solid (VS) is the organic matter fraction that volatilizes when DM is heated to 550°C for 1h. It comprises easily (VS_{ED}) and slowly (VS_{SD}) degradable organic compounds. Among the different components constituting VS (lipid, protein, volatile fatty acids, and carbohydrates), only carbohydrates as crude fibre (lignin) belong to VS_{SD} as recalcitrant to microbial hydrolysis (Hamelin, et al., 2010). Hence lignin and cellulose are the limiting factor. Thereby the biomethane potential (BMP) increases as function of VS_{ED} content which further relies on DM content per substrate wet weight. Degradability values of certain compounds are displayed in Table 2.

C/N ratio: has an optimal value of 20-30 for most wastes (35-40 for high lignin content) for bacterial growth (Christensen, 2011). Too high C/N retards degradation while too low C/N inhibits methanogens. Therefore optimum can be obtained by mixing different co-substrates (Khalid, et al., 2011). Single substrates have wide ranges of C/N ratios, e.g. garden waste (10-20), kitchen waste (15-23), animal manure (15-25), and sewage sludge (5-15) (Christensen, 2011).

Table 2. Degradability of compounds found in pig manure. Retrieved from Jørgensen (2009b)

Compound	Degradability (%)
Organic compound (VS)	60
Protein	47
Lipids, fats	69
Hemicelluloses	65
Celluloses	69
Starch	94

2.2.3 AD process parameters

Several measures can be taken to obtain maximized CH_4 output and minimized content of organic matter in the bioresidual effluent, i.e. increased degradation of organic matter and mineralization of nutrients in AD. These strategies depend on the biogas plant configuration (Frandsen, et al., 2011) and establishment of optimal conditions by several process parameters (Khalid, et al., 2011).

Pretreatment: Different types of physical, chemical, biological and thermal pretreatment are found to enhance biogas yield (Esposito, et al., 2012b). These are e.g. mechanical size reduction (screening), solid-liquid separation (dewatering), alkaline addition, thermal-pretreatment (pasteurization). The mechanical can also sort out unwanted impurities or make organic matter more easily accessible for microbes (Schnürer & Jarvis, 2010).

Mixing: Gentle stirring by agitator or pump enables stable temperature, prevents foaming from sedimentation and facilitates contact between substrates and microbial community to obtain nutrients. Co-substrates should be mixed to a homogenous feedstock prior to entering the digester to provide more stable biomass growth (Lindmark, et al., 2014).

pH value: CH_4 forms optimally between pH 5.5-8.5. Acids released during acidogenesis to lower level may inhibit the microbes. The CO_2 fraction will decline with increasing pH while CH_4 rises. The limiting factor for this is increasing generation of NH_3 (e.g. from slurry) inhibiting the microbes (Christensen, 2011). Higher alkalinity (basic substances) enables greater buffer capacity thus stable pH (Schnürer & Jarvis, 2010).

Temperature: AD occurs under mesophilic (30-42°C) or thermophilic (43-55°C) conditions (Al Seadi, et al., 2008). The optimal temperature may vary with feedstock and digester configuration, but should usually be maintained fairly constant and adjusted to yield trend by operator (Schnürer & Jarvis, 2010).

Retention: Hydraulic retention time (HRT) has a direct link with substrate properties (degradability) and process temperature (Schnürer & Jarvis, 2010). Mesophilic stage typically requires 30-40 days while for thermophilic it is 15-20 days (Al Seadi, et al., 2008).

Loading: Decomposition of organic matter stagnates if no material is continuously added. The organic load rate (OLR) indicates how much raw feedstock should be added by time based on the DM and VS content of substrates, given specific process parameters and plant configuration. OLR must be stable and input homogeneity should be maintained. Stabilized ORL for mesophilic and thermophilic reactors are respectively 2-3 and 4-5 kg VS/m^3 reactor (Schnürer & Jarvis, 2010).

Table 3 displays a summary of the optimal conditions during the initial AD process, and during the methane formation in AD digesters.

Table 3. Optimal ambient conditions in two stage AD. Modified from (Deublein & Steinhauser, 2008).

Parameter	Hydrolysis/acidogenesis	Methanogenesis
Temperature	25-30°C	Mesophilic: 32-42°C Thermophilic: 50-58°C
pH value	5.2-6.3	6.7-7.5
C/N ratio	10-45	20-30
DM content	<40% DM	<30% DM
Required C/N/P/S ratio	500/15/5/3	600/15/5/3
Trace elements	No special requirements	Essential: Ni, Co, Mo, Se

2.2.4 Synergistic effects

Recent research finds co-digestion to improve biogas and CH_4 yield of single substrates, as complementary characteristics can enable more optimal conditions (Khalid, et al., 2011; Nielfa, et al., 2015b). Synergistic/antagonistic effect in the final production is calculated from BMP tests on mixtures by dividing experimental CH_4 production by theoretical production from each co-substrate in mixture summed (Nielfa, et al., 2015b)

Biogas yield is assumed to be 10% higher for co-digestion in a full-scale digester compared to calculated values of single feedstocks (Pöschl, et al., 2010). Sludge co-digested with spent grain, manure, and grease yielded respectively synergy factors of 1.65, 1.1, and 1.31, while e.g. manure and grease obtained 0.9 (Nielfa, et al., 2015a). This boost is also observed by Neczaj, et al. (2013). Slaughterhouse paunch rich on carbohydrates also appeared to improve degradation of mixed fatty acids (Astals, et al., 2014).

Higher specific yield (m^3 CH_4/t VS) is achieved from co-digesting MSW with vegetable oil (686) and animal fat (490), and cattle manure with oil (450) (Esposito, et al., 2012b). AD of up to 42% VS cattle manure with sewage sludge yielded maximum CH_4 with a twofold biogas production compared to single AD (Hasan, 2014). 2:1 VS ratio of pig manure and sludge increased CH_4 by 82.4% per VSadded compared to sludge alone (Zhang, et al., 2014). Raising fruit/vegetable and restaurant waste share increased CH_4 considerably compared to sole sludge due to higher VS content (Cabbai, et al., 2013). Slaughter waste with MSW yielded twofold more biogas than slaughter alone (Cuetos, et al., 2008). Sewage sludge with household waste yielded more CH_4/t VS compared to separate digestion (la Cour Jansen, et al., 2004). Cumulative biogas yields from mixed sludge, household waste, and co-digestion are 181 L, 228 L, and 232 L (Sosnowski, et al., 2008). Several studies are also compiled in Pawlowski, et al. (2013).

3 Treatment technologies

In the following the most common configurations of three major waste treatment technologies are outlined. Incineration and AD technologies are elaborated on as being of substantial relevance to this study including the modelling, focusing on energy and environmental aspects. Information is generally based on (Christensen, 2011), unless otherwise cited.

3.1 Anaerobic digestion (AD)

AD treatment of organic waste can take place in engineered systems to produce marketable biogas and bioresidual. It can be employed at large centralized plants and smaller farmland plants. The technology is becoming widespread and the choice of configuration will have implications for the outcome of the treatment process. The choice can depend on local conditions and end-use strategy (Frandsen, et al., 2011). Biogas plant configurations and associated technologies are briefly described.

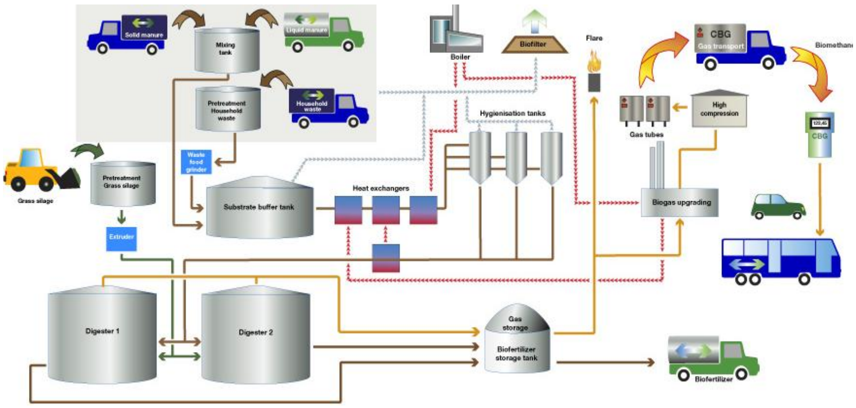

Figure 3. An example of a typical co-digestion AD facility. Retrieved form (Purac, 2014)

A typical biogas plant (Jørgensen, 2009b) looks like in Figure 3. It includes reception tanks for dry (> 25% DM) and wet (< 10% DM) substrates (Christensen, 2011). Some types enter AD directly or after mechanical pretreatment while specific substrates and animal byproducts undergo thermal pathogen sanitation (70°C/1h). This can be before, during or after AD. The feedstock is optionally pumped to buffer tanks for homogenization, and further to agitated and entirely sealed reactor tanks. The thermal energy surplus from treated biomass is transferred to incoming biomass by heat-exchangers. Fresh bioresidual is pumped to a covered secondary storage reactor to recover residual biogas (10-30%) (Luostarinen, et al., 2011) and prevent NH_3 loss before distribution to farmland. It may be dewatered with liquid recirculation or post-treated e.g. composting. The collected biogas is biologically purified from H_2S and pumped to upgrading facility or directly to a sealed storage where flaring regulates the pressure.

Mesophilic (37°C) or thermophilic (55°C) AD processes can both treat co-substrates that include potentially pathogenic biomass. The latter is becoming attractive due to a range of advantages

(Christensen, 2011). Mesophilic plants must enable external facilities post-sanitation of the minimum prescribed 70°C/1h (EC, 2011b), while sanitation is often integrated in thermophilic reactors (Angelidaki & Ellegaard, 2003). Thermophilic gas production is more efficient but operates temperature sensitively. Energy recovery between AD preparation and hygienization are virtually comparable for the two configurations though heat-exchangers (Schnürer & Jarvis, 2010).

Different technical configurations are based on a one-stage digestion model as the most common, where all AD stages happen simultaneously in the same place (Schnürer & Jarvis, 2010):

3.1.1 Single stage batch

All (dry) feedstock is digested at once and is isolated for a long time until digested and removed. It is usual to have parallel batches for the phases of filling, treating, and emptying (Luostarinen, et al., 2011). The organisms have sufficient time to metabolize the organic matter and do not get washed out of the system. The digestion degree is generally higher than in continuous flow and theoretically 100% CH_4 content can be extracted, however high digestion rates may be hard to achieve for very dry feedstock.

3.1.2 Single stage continuous flow

Raw substrate is continuously added to digester enabling a smooth biogas production. Wet feedstock such as sewage sludge and slurry can also undergo the process. Dry process material is added in larger pulses less frequently but dilution and stirring enables pumping for continuous feeding and outflow, providing uniform and diverse supply for the microbes and reducing the risk of overload. Thus the ambient conditions are maintained and microbes remain acclimatized.

3.1.3 Continuously stirred tank reactor (CSTR)

CSTR is one of the most common large-scale configurations (Figure 3) and resembles the principle of Single stage continuous flow where the substrates are completely mixed. The reactor is continuously emptied and sometimes the bioresidual or process liquids are recycled to the process, increasing the retention time and obtaining higher biogas yield.

Pretreatment: Pre-separation is a commonly associated to biogas plants (Esposito, et al., 2012b; Frandsen, et al., 2011). Wet wastes e.g. sewage sludge can be mechanically separated at source with the purpose of using the solid fraction for AD and to minimise transportation needs. On-farm mechanical phase separation of liquid manure may resolve problems connected with P excess. Most of P will end up in the solid fraction delivered to AD while the farm can utilize the liquid fraction rich on N. This allows for redistribution of bioresidual according to need (Luostarinen, et al., 2011).

Post-treatment: The bioresidual can also be mechanically phase separated and the efficiency depends on technology and material properties. For instance sedimentation will retain most organic

N and P in solid fraction while leaving soluble nutrients in liquid fraction. Particle sieving is another option. Centrifugation (decanter) is reported as most efficient for P and solids separation and can be enhanced by polymer addition. The solid fraction may be further dewatered and post-composted into e.g. a culture medium or dried and pressed into P rich fertilizer pellets. The liquid fraction may be cleaned from NH_3 by stripping or membrane separation.

3.1.4 Application of end-products

Two end-products from the AD are created: biogas and bioresidual. The biogas can be applied for commercial use within different energy sectors with or without upgrading (see 3.1.6 Biogas upgrading). The most common are described (Deublein & Steinhauser, 2008).

CHP energy: After purification from trace contaminants the raw biogas can be combusted on site in CHP gas turbines to generate energy partially for internal use and if possible for distribution on national electricity and local district heating grids (Poeschl, et al., 2012b). Alternatively, the biogas can be distributed by low-calorific pipelines to the nearest CHP plant (Rehl & Müller, 2013).

Natural gas network: After upgrading the raw biogas to standard natural gas quality the biomethane can be pumped into the high-quality natural gas pipeline. In locations without a natural gas network the biomethane can be compressed in containers and transported to the location of injection into the natural gas grid for final use in households or process industry (Ministry of Economy, 2010b).

Vehicle fuel: The procedure is similar to the above as quality requirements are the same. Both light and heavy duty vehicles can drive on compressed biogas (CBG) or liquefied biogas (LBG) upgraded on site. If there is no natural gas pipeline near the biogas plant the biomethane can be stored and distributed to gas stations as illustrated in Figure 3. It is also possible to mix the biomethane with fossil vehicle fuels as partial substitution (Rehl & Müller, 2013).

The bioresidual can be applied for nutrient enrichment in several ways, with or without significant utilization depending on its prescribed quality requirements. Common uses are (Christensen, 2011; Visvanathan, 2014):

- Direct crop fertilizing: Fresh bioresidual in wet or dry condition can be spread on agricultural land equivalently to conventional manure management.
- Commercial use or soil amendment: The most common bioresidual management is composting. After dewatering it can be used as soil amendment in agriculture if fulfilling the quality requirements. If lower quality, the compost can be used as growth medium in e.g. public areas.

- Incineration: As a prime disposal route incineration is possible for recovering the remaining energy content from bioresidual which has not been fully transformed into CH_4 during AD. Alternatively it is incinerated due to inadequate quality for compost production.

3.1.5 Environmental aspects

Biogas as a renewable resource and bioresidual as a highly valuable bio fertilizer makes integrated AD offer several benefits beyond pathogen and odour reduction. Among these are (Lukehurst, Frost, & Al Seadi, 2010; Holm-Nielsen, Al Seadi, & Popiel-Oleskowitz, 2009; Tybirk & Jensen, 2013):

- Reduced fossil fuel use
- Reduced mineral fertilisers use
- Reduced GHG emissions from open manure stores
- Improved fertilization efficiency
- Closes the production cycle

However, there have been found significant contributors to environmental impacts from biogas production, primarily as CH_4 leakage from processes and N_2O emissions from bioresidual application, but partly also the internal energy consumption, sometimes also in comparison with other utilization technologies. Figure 13 also illustrates emissions in the manure based biogas production chain.

3.1.6 Biogas upgrading

Commercial upgrading technologies separate the CO_2 content from raw biogas to obtain high CH_4 concentration. ISO/DIS 15403-1:2006 standard prescribes natural gas quality of > 96% CH_4 content in the purified gas (Deublein & Steinhauser, 2008). Some technologies require pretreatment of impurities as H_2S (corrosive) and N2 while others have inbuilt cleaning. All technologies provide delivery as CBG and the cryogenic process in addition as LBG with a higher energy density than CBG. The descriptions are based on (Bauer, et al., 2013). Niesner, et al. (2013) presents the concepts of different biogas upgrading technologies:

Water scrubbing (WS): Raw biogas is introduced from below a water filled absorption column where CO_2 sorbs due to the very high solubility compared to CH_4. CH_4 leaves at top while saturated water is let out at bottom and either regenerated with CO_2 released to atmosphere or only used once and discharged with the CO_2 to a WWTP. Any CH_4 dissolved in water is captured and recycled in absorption column. Operating pressure is 6-12 bar, and CH_4 yield reaches 94% with 98% purity.

Amine scrubbing (AS): Operates principally as WS (1 atm), only using solvent with much higher CO_2 sorption capacity. This is typically mono- or diethanolamine. The solvent regeneration however is more heat and energy demanding. Operating pressure is 1 atm and CH_4 yield reaches 90% with 99% purity.

Pressure swing adsorption (PSA): CO_2 has smaller molecules than CH_4 and thus can be retained when flowing through columns packed with proper adsorbent material e.g. molecular sieves, zeolites, and activated carbon. The efficiency depends on material, temperature, and pressure, where the pressure is variable. During depressurization the captured CO_2 is released to atmosphere while CH_4 leaves the column in a biomethane stream. Operating pressure is 4-10 bar, and CH_4 yield reaches 91% with 98% purity.

Membrane separation (MS): Hollow fibres, silicone rubber or polyamide membranes allow different compounds to pass through. The high pressure type employs gas flow in both permeating CO_2 and retained CH_4, while the low pressure type employs gas flow on the retentate side and liquid on the permeate side. Operating pressures are 20-36 bar and 1 atm, respectively, and CH_4 yield reaches 78% with 90-97% purity. However, mandatory multistage systems accompanied by PSA can achieve a 99.5% CH_4 yield with 99% purity.

Cryogenic separation (CS): Separates most unwanted gasses (except N2) by cooling the biogas stream until their condensation point being 78 °C for CO_2 compared to -161°C for CH_4 (Bauer, et al., 2013). Compression can additionally raise the boiling point. Liquid N2 can further be used to condensate upgraded CH_4 into LBG. The CO_2 Wash® process introduces raw biogas up through a column. Its CO_2 content is condensed at top and released to dissolve impurities. Effluent and CH_4 leave next to > 80% of the remaining liquid CO_2 of marketable quality (Acrion, 2011). This step is usually combined with other processes to produce CBG or LBG. CS is however currently not viable at large-scale due to the high energy consumption (Deublein & Steinhauser, 2008).

Methane Gas for Storage of Renewable Energy (MeGa-stoRE): A novel technology is being tested. The concept is to store energy in H_2 from electrolysis with wind or solar power. The Sabatier process (9 bars and 275 °C) upgrades biogases to natural gas quality letting H_2 react with all CO_2. Up to 50% more CH_4 is created compared to conventional biogas upgrading technologies that remove all CO_2 (Godske, 2014; Lemvig, 2014).

3.2 Composting technologies

Numerous configurations of composting technologies exist. The typical ones are presented below (Christensen, 2011). The curing time depends on the waste mix composition and applied technology.

3.2.1 Open composting

- Windrows are elongated feedstock piles used for complete composting or just stabilization. Those are typically turned by machines for mixing but the oxygen supply happens to a greater extent by natural aeration. Retention time is 12-20 weeks for biowaste.
- Static piles provide no agitation implying need for adding bulking waste for structure. It is widely used for treating mixtures containing sewage sludge where the piles are covered with already matured compost to prevent heat loss. Some facilities may have passive aeration pipes in the base layer. Retention time is 3 weeks prior to 6-8 weeks of windrow maturing. In other types the feedstock is distributed in composting cells and moved around to homogenize temperature and moisture.

3.2.2 Enclosed composting

- Channel composting happens in a hall with feedstock stacked in uncovered piles, often divided by walls serving as a track for turning machines or a feed-in conveyor belt for compost material. Active aeration systems below channels and turning are used to supply air, and aeration and water addition composting processes are controlled for each channel. The compost moves along the channel during turning and the composting phases may last 6-8 weeks, or 3 weeks with a 6 month maturing in static piles.
- Aerated pile composting with turning machines happens inside a building with the feedstock placed in one large pile to be turned alongside the hall length. Forced aeration system is also installed underneath the compost bed, and usually water is added during the automatic turning, collecting the off-gasses for treatment in a biofilter. Retention time is of 4-12 weeks depending on the turning frequency and thereby the moisture gradient.
- Brikollari is prepared by amending biowaste with bulking agent and compressing it into blocks with surface channels for natural aeration. Compressing requires electrical energy but saves space. The block stacks are conveyed to a high-rack warehouse with several ventilation areas, for efficient degradation and stabilization during 5-6 weeks. The stabilized compost (20% moist) can be marketed after grinding, or further cured in windrows for 8-10 weeks.

3.2.3 Reactor composting

- Tunnel/box composting has different levels of process controls and is widely used to compost MSW, sewage sludge, and manure. These maintain homogeneous temperature and moisture in the spacing between the compost owing to recycled exhaust gases, thus rarely needing turning. Fresh air or recycled gases are injected from below the tunnel reactors and irrigated from above, and compost is fed or removed through end hatches with conveyors. Turning for longer retention can happen within or between tunnels. Usual retention time is 1-7 weeks.

- Rotating drum composting is a widespread, dynamic treatment of especially MSW. The biomass is aerated by rotation mixing in a "ball mill" or an aerated fan, and the drier and homogenized compost output enables more efficient reject removal. Slow rotation ensures no compaction of wet feedstocks. Retention time is 1-10 days, where longer retention ensures high-rate degradation, however with a necessary windrow composting lasting 2-3 months.

3.2.4 Application

In EU compost is mostly used in the agricultural sector as soil fertilizer (see Table 1). Compost low in nutrients (e.g. yard compost) is well suitable for non-agricultural applications. The bioresidual from AD may contain abundant NH_3-N harmful to young roots, why this can only be applied in non-agriculture if previously composted.

When manufacturing top soils for landscaping, cured compost is commonly refined by large screeners and blended with growth media to reach desired physical characteristics. For general (non-agricultural) use, plant nutrients and heavy metals along with biodegradability is normally requested declared, as the purpose is to produce quality vegetation by slow nutrient release from cured compost rather than maximizing harvest. Compost is required well matured prior to application as to fully provide the soil with nutrients.

When using compost for backfilling larger plants, the excavated soil is mixed with compost in a 1 (nutrient-poor) to 2-3 parts soil ratio or 1 (nutrient-rich) to 4-6 parts soil, or simply applying a thickness layer of compost around the plant, depending on quality. Once compost has been mixed into the soil, dissolved plant nutrients and salinity may not be excessive for damaging plant roots. Therefore compost with lower soluble nutrient content from yard waste or sewage sludge (although often high on P content) are more adequate for landscape application compared to nutrient-rich compost from substrates such as kitchen waste and manures.

Table 4: Sectors and distribution of utilized compost within the European Union by volume (Christensen, 2011)

Application	EU average (approx.)	Individual EU countries
Agriculture and field horticulture	40	10-70
Landscaping, reclamation, manufactured top soils	30	20-60
Residential/private gardens	20	10-50
Others (greenhouses, nurseries, landfill cover, etc)	10	5-20

3.2.5 Environmental aspect

N losses occur during composting but are hard to quantify and data is very limited. Since most of the N in biowaste is organic, mineralization releases N as dissolved organic N and NH_4^+ and the amount not used for microbial growth easily dissociates to NH_3 and H+ during composting. The release of NH_3 depends on N in waste, degradability, temperature and pH, and amount of gas released through the compost as ventilation enhances H+ (acid) formation. Highly relying on the degradability and moisture content, microbes oxidize NH_4^+ to NO_3^- and intermediates appear as NO_2^- (dissolved) and N_2O (gas). Most pollution in general is found in leachate and if generated, N is found in high concentrations as organic, NH_4^+, or NO_3^- bound. If not appropriately managed compounds may percolate to environment, contributing to global warming (N_2O) and soil acidification (NO_2^- and NO_3^-).

The very potent greenhouse gasses CH_4 and N_2O are generated in less aerated pockets and the formation depends on its characteristics and structure as well as technological feature of facility. It is unknown how much CH_4 is oxidized prior to release and sometimes even biofilters may be a source of e.g. N_2O when NH_3 volatilizes (Christensen, 2011). Complete oxidation is imperative in terms of avoiding negative effects on plant growth from marketed compost.

3.3 Incineration

Today municipal solid waste incineration (MSWI) combustion plants can include energy recovery as electricity and/or heat supported by air pollution control (APC) systems and ability to replace fossil fuel consumption (Christensen, 2011). A typical incineration process is sketched in Figure 4.

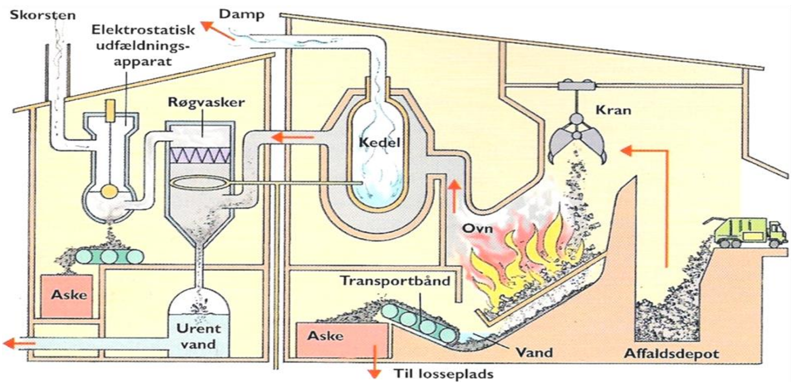

Figure 4. Cross section of a common incineration plant with a moving grate furnace

Waste undergoes thermal conversion (850-1200°C for 2s) to ensure complete combustion under mixing and optimal surplus air supply. The flue gas heats the boiler to create high-pressure steam expanding in the steam turbine to run an electrical power generator. In case of a CHP plant configuration, the steam recirculates and the residual heat is regained in a heat-exchanger connected to a district heating grid. Flue gas properties now enable optimal cleaning conditions. Incineration of mixed wastes generates solid residues mainly in the form of bottom ash (150-300 kg/t) and flue gas derived fly ash (10-30 kg/t) (Christensen, 2011). Fly ash after the chemical air pollution control (APC) treatment can be collected with e.g. simple fabric bags or high-efficiency electrostatic precipitators. Cleaned flue gas is emitted to atmosphere and fly ash is usually landfilled as hazardous. Also residuals emerge from cleaning the acidic gas using basic chemicals, where e.g. gypsum is commercialized and wastewater is discharged to WWTP.

Three key parameters characterizing waste for optimal operation are: heating value, ash, and moist content. A certain combination of these (from Tanner's diagram) will not require auxiliary fuel input. Depending on the furnace design the nominal heating value may typically be 10-12 MJ/kg, or lower in case of sole household mixed waste treatment with an allowable variation of 8.5-14.5 MJ/kg (Christensen, 2011). When accounting for waste calorific values/thermal input, and energy efficiencies in Europe the lower heating value (LHV) is commonly applied. As opposed to the higher heating value (HHV), LHV indicates the net energy content released upon complete combustion as the water content is transformed from its liquid to its evaporated state (Christensen, 2011).

Several incineration plant configurations exist today. The most common commercially available technologies are characterized by mass burning of heterogeneous waste, with limited or no physical pre-treatment, or of homogenous waste after sorting, screening and/or shredding, if appropriate.

3.3.1 Moving grate

This technology is widely employed for receiving mixtures of MSW. The grate systems can have different designs. The main advantage of moving grate is the large capacity and ability to allow for greater variations in waste composition and heating values, and air supply from below. As illustrated in Figure 4 this type ensures optimal retention time of the waste fed into the furnace as continuously conveyed into the furnace, allowing for new incoming mass. Once dried and combusted a solid residue (including incombustibles) partly crumbles through the grate to a bunker and conveyed outdoor as bottom ash (slag), containing minor amounts of fine and/or molten inorganic particles.

3.3.2 Fluidized bed (FB)

FB has limitations with respect to heterogeneity, composition, and demanding pre-treatment requirements of input waste. However, energy and environmental performance can reach overall higher efficiencies. Fuel and solid waste are fluidized by vertical air injection through a granular bed layer in a chamber. These are suspended within the bed improving chemical reactions and heat transfer allowing for lower air surplus and thus a thermal efficiency up to 90% (Christensen, 2011). NO_x formation is limited owing to lower temperatures (around 850°C) and longer residence time in the upward flow of flue gas and particles. The cleaning and energy recovery principles are similar to the moving grate. Circulating fluidized bed (CFB) is FB upgraded with an aggregate cyclone enabling recirculation of the flue gas where the unburned components flow back into combustion while the gas is simultaneously cleaned. The FB technologies are advantageous for very wet wastes such as municipal sewage sludge, household organic waste (Li, et al., 2014), animal byproducts (Miljøministeriet, 2004), and for that matter also animal manure (Hjort-Gregersen & Petersen, 2011). In order to ensure optimal temperature these low calorific wastes would require air preheating and exclude energy exploitation from the chamber (Christensen, 2011). Despite of managing very wet waste, Li, et al. (2014) found that it can reach a combustion efficiency of over 98% while the overall practical efficiency of sewage sludge incineration including losses reaches almost 75%.

3.3.3 Energy aspect

Different boiler types enable various energy recovery options (Table 5). The choice of technology relies on market conditions and infrastructure availability. CHP offers maximal energy utilization under given conditions, however implying a trade-off between power and heat recovery when the plant is strategically designed. Downstream flue gas condensation may additionally recover 10% of the energy input (Christensen, 2011).

Table 5. Energy conversion technologies employed and belonging maximal efficiencies (%). Retrieved from (Christensen, 2011)

Energy utilization	Recovery		Overall efficiency a)
Heat only	Heat	75-90 (100 b))	75-90 (> 100 b))
Steam only	Steam	75-90	75-90
Power only	Power	25-35	25-35
Combined steam and power	Steam Power	60-75 15-20	75-90
Combined heat and power (CHP)	Heat Power	60-65 (85 b)) 20-27	80-92 (> 100 b))
a) Useful energy output from boiler relative to the LHV of the waste; b With flue gas condensation			

3.3.4 Environmental aspect

There is a risk of environmental pollution locally and globally. APC minimizes emissions to air which may cause e.g. acid rain. CO_2 from organic waste is regarded biogenic thereby formally not contributing to global warming (Christensen, 2011). Ashes rich on heavy metals are usually disposed of in controlled inorganic landfills. After neutralizing flue gas acids the APC residuals from mixed MSW may undergo chemical stabilization to provide acceptable environmental profile (Christensen, 2011). Quina et al. (2008) reviews multiple applications of MSWI residues that can be used as filler in asphalt or e.g. replace 20% cement in energy intensive concrete (Quina, et al., 2008). Bottom ashes are usually stabilized for use as road base, or concrete aggregates with desired physical properties. The recycling rate of bottom ash in Denmark is above 90% (Christensen, 2011).

Damgaard et al. (2010) studied air emission profile from certain MSWI applying eight APC technologies ranging from no treatment to the best available technology (BAT). Key air pollutants were reduced a factor 100-1000 compared to no treatment applied. Downstream measures other than end-of-pipe should be taken for meeting strict requirements and recommends to holistically consider environmental and energy efficiency issues occurring from solid residue, air and wastewater emissions, fossil fuel savings, and consumption of energy and chemicals (Damgaard, et al., 2010).

4 Literature study

This chapter thoroughly comprises findings from scientific literature. Most studies are limited to a Scandinavian (particularly Danish) context, with very few other European and international studies. Sections are divided by lifecycle studies focusing on methodologies, co-substrates, treatment systems, and technologies of pretreatment and post-treatment. Presented are also other studies relevant for the scope of this research project.

4.1 LCA methodologies

Cherubini & Strømman (2011) analyzed recent studies on various bioenergy systems, including biomethane. Many are in a European context and perform LCA or energy and/or GHG balances with different site-specific input data (e.g. feedstocks, conversion, end-use, age and source) and methodological assumptions (e.g. functional unit1, allocation method, reference system, system boundaries). Difference in these factors bias lifecycle results, creates uncertainties and complicates study comparisons (Cherubini & Jungmeier, 2010) as also stated by Clavreul, et al. (2012). Some studies apply residues and all biomethane studies assessed transportation biofuel provision rather than stationary energy production. Most found that biomass based electricity makes up 5-10% of the net GHG energy compared to fossil based, being even lower if biomass is produced with low energy input, derived from residues, converted efficiently (CHP), or inversely, if the fossil fuel reference is carbon-intensive (unlike renewable-based). Key parameters such as indirect effects are strongly dependent on the context (Cherubini & Strømman, 2011). N-based soil emissions (e.g. N_2O) induced by fertilizers have a high data uncertainty and are hard to control (Meyer-Aurich, et al., 2012; Amon, et al., 2006) and may be decisive for the GHG balance of some biomass (Crutzen, et al., 2007). That assumptions regarding energy production are often decisive for the LCA outcome is also stated by inter alia by Ekvall (1999), and regarding marginal energy production by Mathiesen (2008). Several ways to identify marginals and how they react to a market change have been reported in (Hamelin, 2013c) discussing how to apply realistic and long-term marginals in LCA.

Clavreul, et al. (2012) argues that overall results of LCA modelling of waste management systems can be significantly affected by uncertainties associated with model, scenarios, parameters, and data gathered from different sources. They also state that results suffer from process specific data uncertainties. Solutions to identify those thoroughly have been suggested and methodological awareness during conduction is required. Rehl & Müller (2012) state that many LCA studies avoid a strict differentiation between attributional and consequential methodology, and performing LCA on energy generation from biogas reveals considerable results differences when applying the different methodologies.

[1] According to Poeschl *et al.* (2012), a wide range of LCA apply feedstock mass processed as the funcitonal unit

4.2 LCA of co-substrates

Hamelin, et al. (2014) studied 1 tonne manure individually co-digested with six underexploited co-substrates (including maize and straw), fixing DM to 10%. Sole manure digestion was compared, showing significant savings from avoiding conventional manure spreading, stressing the importance of AD (Hamelin, et al., 2011; Meyer-Aurich, et al., 2012). Bioresidual handling was most responsible for positive impacts. Source-segregated manure performed best in all categories, and within global warming accounted per FU, Nm^3 biogas, and t DM, respectively (-1256, -6.5, -2.7 kg $CO_{2\ eq}$). This is mainly owing to avoided spreading of raw manure on land (and marginal electricity), the liquid fraction being important marginal substituent despite significant impacts from handling. Garden waste performed the second (-313, -4.2, -1.3), owing to composting avoidance, along with utilization of biogas and bioresidual. AD of household (-101, -0.8, -0.3) and commercial (-32, -0.2, -0.1) biowaste performed modest, however benefiting more when co-digested with manure compared to direct CHP combustion along with raw manure spreading. Varying the AD fugitive CH_4 loss from 1% to 10% (only maize) increased the impact from 1018 to 1542 kg CO_2 eq per FU due to less energy utilization thus more global warming. Changing lost alternatives from INC to landfilling for the household and commercial biowaste (from -101 to -184, and -32 to 6 kg CO_2 eq per FU, respectively) indicated poor performance of the lower DM commercial biowaste as INC substrate.

Similarly Vega, et al. (2014) studied source-separated manure, organic household waste, and straw compared against conventional manure management. Low nutrient content and high DM and CH_4 potential made straw perform best. AD performed considerably better for fossil depletion and marine eutrophication compared to reference, while the remaining depend on co-substrates used. Household waste was worst ranked in fossil depletion due to pretreatment and digester heating. Co-substrates diverted to AD from a treatment otherwise substituting e.g. fossil marginal yield less net environmental benefit. This confirms the statements in Cherubini & Strømman (2011). Interestingly, climate change savings for source-separated manure compared to household waste had a net negative impact difference of factor 12 in Hamelin, et al. (2014) compared to a net positive impact of a factor four in Vega, et al. (2014), in favour of the manure. Source segregated manure performs worst in climate change when field N_2O emission factor is adjusted from 2 to 6% N_2O-Ntot, doubling the global warming magnitude, altering the ranking as mentioned in Crutzen, et al. (2007). CH_4 emission from AD and upgrading, and biomethane potential do not affect the ranking, but alter climate change and fossil depletion. Varying NH_3 only shifted the ranking of separated manure in terrestrial acidification and marine eutrophication compared to reference, and avoiding eutrophication when high NH_3 field emission due to less N leaking potential. Transportation aspect was unimportant.

Vega (2012) also compared conventional manure management to AD of manure with co-substrates from Vega, et al. (2014). The second best is the solid manure fraction, performing better than baseline in all five impact categories except fresh- and marine water eutrophication, while household waste performed worse than the baseline in all categories due to its connection to energy recovery from incineration. System expansion showed to be crucial for the performance. The most important hotspot occurs from GHG emissions from organic matter field application, and a prior degradation lowers these. During AD the hotspots occur from heat consumption, especially from treating more wet substrates, and fugitive CH_4 emissions from upgrading. Other hotspots depend on co-substrate chemical composition and on replaced use in current practice.

Poeschl, et al. (2012a) tested variants in the biogas system: (i) Feedstock supply, (ii) biogas production, (iii) utilization pathways, (iv) bioresidual handling. Emission level variations are significant and mainly influenced by fossil CO_2 and CH_4. CO_2 from (i) for MSW is a factor 53 higher than cattle manure (1.8 kg/t waste) as it besides transport includes collection and more demanding pretreatment. Food waste yields the largest net savings of CO_2 (-79 kg/t), owing to the savings from biogas utilization, followed by slaughterhouse (-61) and MSW (48), WWTP grease (-40) and cattle manure (-26). The largest saving occurred in process (iii). Two feedstock mixes treated (%): cattle manure (13/0), MSW (14/90), Food (10/6), slaughter paunch (14/4), and grease sludge (49/0), assumed CHP without external heat utilization, and direct bioresidual spreading form open storage. Respective biogas yields are (MJ/t) 1475 and 2810. Overall emissions CO_2 (kg): -63 and -99; and fossil CH_4 (g): 312 and 709. (i) and (ii) cause hotspots for the residue mixes, for CO_2 and CH_4 respectively. Emissions from (iii) mainly relied on conversion efficiencies and potential fossil substitution from biogas. Net CO_2 system emission (kg/t) was lowest for fuel cell technology with external + internal heat recovery (-110) and CHP without external (-63), due to significant fossil savings. No utilization savings arose from biogas upgrading (162), transport upgrading (167). In (iv) there is a trade-off between loss of nutrients from digestate separation and efficient transportation. 90% NMVOC is saved by transporting the solid fraction, while only 6% of SO_2 from fertilizer substitution with raw digestate is saved by the separated solid. Harnessing the biogas remained in the stored bioresidual could reduce the CH_4 emission by a factor up to 14.

Pöschl, et al. (2010) found significant variation in plant energy efficiency arising from feedstock type and applied process. Primary energy input/output ratios of biogas system for the respective mixtures of 34%, 55%, and 45% with the major single shares from (ii). Energy input highly relies on e.g. industrial waste demanding pretreatment. Ratio depends on biogas yield, utilization efficiency, and energy value of replaced fossils. Non-agricultural residue ratios range 52% (grease) to 64% (manure). Negative energy balance is reached when transporting manure and grease 17-22 km, slaughter and food 75 km, and MSW 425 km. Energy ratios for biogas upgrade from small-scale CHP heat (replacing fossil) for internal use (1.3%) and without CHP (12%), CHP without heat export (34.1%), and vehicle fuel upgrade (8.7%), and fuel cell with short distance heat transmission (6.1%). For each variant (i) maintained the largest energy input proportion. Primary energy savings (GJ/t DM) are respectively: 650, 750, 100, 1’000, and 150. The ratio of bioresidual treatment ranges between 30.5-36.4%[2]. The best of several options includes screw-press separation and direct spreading, unlike decanter separation and solid composting for utilization. Drying is appropriate only when using surplus heat. The separation technology is key for energy input (handling efficiency). Screw-press dewatering of manure + MSW saves 50% transportation energy input within 40 km, and with decanter the respective saving is 34% and 42% within 120 km.

Poeschl, et al. (2012b) suggests that co-digesting more residues and organic waste minimizes the environmental load and energy balance. Biogas recovery from bioresidual storage reduced impacts from this process tenfold due to both reduction of GHG impact and additional energy recovery. Climate change impacts (kg $CO_{2\ eq}$/t) are for cattle manure (-23), MSW (-53), food (-52), pomace (-86), slaughterhouse (-51), sludge (-29). All impacts are negative for human toxicity, water depletion and fossil depletion. Impacts from (ii) per tonne of two co-digested mixtures in Poeschl, et al. (2012a), respectively (fixing CHP utilization without heat export, and direct bioresidual spreading): climate change (-61 and -120 kg $CO_{2\ eq}$), terrestrial acidification (-0.05 and -0.16 kg $SO_{2\ eq}$), marine eutrophication (0.02 and 0.03 kg N_{eq}), and fossil depletion (-19 and -32 kg oil $_{eq}$). Climate change and depletion impacts from (iii) upgrade for grid and vehicles (fixing the former co-digestion mix and direct bioresidual spreading: (191 and 204) and (8 and 35). From (iv) e.g. screw-press + composting and decanter + solid impacted the most on climate change (79 and 78) and largest saving from screw-press + solid/liquid + residual biogas utilization (-22). Acidification impacts are low and for fossil depletion impacts are respectively (0.1, 2, and -5.2).

Lyng, et al. (2012) developed a LCA model to identify climate hotspots in the entire biogas production value chain. Biogas from manure and household waste is suitable and substrate mixing is beneficial. Highest GHG benefit originates from biogas upgraded to replace diesel fuel when bioresidual from all the waste is spread on land. Best performance occurs from maximal biogas output, and highest impacts occur from N_2O emission from several processes. This one has the highest uncertainty.

4.3 LCA of waste treatment systems

According to Münster & Lund (2010), INC is most commonly compared with biogas production in LCA, but conclusions are usually unclear. Under certain conditions environmental performance from biogas with CHP utilization are comparable to INC (Kirkeby, et al., 2006; Baky & Eriksson, 2003), but may be even better if combined with refuse derived fuel (RDF) (Cherubini, et al., 2008).

Clavreul, et al. (2012) performed a case study of AD and INC of kitchen waste in Denmark source sorted and comingled with residual waste. AD yielded global warming savings of -301 and -357 kg

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[2] System energy efficiency improves app. 5-6% when recovering residual biogas from bioresidual (Pöschl, et al., 2010)

$CO_{2\ eq}$/t, respectively. Largest net saving contribution is owing to electricity recovery in INC and AD, and considerably more from heat in INC compared to AD (21% more energy). This is due to less calorific value of biogas than of waste and due to the different energy recovery efficiencies, as also stated in Hjort-Gregersen & Petersen (2011). Some AD saving arises from fertilizer substitution but is somewhat counterbalanced by load from emissions during bioresidual spreading and utilization and collection and transport. Water content in waste is the most sensitive parameter (INC and AD) followed by heating value and vegetable ratio (INC), waste BMP, CH_4 yield potential in digester, and electricity recovery from gas (AD). The relative difference generally remained the same after methodological variations, but the performance ranking altered by change of incinerator. Uncertainty analysis reveals INC 80% likely to be preferable over AD.

Fruergaard & Astrup (2011) found that CHP recovery INC of source separated household waste compared better than AD in most categories. Impacts from bioresidual utilization and fugitive process emissions were significant for the AD performance. For INC it depends on the flue gas cleaning efficiency. The NO_3^- and Hg emission load from bioresidual could be avoided by incinerating it, however potentially losing benefits of Cd and Cu savings from inorganic fertilizer replacement. Nutrient enrichment suffers the most from AD scenarios. Global warming net savings from INC were respectively threefold and six-fold compared to AD CHP and transportation fuel use. AD yielded considerable net savings in water ecotoxicity, but similar net impacts on water human ecotoxicity. Marginal energy mixes are important but not decisive. INC mainly replaces coal, natural gas and some biomass for AD CHP, and oil for AD transportation fuel. A 100% coal substitution for AD CHP improved the GHG savings to half of the level of INC as its energy efficiency is twice higher than of AD CHP.

A Danish study similar to Rehl & Müller (2013) took a consequential approach. Among eight chosen technologies, including CHP INC, Münster & Lund (2010) found biogasification (based on household organic waste and manure, incinerating the fibre fraction CHP) as the alternative with the highest potential for annual CO_2 reduction for biofuel (Mt): (>3), for local biogas CHP (>2.5) and for regional INC (<1) in a present energy system with optimal use of full resource potential. Sole household waste undergoing AD produces net CO_2 when used for CHP but less for biofuel. Remaining wastes should still be incinerated with high efficiency CHP after the waste hierarchy priority.

Hjort-Gregersen & Petersen (2011) based its consequential study on the strategy from Fødevareministeriet (2008). AD is the most beneficial way of exploiting energy from separated manure fibres and the efficiency is crucial. INC saves on N leaching, but N is lost to flue gas during combustion. P is lost to bottom ash, but is unfeasible to extract (Miljøministeriet, 2013b). This loss induces more commercial fertilizer. Heavy metals also redistribute differently. In a Danish context individual energy yields from solid fraction is 2.1 MJ/t with AD and 2.5 MJ/t with INC, as AD only converts 50-70% organics. However, enabling a requisite co-generation with 4 t raw manure yields 3.6 MJ/t. AD of cattle manure of which 11% is separated fibre reduces GHG by 40 kg $CO_{2\ eq}$/t gasified manure and 70 kg $CO_{2\ eq}$/t input primarily from avoiding N_2O and CH_4 emissions from storage, spreading and natural gas use. INC of separated raw (pre) and degassed (post) manure fibres reduces 26 and 19 kg $CO_{2\ eq}$/t, respectively. Since no C is returned to soil savings drop to 14 and 19 kg $CO_{2\ eq}$/t. Biogas production with subsequent INC yields a fivefold GHG reduction compared to INC of fibre from raw manure. Currently it is easier to divert the fibre to INC, imposing lost alternative for the emerging dry process biogas plants. This has implications on GHG emission reduction and nutrient recovery as similarly analyzed in Rehl & Müller (2013). Fødevareministeriet (2008) also states that INC of manure fibre is not climatically interesting. With results normalized to composting, which has slight net impacts in all categories, proved INC of food waste the highest total energy saving, followed by AD with CHP and fuel use, respectively. For non-renewable energy and global warming the only savings came from AD. The latter is sensitive regarding the alternative fuel and transport distances. AD and INC ranking varies in different cases, but the Waste hierarchy is valid as a rule of thumb (Finnveden, et al., 2005).

In Börjesson & Mattiasson (2007) biogas has often much better performance compared to liquid fuels. Biogas fuel from liquid manure yields the largest saving among vehicles fuels from energy crops, compared to fossil petrol/diesel, of -180% ± 40% down to -60 g $CO_{2\ eq}$/MJ fuel (Concawe, et al., 2006; Börjesson & Berglund, 2007). This is owing to both CH_4 emissions reduction (mostly from conventional storage) and fossil fuel replacement. The reduction uncertainty significantly depends on local conditions, as CH_4 emitted from manure and N_2O from soil. Indirect effects other than energy system replacement are rarely accounted in biogas system analysis. This e.g. in terms of lost alternatives, such as usual manure storage and treatment of organic waste otherwise composted, which may emit NH_3, N_2O and CH_4, being significant if no gas cleaning applied. This might vary significantly depending on local conditions (Börjesson & Mattiasson, 2007). Nutrient recycling replaces energy intensive fertilizer production, potentially making up 10% of the energy content in biogas product, avoiding CO_2 and N_2O from process natural gas (Berglund & Börjesson, 2006a). However, also this primary energy can vary between 42-70 MJ/kg N, and 3-9.6 kg $CO_{2\ eq}$/kg N produced, as stated by Cherubini, et al. (2009). These indirect effects are considered as background impacts in LCA and in principle the allocation can continue in a long downstream production chain (EC, 2010). This lifecycle thinking is commonly practiced in more recent consequential LCA studies (Hamelin, et al., 2014; Vega, et al., 2014).

Börjesson & Berglund (2006b) specifically studied gas emissions from fuel-cycle of biogas from organic MSW, liquid pig manure, and slaughter waste used for CHP or vehicles. The emission factor between two biogas systems may vary a factor 3-11. Those are significantly affected by feedstock properties, biogas production energy efficiency, and status of end-use technology. Waste (MSW) collection and transport distance is often a significant source, as well as CH_4 leakage from storage or upgrading; 1% biogas loss equals >85% emissions from production phase. Also the source of electricity used, emission data, and vehicle type may substantially affect the results. Fuel-cycle emissions are usually much lower for substrates requiring less handling, i.e. residuals (slaughter), while the differences are smaller for manure and MSW based systems. However, the fuel-cycle emissions presented differ considerably from previous studies due to methodological differences, including indirect effects from system expansion (Börjesson & Berglund, 2006b). Börjesson, et al. (2010) reported GHG savings (%) from fossil replacement using: grass (-86), manure (-148), food industry waste (-119), and household waste (-103). Replaced fertilizer and decreased CH_4 and N_2O leakage yield more savings.

Zah, et al. (2007) assessed GHG balances from transportation fuel provision. All biomethane fuels performed better than fuels (in order of descending impact), from: manure, manure + co-substrate, biowaste, grass, and sewage sludge[3]. Most impacts stem from production. Fuel from manure (optimized[4]) had more negative GHG emissions than from manure + co-substrate (optimized). Savings stem from avoided spreading of raw manure[5]. The large impacts are caused by CH_4 and N_2O emissions from secondary residue fermentation and CH_4 leakage from upgrading. All CH_4 fuels thus have GHG less than 70% and environmental impact less than 100% of that of petrol. Relative to petrol diesel causes 90% GWP and natural gas 80%. Fuel from manure (optimized) has < 10% in GWP and holds least impact in all except eutrophication, followed by fuel from manure + co-substrate (optimized). In other categories methane fuel generally has least impact compared to fossils, except in eutrophication, where the manure CH_4 exceeds 400% of the one from petrol. By relating net utility from energy use to environmental impact from GWP for CHP and car fuel, manure based performs best in both while modest for biowaste and sewage sludge. For average technology MSWI, those two are rated similarly, while in latest technology MSWI biowaste performs well and sludge worse.

Møller, et al., (2009) found that AD treatment of source separated household and garden waste, with biogas for CHP or vehicle fuel use and fertilizer for substitution, results in GHG ranging -375 to 111 kg $CO_{2\ eq}$/t waste, and -95 to -4 kg $CO_{2\ eq}$/t for a specific plant. The range is due to variation of key parameters: energy substitution by biogas, bioresidual N_2O soil emission, fugitive CH_4 emission, unburned CH_4, carbon binding in soil, and fertilizer substitution. For a specific type of AD facility the range was -95 to -4 kg $CO_{2\ eq}$/t waste. For comparison, GHG impact from N_2O soil emission would be higher than the impact from even a 100 km transport distance. Emissions of CH_4 (fugitive and from biogas production) are proportional to the produced CH_4/t waste, also regarding

[3] Miljøministeriet (2013b) performed an LCA on sewage sludge spreading on land with respect to P utilization
[4] "Optimized" includes covering the second fermentation tank

[5] Liquid manure storage in tanks leads to release of GHG and NH_3 which also causes nutrient loss. The efficiency of cover materials are compared, and regarding NH_3 reduction impermeables reach 95% while e.g. straw (cheap) has 25-85% and significantly reduces CH_4 (English & Fleming, 2006).

avoided emissions from energy substitution. Uncertainty ranges of especially fugitive CH_4 losses highly influenced the results. This also applies for the possibility of N_2O reduction by substituting mineral fertilizers with bioresidual considering slow-release N source (Møller, et al., 2009). Thus a high biogas production replacing CO_2 intensive energy can have substantial saving, while a low CH_4 yield along with upgrading and high N_2O emissions from land could make AD a net GHG emitter.

Findings from Smith, et al. (2001) disregard CH_4 and N_2O losses, only considering sequestered C. Total GHG load from AD of MSW (paper and putrescible), depending on replaced energy mix (wind, coal, EU-average), is -51 to -165 kg $CO_{2\ eq}$/t waste (only electricity recovery) and -132 to -246 kg $CO_{2\ eq}$/t waste (CHP recovery). Composting the same waste, if used to replace fertilizer, results in more than -32 kg $CO_{2\ eq}$/t waste of which N is responsible for 33 of 36 kg $CO_{2\ eq}$/t waste material savings. For CHP INC of mixed MSW the net GHG emissions are -161 and -563 kg $CO_{2\ eq}$/t waste (replacing wind and coal, respectively).

Comparing INC, Quiros, et al. (2014) found that autoclaving of MSW (household) and sorting into an organic fibre fraction for AD with or without different composting, and inorganic for recycling and incineration, performed environmentally best in eutrophication and global warming. This is mainly owing to biogas recovery and compost utilization. Ranking of alternatives relied on material (most GHG saving) and energy recovery of inorganics (substituting virgin) and energy efficiency and LHV. Sorting credit only partly offsets autoclave net impact, as energy consumption from autoclaving caused 2-6% of total impacts, and electricity in the biological treatment scenarios caused major impact in most categories. Mainly emissions impacted acidification and eutrophication in all scenarios, while for composting electricity highly impacted acidification. The resulting compost as mineral fertilizer substituent yielded noteworthy savings. INC had the lowest impact on acidification, fossil depletion, and energy demand. The study did not substitute national marginal energy.

Hamelin, et al. (2010) investigated co-digestion of raw pig/cattle slurry with fibre fraction from centrifuge + PAM pre-separation for CHP with centrifuge post-separation for P recovery. Alternatives are screw-press pre-separation for CHP without post-separation, and biogas production based on raw slurry and fibre pellets. LCA concluded similar to most other literature reviewed here. Environmental benefits depend greatly on pre-separation efficiency of nutrients in manure, especially biogas potential. Decanter centrifuge very efficiently transfers VS to the fibre fraction even without polymer use (Hamelin, et al., 2011), allowing liquid fraction and bioresidual less CH_4 release during storage. Global warming can be reduced by covered and short storage time of fibre prior to biogas plant, a two-step biogas production with an airtight covered post-digestion tank, and a covered storage of degassed fibre fraction. Benefits are highly reliant on energy source substituted. Acidification and N-eutrophication are insignificantly lower due to data uncertainty. Efficient P recovery reduces P consumption and P-eutrophication if applied in fields with P deficiency only. Transport and electricity contribute significantly to non-renewable energy, but is by far counterbalanced by fossil replacement from CHP. In all cases major hotspots originate from in-house storage (NH_3 and CH_4) and field processes.

4.4 LCA of pretreatment technologies

Hamelin, et al. (2011) compared conventional manure spreading with biogas production using three different manure-separation technologies[6], where the solid and liquid phase is AD treated and spread on field, respectively. Performance highly depends on efficiency. A DM separation of 87.2% (opposed to 60.9% and 29.6%) enabled a relative 40% net reduction of global warming from CHP, being only 29% if biogas replaced natural gas. Most VS ended in biogas plant avoiding emissions during outdoor storage and application, and simultaneously greater marginal displacement from biogas production. The liquid fraction storage and application accounts for 16%, 47%, and 46% GHG emissions, respectively, being 30-33% from inhouse storage. The PAM decanter centrifuge option yielded a net impact less than reference in all categories, and almost in every case less than the two other technologies. Keeping 100% natural gas heat marginal, and changing the power marginal from equal coal and natural gas share to sole natural gas, the difference would not exceed 5% in all categories.

Prapaspongsa, et al. (2010) compared management variants for sole manure and mixture with grass + glycerine, substituting energy used for mineral fertilizer production and natural gas. INC combined with liquid/solid separation and efficient solids drying yields energy utilization and GHG savings higher than AD scenarios. Manure INC reaches net 1019 MJ/t (81% of energy content) and for AD scenarios average 49%. 16.8% is recovered from conventional manure management. GHG savings are proportional and the largest occur in INC scenarios for manure and mixture, respectively 70 and 133 kg $CO_{2\ eq}$/t. Only the references impose net emissions. For energy rich mixtures savings and pre-storage emission impacts are larger. Manure treatment compared to reference increases energy recovery (up to 64%) and saves on GHG (up to 112 kg $CO_{2\ eq}$/t). Combining INC with AD, including treatments, energy production increases by 24% or up to 41% of calorific value if electricity recovery is maximized. In this case individual treatments in optimized AD and INC scenarios obtain similar efficiencies from production and fertilizer substitution. Despite energy yield from mixtures being twofold, the choice of applied technology is key for efficient energy utilization. Additionally, AD produces transportable gas, increasing the energy utilization potential and system flexibility.

[6] Møller & Chitra (2005) found that the biogas yield from AD of manure increased up to 64% when applying pressure-cooking and chemical pretreating. On average, the increase is 26-44%, compared to no-treatment yield of 236 L CH_4/kg DM. Applying decanter centrifugation highest nutrient recovery occurred at higher G-force.

In Prapaspongsa, et al. (2010) all scenarios considered similar yields of P (no losses). N recovery is similar in non-INC scenarios, with minor losses from storage evaporation, while INC transforms N into other gases. Total GHG emission is most sensitive to mass and DM in separated solid fraction, and DM and VS in manure. DM separation efficiency controls energy output from INC, while DM in raw manure affects energy potential in both INC and AD scenarios. The mass affects direct GHG emissions from untreated liquid fraction, while VS affects storage CH_4 emission and AD energy yield, similar to findings of Hansen, et al., 2007. Both energy exploitation and GHG balance improves by first increasing the dry matter content, e.g. by enabling in-stable passive system separation, such as slatted floor (Schuchardt, et al., 2011), or improving mechanical process efficiency (Hjorth, et al., 2010).

Carlsson, et al. (2015) evaluated methods of pretreating source separated organic MSW, with slurry directed to AD and refuse to INC. The performance was influenced by the DM content and distribution in the fractions. A net GHG decrease is achieved when more DM is diverted to AD, even though the net energy generation decreases when more material undergoes AD unlike INC. Keeping good slurry quality, the amount of refuse should be minimised, as well as the water content during pretreatment, to increase system efficiency. GHG emissions are very sensitive to electricity generation/use in pretreatment and assumptions of marginal energy.

Wesnæs, et al. (2009) compared reference slurry management (0) to technologies of (1) acidification (to reduce N_2O and CH_4) and (2) mechanical separation with direct fertilizing with liquid and (a) spreading fibre pellets on land or (b) for heat production. Savings by (1) for e.g. global warming and acidification are more than 20% and 60%, with a 10% net impact for N-eutrophication and non-renewable energy, slightly higher for pig than cattle manure, especially in acidification. Inhouse CH_4 emission data are sensitive, as retention time below one month has 20% less global warming unlike more than a month. Savings by (2a) are most significant for acidification (10%) and N-eutrophication (5%). The major hotspot in non-renewable energy (140%) is from energy consumed for pellet production. Transportation to field contributed somewhat. Savings by (2b) in the most impact categories are around 5-10%. N-eutrophication is smaller as most of the N is removed to the fibre fraction. A net impact of 20% in non-renewable energy is due to larger saving from applying liquid manure as fertilizer and replacing production of mineral fertilizer, instead of replacing fossil energy by incinerating. All savings have high uncertainties but the trend is confident, except for global warming. The performance certainly depends on separation efficiency for nutrient distribution, as the highest impacts originate from the liquid fraction N (storage and field).

Hamelin, et al. (2013a) carried out a LCA of manure management techniques in the context of Poland (pig manure pelleting and thermal gasification with mineral P production) and Denmark (co-digestion with source separated solid and digestate separation for P optimization, and, pig slurry cooling). For Poland, thermal gasification allows improvement in all categories compared to conventional handling. P is recovered from ashes avoiding P-eutrophication. However, N is also lost like during incineration. Sensitivity analysis shows importance of P separation efficiency and of covered solid fraction storage in all impact categories. Cooling lowers impacts from inhouse NH_3 and CH_4 formation but N-eutrophication remains intact due to the fertilization rules. However, the choice of electricity margin will influence the cooling system performance. Alternative emission reduction options are suggested, such as acidification of slurry (NH_3) or limit inhouse storage duration (CH_4).

4.5 LCA of post-treatment technologies

Rehl & Müller (2013) accounted GHG cost-efficient use of food, garden, slaughter waste for AD, applying eight different biogas conversion techniques to produce power and/or heat, or biofuel, compared to a fossil fuel based system. The most cost-efficient biogas technology uses CHP and is located nearby heat using sectors. Another viable option is raw biogas distributed in the medium calorific gas grid. A plant with district heating grid or a system based on biogas imposes highest GHG costs. The result from all systems are significantly influenced by the choice of marginal or average mix data as reference, including especially lignite/hard coal and natural gas, and is mainly decided by the electricity credit. They also depend on local infrastructure availability and environmental profile of substituted systems. Fugitive CH_4 emissions from biogas production, and grid injections (1-2%), resulted in decreased energy efficiency and increased GWP.

Comparing the widely used water scrubbing (WS) with two novel biogas upgrading technologies utilizing INC ashes to store CO_2, Starr, et al. (2012) found that using alkaline and fly ash (AwR) had 84% higher impact than alternatives in all categories mainly due to the energy intensive manufacture of reactant, modestly followed by electricity use. Global warming had 10% savings due to CO_2 storage ability. BABIU provides bottom ash for use and yields lowest impacts in all non-transport related categories and with a global warming saving of 80% owing to CO_2 storage. BABIU electricity use (55%) for drying and biomethane preparation, and transportation (23%) posed the largest impacts. WS is mid-rated with overall impacts caused by electricity use (97%) and with GHG saving of 15%. Including the CH_4 recovery factor, different technologies obtained GHG savings of (in t CO_2 eq per t CO_2 removed): BABIU (9.1), WS (9.1), MS (8.7), PSA (8.5), AS (8.1), AwR (8.0), Cry (7.5), and OPS (7.5). Electricity remained the largest contributor (especially for PSA and Cry), where loss of CH_4 is the case for MS.

Comparing with natural gas provision for CHP Pertl, et al. (2010) also found upgrading of biogas from pig manure + organic MSW (household) with BABIU (32) as the most GHG saving option among WS (109), PSA (162), MS (207), and NG (294) alternatives per FU (kg $CO_{2\ eq}$/100m^3 upgraded). This is owing to the unique credit from CO_2 sequestration. Considering sole upgrading process ranking is similar. In all alternatives the largest load originates from upgrading and AD process, and considerably from waste collection between 11% (MS) and 20% (WS).

Anderson-Glenna & Morken (2013) found scarce literature with GHG emissions from stored bioresidual (pig and cattle manure). CH_4 emissions are very small compared to conventional manure management but relatively high after short hydraulic retention time, as in Hansen, et al. (2006). Summer emissions were 10-fold compared to winter, for both bioresidual and raw manure. Considerably less CH_4 and NH_3 escaped from covered storage, while more N_2O escaped from only bioresidual (warmer periods). However, this was regarded as the best scenario. In the worst case scenario, with open-tank cattle-manure bioresidual the GWP approaches 100 kg $CO_{2\ eq}$/t DM from CH_4, 275 when including N_2O emission. The largest GHG savings in all scenarios of bioresidual storage are owing to substitution of mineral fertilizer and fossil energy by the AD byproducts. In general the impacts are primarily from transportation to, and storage of bioresidual, closely followed by transportation and spreading. The smallest net savings are from the cattle reference, medium from pig reference, and largest from a basis scenario of closed storage of pig-manure bioresidual. Thus biogas production with uncovered bioresidual storage performs better than solely manure use as direct fertilizer, making covered storage even better than the best case uncovered storage of pig and cattle manure.

Visvanathan (2014) sought post-treatment options to avoid GHG emissions from MSW bioresidual in Asia. Increase in DM increases VS and consequently the GHG potential of bioresidual, similar to statements in Prapaspongsa, et al. (2010) from pretreating. Raw bioresidual storage for 2 months would emit 10% of its GHG potential, suggesting reduction of storage time to avoid GHG (C) and also nutrient (N, P) loss (Visvanathan, 2014). The storage phase accounts for 27% of global GHG emissions from the AD value chain (Gioelli, et al., 2011). GHG emission potentials were found for bioresidual spreading (g $CO_{2\ eq}$/kg waste): direct raw (139), stored (125), stored-cured (80), and compared to raw MSW (568). The net emissions (g $CO_{2\ eq}$/kg bioresidual) are: -11, 12, and 13. Prior to land application of stored-cured, passive dewatering and curing takes place. Storage performs best mainly owing to avoided CH_4 emission compared to other scenarios, but also highest saving on mineral fertilizer manufacture, as fossils and nutrients are preserved. However, C/N ratio and time of application must be considered for proper management. The results depend on bioresidual characteristics relying on AD substrate type and digestion process.

4.6 Other related studies

Hansen, et al. (2007) tested pretreatment technology effects on quantity and quality of source-separated household waste. Quality is high biogas potential and less impurities having implication on biogas production and bioresidual application. Shredder + magnet (98%) recovered most of biomass (though removing only metal impurities), followed by screw press (59%), and disc screen (66%), with the remainder being reject mostly consisting of organic matter. The yield of the former is 102 m^3 CH_4/t waste collected and 40-60 m^3 CH_4/t for the latter ones. The technologies showed differences in chemical component distribution, for dry matter, easily degradable matter, and collection bag material. Plastic collection bags contribute to a 10% reject. CH_4 yield depends on the weight based separation efficiency whereas factors of city, technology, dwelling and season show insignificant variations of average potential (459 STP m^3 CH_4/t VS). A similar study on CH_4 potential finds considerable influence on the waste chemical composition (Hansen, et al., 2003). This is in agreement with the statements from Davidsson, et al. (2007). Ariunbaatar, et al., (2014) reviewed numerous studies on organic MSW pretreatment and concluded that low temperature thermal pretreatment and two-stage AD methods is most favourable to enhance AD.

Fødevareministeriet (2008) analyzed several measures to reduce direct GHG emission from agriculture and from producing biomass to replace fossil energies, respectively, without imposing significant structural changes in agriculture. For selected measures the reduction is stated in brackets (in kt $CO_{2\ eq}$/y by 2020), based on different use of available potential, substituting natural gas. AD: livestock manure (546+350), grass clippings (-45+148); INC of degassed manure (73+59), of separated pig manure (52+43); Manure management: as cooling in stable (4), frequent emptying (-12), covering slurry tanks (41), covering solid manure (1), increased fat in feed (248), reduced N-norm (93), nitrification inhibitors in fertilizer (272). Biogas production also has the potential for reducing N leaching (Fødevareministeriet, 2008).

Hansen, et al. (2006) developed a model to estimate CH_4 emission from stored co-digested organic MSW bioresidual under typical Danish climatic conditions. Temperatures in on-farm storage tanks are linearly correlated with air temperature. Since storage tanks are typically emptied around spring the bioresidual content stored is low meaning limited CH_4 emissions during the year. Short reactor retention time leaves only 75-85% organic matter degraded while the remaining potential is usually transformed during the month-lasting storage before spread on land. Emission to air is governed by climate and storage ability to recover biogas. Linke, et al. (2013) found that residual CH_4 yield in digestate storage tank was a function of temperature and storage time, also stated by Wesnæs, et al. (2009) regarding loss of dry matter for slurry storage, adding that the emissions should be modelled as a function of time. Angelidaki & Ellegaard (2005) had found that two-stage AD reactors may increase the biogas yields by up to 15%, thereby lowering the potential for GHG emissions. Similarly, German Muha, et al. (2015) developed a dynamic model and concludes that CH_4 emissions from manure storage tanks are high and should be covered. Sommer, et al. (2001) modelled predicted emission reduction from agriculture in Denmark by co-digesting manure.

Tonini, et al. (2014) performed MFA and SFA on nutrient and energy recovery from enzymatic treatment of household waste with downstream separation and energy conversion (AD of bioliquid and INC of residual solid). Waste refinery may recover 56% of the DM input as bioliquid (as 6.2 GJ biogas energy) and the potential for N, P, and C recovery is 81-89% of the input. Digestate quality is a challenge as it may cause emissions from land application.

5 Biogas potential

The theoretical biogas potentials for Denmark and Poland depend on the availability of feedstock for anaerobic digestion, in terms of mass quantity and calorific quality. Selected residual substrates falling within the scope of the study are included, while other agricultural substrates (i.e. energy crops and straw) are generally omitted.

5.1 Biogas potential in Denmark

The biogas production in Denmark is planned to be more than doubled by 2020, from 4.3 PJ to 10 PJ (Energistyrelsen, 2014a). The total potential estimate is 40-60 PJ (Birkmose, et al., 2013). Currently only 0.5% of the total Danish energy consumption is covered by biogas and can rise to a 10% share in 2040 (Danmarks Naturfredsforening, 2014). Utilization of biogas will primarily take place in the heating (7 PJ) and power (9 PJ) sector in 2020 (Appendix A: National energy data).

Biogas yield potentials for wastes are presented in Table 6[7]. Straw and crop residues (omitted from Table 6) will have a considerable potential in the long term, respectively 390-870 Mm^3 and totally around 250 Mm^3 CH_4 (12% of the input of expected biomass). However, environmental benefits of energy crops are limited (Hamelin, et al., 2014; Vega, et al., 2014) and e.g. maize use is politically restricted to 1.5% thus resources must be found as residues (Wittrup, 2014). Slurry and manure is expected to decrease slightly due to livestock decrease but keeps a high biogas potential, min. twofold compared to all remaining residues in Table 6.

Table 6. Present and future total annual dry matter and methane potential of agricultural wastes. Modified from (Birkmose, et al., 2013). Substrates irrelevant to the scope of this project are omitted (energy crops and straw)

Organisk affald	VS/DM (%)	DM, 2012 (1000 t)	DM, 2020 (1000 t)	CH_4, 2012 (10^6 Nm^3)	CH_4, 2020 (10^6 Nm^3)
Gylle	80	2,106	2,004	402	383
Fast staldgødning	80	900	20	16	4
Ajle	80	5	0	1	0
Naturarealer	90	236 – 365	236 – 365	60 – 90	60 – 90
Randzoner	90	14 – 72	14 – 72	15 – 35	15 – 35
Grøftekanter	90	70 – 140	70 – 140	3 – 16	3 – 16
Have-park	90	108	130	10 – 24	12 – 29
Husholdning	90	200 – 250	Mindre end i 2012	72 – 98	Mindre end 2012
Industriaffald	90	Ukendt	Ukendt	Meget varierende	Meget varierende

Manure as an energy resource is fairly unexploited in Denmark, but only 5-7% of the generated 35 Mt/y is co-digested (Birkmose, et al., 2013; Hamelin, et al., 2011). Pig and cattle slurry is reported to have a methane potential of 427-871 Mm^3 CH_4/y, with a feasible potential of 50% the range (Luostarinen, 2013), lying within the range of Birkmose et al. (2013). Excluding straw and energy crops, manure alone had a potential to cover 80% of the total biomass potential for energy in Den-

[7] The data serves as a base for the revised version in (Energistyrelsen, 2014a)

mark (Angelidaki & Ellegaard, 2003). The geographic density of manure biogas potential is illustrated in Figure 5.

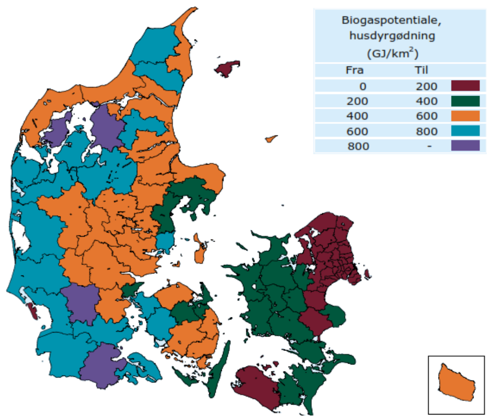

Figure 5. Biogas potential from livestock manure per municipality area in Denmark. Modified from (Ea Energianalyse, 2014c)

Figure 6 shows that straw has been recognized as a highly energy potential feedstock substrate. In 2012 it had fairly the same potential as manure, but is expected to increase slightly in 2020 while manure decreases. In addition, energy crops (agricultural resources) are anticipated to play an important role as co-substrates in 2020. An optimal combination of these resources might, however, lead to a sustainable utilization (Bentsen, 2012). Energy crops and straw is outside the scope and will not be elaborated on. Projected energies from biomass types including MSW and sewage sludge can be seen in Appendix A: National energy data.

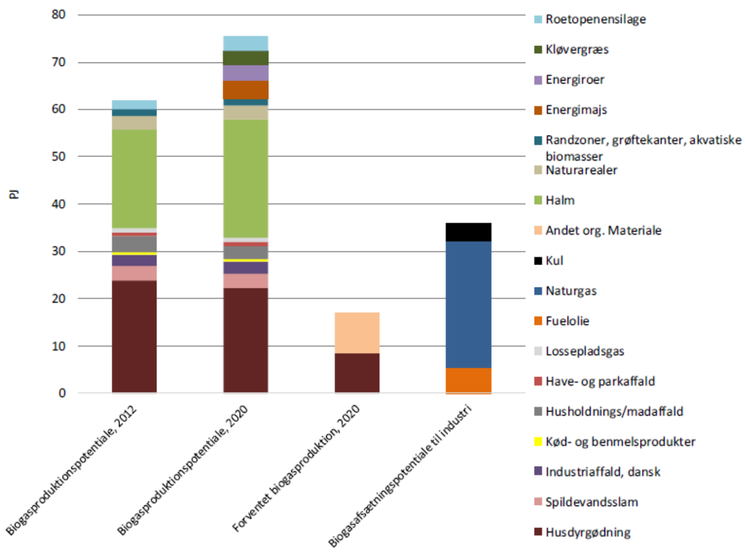

Figure 6. Biogas potential and substitution potential of organic resources in Denmark (Ea Energianalyse, 2014a)

Figure 7 exhibits a more exact picture of the column in Figure 6. Given that all the biogas is utilized in the highly energy consuming industries, biogas will be able to substitute more than half of this consumption in practice, mainly in the form of natural gas, but also fuel oil and coal as the marginal energy mix sources in industry.

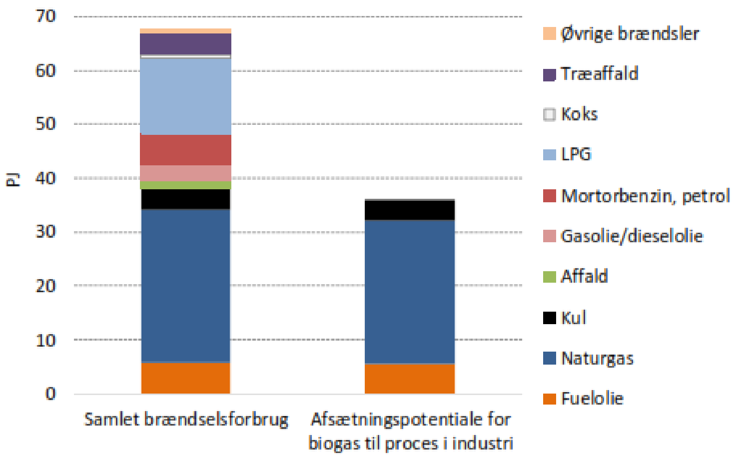

Figure 7. Total industrial fuel usage (in >10 TJ companies), and the potential for substituting fossils with biogas (Ea Energianalyse, 2014a)

Estimates show a total potential of organic wastes and residues of 10.7 Mt in 2008 (see also 6.3 Practice in Denmark). Of those 9.3 Mt was reprocessed, and the remainder was incinerated or recycled (Miljøministeriet, 2014). Table 7 shows that four of the six organic residual/waste sectors had a very high recycling rate. This suggests that household waste and organics from the service sector are unexploited resources, leaving space for a high utilization potential. The recycling rates of the various substrates points out that there will be a lost opportunity of utilizing them in alternative waste treatments, calling for making decisions on the best use.

Table 7. Recycling potential of organic residuals and waste (wet waste), 2008 (Miljøministeriet, 2014)

Commercial sources	Recycling rate (%)
Households	8
Garden waste from households	95
Service sector (retail, restaurants, canteens, institutions)	20
Public garden/park	99
Industrial residues	99
Wastewater treatment plants (public and private)	83
All sectors	87

At the same time, approximately 7.7 Mt of the 9.3 Mt reprocessed was utilized for raw material substitution or spread on agricultural land, while only 3.4% was biogasified, mostly industrial waste and less from the service and negligible amounts from the households. This statement in fact corresponds to the distribution in Table 7. The conclusive remark is here that there is a potential for an increased recycling of almost 1.2 Mt organic residues (Miljøministeriet, 2014). This is graphically illustrated in Figure 8.

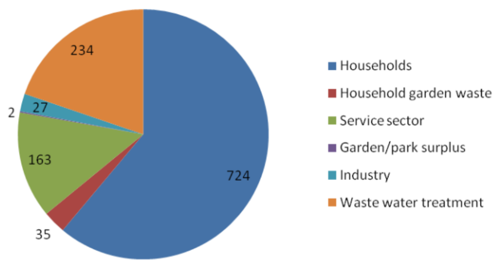

Figure 8. Potential for increased recycling of organic residuals (in 1000 t ww) estimated to 1,185,000 t (2008) (Miljøministeriet, 2014)

According to Angelidaki & Ellegaard (2003), owing to their higher gas potential combining manure with household and industrial waste in biogas production is very attractive. However, these feed-stocks are physically becoming unavailable. Olesen (2011) analyzed the biogas potential of various substrates in Danish municipalities and stated that especially the industrial waste (slaughterhouse, food processing) is already becoming fully utilized to the extent that import is necessary. Table 8 shows an overview of the total biogas potential from different sectors, showing that the agricultural sector represents the most unutilized potential.

Table 8. Exploited and unexploited biogas potentials of various substrates (Olesen, 2011)

Affaldstype	Udnyttet (PJ)	Uudnyttet (PJ)
Husdyrgødning	1	22
Husholdnings- og madaffald		2.5
Enggræs	-	3
Spildevandsslam	0.9	1-3
Industriaffald	1	<1
Kød og benmel	0.03	0.5
Have- og parkaffald	0	1
I alt (inklusiv energi- og efterafgrøder)	3.2	30-32 (83-86)

The gross biogas potential from pig and cattle manure is 29 PJ, but it is expected that only 75% can be exploited. In all Denmark 22 PJ corresponds to about 546.5 million m^3 which is similar to values in Table 6. Sewage sludge is limited in biogas yield as it is in most cases secondary sludge. Total biogas potential is estimated to 4 PJ of which 1 PJ is already exploited. Biogas plants treat around 350 kt industrial and food processing industry waste today, and the scarcity of these resources has induced import of 0.7 PJ (Olesen, 2011; Vega, 2012). The total biogas potential is 2.5 PJ of which 1 PJ is exploited (old data). Furthermore the remaining potential is used for animal feed or consists of low DM biomass. No more than 0.5 PJ of meat residuals currently used for feed, including stomach and intestinal, can provide biogas. Waste from households and catering is already well used for energy purposes but has 2.5 PJ biogas resources. Garden and park waste potential is estimated to 1 PJ in Denmark, as technical solutions are in the test phase (Olesen, 2011). The values in Table 8 are in accordance with the 2020 projections in Klima- og Energiministeriet (2010, p. 92, 125), where the energy production from biogas in 2008 is 4 PJ and the unutilized potential in 2020 is 35 PJ.

5.2 Biogas potential in Poland

Biomass is expected to cover 79% renewables in the 2020 energy system of Poland, with 20% from biogas alone (98 PJ), regardless of energy crops and biofuels (Baum, et al., 2013). 1,700 Mm^3/y of biogas can be produced from sewage sludge, slaughter byproducts, agriculture waste, and food industry, and could cover 10% of the national gas demand after upgrading (Hamelin, et al., 2013a). The energy from biomass production in Poland is planned to increase significantly between 2015 and 2020. A high energy potential is to be found in forestry, agriculture, but also in organic MSW comprising public areas, households, restaurants, commerce, and industry (Table 9).

Table 9 indicates that the energy potential from some organic wastes exceeds the potential of a country such as Denmark (Baum, et al., 2013). The potential is projected to be more than 57 PJ contributed by 5 PJ from sewage sludge. Similarly, the energy production from biogas is expected to increase significantly for use in the heating (19 PJ) and power (14.5 PJ) sector in 2020 (Appendix A: National energy data).

Table 9. Estimated domestic biomass energy supply in primary energy production (heating and cooling) in Poland (ktoe). Relevant sources are highlighted (Ministry of Economy, 2010a)

Sector of biomass origin	Biomass	2006	2015	2020
Forestry	Direct supply (forest)	1894	1071	1016
	Indirect supply	2279	931	1065
Agriculture and fisheries	Crops, fishery products, grass	124	405	1156
	Agricultural byproducts, processed residues	337	1358	1773
Waste	Organic MSW (garden/park, food, kitchen, households, restaurants, caterers/retails, food processing plants)	19	932	1369
	Biodegradable industrial waste (including paper)	5	154	269
	Sewage sludge	203	65	120

In the slaughter industry 27% of the animal weight ends as byproducts for utilization. 62% of the collected residues from processing consist of swine remains. The Polish market is deficient on animal fat from food processing, but waste fats amounting 80-100 million L could be considered as a very valuable biogas substrate (Hamelin, et al., 2013a). Namely, it has a methane potential of 700 m^3 CH_4/t DM (Chodkowska-Miszczuk & Szymańska, 2013). Lower quality residuals from fruit-vegetable food processing have the highest share in the processing group and can be used for biogas production. Used cooking oil has a potential of 100 million L/y but collection in Poland has been limited. Also expired food from e.g. bakery and retail is good feedstock. The planned expansion of WWTP will lead to more than 700 Mt of dry matter sewage sludge, of which 400 Mt will be subjected to thermal treatment (from larger WWTP). However, the technical potential for biogas production is high and 1 m^3 of sludge (4-5% DM) can provide 10-20 m^3 of biogas with 60% methane content.

70 Mt of manure is generated a year of which pig solid (50%), cattle solid (30%), and pig and cattle slurry (10%). Of this number 2.7 Mt is digested (Luostarinen, 2013). The theoretical biogas production from manure and liquid excrements is around 1,000-3,200 Mm^3 of which 90% is covered by manure as a very abundant resource (Ministry of Economy, 2010a). The theoretical energy potential from pig and cattle slurry and manure alone ranges between 1,740-2,700 Mm^3 CH_4/y. The techno-economical potential is then around 660-860 Mm^3 CH_4/y. This and corresponds to an energy potential of 6.6-8.6 TWh/y (Luostarinen, 2013).

In 2011, Polish agricultural biogas plants received 25% of feedstock as energy crops (mainly maize), and of the remaining 75% manures constitutes 62%, vegetable/fruit is 2.3%, 0.3% stomach content, and 0.06% fat waste (Chodkowska-Miszczuk & Szymańska, 2013). The analysis above suggests that remaining residual feedstock is not yet fully utilized for biogas production and there is a large space for improvement.

6 Legislation and Practice

This chapter introduces EU legislation within segments of farming, energy, and waste management. Subsequently Danish and Polish legislation is presented followed by description of farming and organic waste management practices.

6.1 EU legislation

Regulation is enforced in all EU due from a specific date, while Directive requirements enter into force few years after adoption and are to be implemented on a national level. The targets to be achieved may vary between the member states. Directives are legally binding only as to the result to be achieved by individual member states, leaving the national authorities the choice of form and method (Welford, 1996).

As EU policy making has progressively raised sustainability targets certain directives have been amended into more extensive ones. Currently applicable EU laws are presented in Table 10. The legislation is obligatorily transposed into national law of the member states. Directive 2009/28/EC (RES) is elaborated upon as being key with respect to the great focus on climate issues especially in LCA modelling (Cherubini & Strømman, 2011). Some of the remaining key directives transposed into national law are explained in forthcoming subchapters.

6.1.1 Directive 2008/98/EC (Waste Framework)

This overall legislation on waste provides general requirements in a framework of basic waste definitions and concepts. It comprises several directives and few regulations on waste management operations, definition of waste streams, byproducts, waste hierarchy, and requires member states to adopt national waste management plans and waste prevention programmes, integrated in environmental policy (EC, 2015b). The waste hierarchy is a guide model that illustrates the aim for achieving higher environmental benefits from e.g. recycling as opposed to landfilling or even incineration (Kiatkittipong, et al., 2009). It serves for member states to apply waste policy and legislation in the following order of priority (Münster & Lund, 2010; EC, 2015b):

- Waste prevention (preferred option)
- Re-use
- Recycling
- Recovery (including energy recovery)
- Safe disposal (last resort)

Table 10: Summarization and categorization of interrelated EU legislation (directives and regulation) in the context of this project. Compiled from several sources (referred)

Directive No.	Title of rule	Main points
Waste		
2008/98/EC a), b), c), d), e), f)	Waste Framework	Applies for products and byproducts, including organic waste Five-step waste management hierarchy to promote material recycling Environmentally best waste management option to be determined by lifecycle thinking (LCT) by member state (Article 4(2)) Requires establishment of national waste prevention programmes Authorities to set management plans governing waste type, quantity, origin Incinerators with high energy efficiency classified as recovery operation
1999/31/EC a), b), d), e), f)	Landfill	Standards for landfill design, operation, and aftercare Reduce emissions of methane and leachate Reduce waste disposal Some nations like Denmark (1997) entirely banned organic waste disposal
86/278/EEC a), b), f)	Sewage Sludge	Regulates sewage sludge application in agriculture to protect particularly soil Sludge has valuable agronomical values and application must not compromise soil and agricultural product quality Prohibits use on grassland/forage crops if to be gazed or harvested within a certain period Prohibits use on soil for fruit/vegetable crops, except fruit/vegetable crops in direct contact with soil and normally eaten raw Quality criteria set (heavy metal limit values)
749/2011/EU (1069/2009/EC) Regulation a), d)	Animal By-products	Health rules for animal by-products and derived products not intended for human consumption (including food waste) Requires pathogen inactivation by heat sterilization (hygienization) prior to (an)aerobic treatments Quality criteria set (categories)
Energy		
2014/52/EU h)	Environmental Impact Assessment (EIA)	Effects of e.g. energy or transport project on the environment
2010/75/EU a), c), d), e), f)	Industrial Emissions	Concerns integrated pollution prevention and control (IPPC) from combustion facilities and other associated processes Emission limits for (waste) incineration facilities Energy efficiency optimization through self-sufficiency Management of incineration flue-gas and bottom ashes To limit environmental impacts from emissions to air, soil, waters etc. Meeting Framework Directive requirements (e.g. Annex IIc) requires energy efficiency of min. 0.60 or 0.65 for MSWI)
2009/28/EC e), f)	Renewable Energy Sources (RES)	Promoting renewables for achieving 20-20-20 goal, supported by improved energy efficiency
Agriculture		
2000/60/EC g)	Water Framework	To prevent/reduce pollution and protect bodies of surface- and groundwater, an accordance with local conditions
91/676/EEC f), g)	Nitrates	To reduce freshwater pollution from agriculture source nitrate Designate vulnerable zones of land draining, and monitor eutrophication Establish codes of good agricultural practice (voluntary use)
a)(MWE, 2010) b)(Christensen, 2011) c) (EC, 2011a) d) (EC, 2015a) e)(EC, 2008) f)(Arvanitoyannis, 2008) g) (Frandsen, et al., 2011) h) (EC, 2015d)		

6.1.2 Directive 2009/28/EC (RES)

The EU Climate and Energy Package 2020 set a so-called 20-20-20 target with the key objectives to reach by 2020 (EC, 2015c):

- 20% reduction of GHG emissions (from 1990 levels)
- 20% share of renewable resources in the gross final energy consumption
- 20% improvement in energy efficiency

The EU Directive on Renewable Energy Sources (RES) sets goals to enhance implementation of renewable energy resources to 20% in the energy sector by 2020, and in addition 10% renewables in the transport sector on an overall EU level. Also, energy from bioliquid (and biofuel) should contribute to at least a share of 35% GHG emission reduction (increased to 50% by 2017). Raw material used for biofuel producing must be qualified as sustainable in line with the Directive criteria (EC, 2010)

6.1.3 Regulation 1069/2009/EC (Animal By-products)

The Regulation on Animal Byproduct (Article 8, 9 and 10) categorises different waste materials from derived from animals and being of high (Category I) to low (Category III) risk in terms of pathogens and human health. These are displayed in Table 11.

Table 11. Categorisation of certain material requiring special treatment and application. The project -relevant categories are II and III. Based on (Fødevareministeriet, 2015; Miljøministeriet, 2004)

Cat.	Material type	Application / disposal
I	Infected animals and byproducts International kitchen-/food waste Category I-III mixes	Incineration with controlled ash disposal (with/without pre-processing) Co-incineration (e.g. for cement)
II a	Livestock manure and gut content	Direct fertilizer use or after biogas or composting treatment
II b	Other byproducts not intended for human consumption Byproducts collected from wastewater treatment Category II-III mixes	Biogas or composting after pretreatment (sterilization) for soil improvement
III	Pathogen-free material from slaughterhouse, incl. blood, tissue, skin Animal byproducts for feed Households, restaurants, catering food waste, former food	Biogas or composting after pretreatment (sterilization) for soil fertilization Incineration/gasification

According to Chapter I-III of Annex V of the consolidated Animal Byproduct Regulation 142/2011/EU, biogas plants must be equipped with pasteurization/hygienization unit, and the composting plant with a closed reactor and monitoring installations. It requires that Category II and III material for treatment in biogas plant and composting plant must have a max particle size of 12 mm before entering the unit and a min. temperature of 70°C for min. 60 minutes, and the bioresidual and compost must fulfil pathogen standards including reduced spore-forming bacteria and toxin formation (EC, 2011b). Regarding incineration of certain animal by-products (850 °C for at least 2 s), ash residues should be disposed of or recycles in accordance to EU law, as it would allow for recovery of phosphorous resources from the ash (EC, 2011b).

6.1.4 Directive 86/278/EEC (Sewage sludge)

According to this directive, it is prohibited to apply untreated municipal sewage sludge for agricultural purposes and the application is limited by the extension of pretreatment. The EU limit values are displayed in Table 12, measured in soil or in sludge for final use in agriculture. The lower and upper limits are defined for pH values lower and higher than 7, respectively. Limit values and other specifications for each EU27 member can also be found in (ORBIT, 2008).

Table 12. EU limit values of different heavy metals in soil or in sludge for use in agriculture (EC, 1986)

Parameter	In soil (mg/kg DM)	In sludge for use on land (mg/kg DM)
Cd	1 – 3	20 – 40
Cu	50 – 140	1000 – 1750
Ni	30 – 75	300 – 400
Pb	50 – 300	750 – 1200
Zn	150 – 300	2500 – 4000
Hg	1 – 1.5	16 – 25
Cr	–	–

Holm-Nielsen, et al., (2009) have summarized the multipurpose legislative value of biogas from AD in Europe compared to conventional manure management:

- Environmental (climate change)
    - Reduce energy consumption
    - Cut emissions from: transport sector
        - Electricity production and distribution
        - Livestock production
- Agriculture (nutrient management scheme)
    - Better control of ammonia emissions
    - Easier management of P/fibre separation
    - Reduced mineral fertilizer application
- Health and hygiene
    - Improved bio-security from pathogen reduction
    - Treatment of animal byproducts, kitchen, catering, restaurant waste and utilization for energy and bio-fertilizer production
- Waste reduction and recovery/recycling
    - Reducing amount disposed to landfill
    - Increase recycling and recovery

6.2 Legislation in Denmark

This subchapter is divided into three sections of Danish national legislation. The segmentation and overview of national strategies and legislation which is in accordance to, but more specific than of the overall EU law, is presented in Table 13.

Table 13. Overview of relevant Danish national strategy and law for the waste segment

National Strategies and Law
Waste
Resource Strategy (Ressourcestrategien 2018-2022, 2013)
Sludge Decree (Slambekendtgørelsen; BEK nr 1650 af 13/12/2006)
Energy
Energy Strategy 2050 (Energistrategi 2050, 2011)
Energy Agreement (Energiaftalen, 2012)
Green Growth Agreement (Grøn Vækst aftale, 2009)
National Action Plan - For renewable energy in Denmark (Klima- og Energiministeriet 6/2010.)
Manure management
Fertilizing Decree (Gødskningsbekendtgørelsen; BEK nr 903 af 29/07/2014)
Livestock Manure Decree (Husdyrgødningsbekendtgørelsen; BEK nr 594 af 04/05/2015)
Environmental Protection Law (Miljøbeskyttelsesloven)
Agriculture law (Landbrugsloven)
Action Plan for the Aquatic Environment III (Vandmiljøplan III, VMPIII, 2005-2015)

6.2.1 Waste

The Resource Strategy (Miljøministeriet, 2013a) is the major national strategy on solid waste management. It recognizes that the use of organic waste (typically household) can be used to produce biogas valuable for the energy system, and help reducing environmental impacts from livestock. Nutrients from organic wastes have a fertilizing role which would be lost if incinerated. Most garden waste is composted for application as fertilizer, but large branches scarce on nutrients are more suitable for incineration. Phosphorous from livestock manure is to continue to be used as a fertilizer once the manure has been used for biogas energy recovery. The expected effects of its initiatives are collected in Table 14.

Table 14. National 2018-2022 targets on management of generated organic waste (Miljøministeriet, 2013a)

Initiative	2018 Level (%)	2022 Level (%)
Household waste to be recycled, including increase of wet organic waste separation from 50,000 to 300,000 t, at source and central sorting facilities		50
Energy recovery of garden waste (branches less suitable for composting)	25	
Commercial organic waste (restaurants, supermarkets etc.) collected and exploited for biogas	60	
P from sewage sludge to be recycled by recovering P from incineration ash and apply it as fertilizer, or by spreading it on agricultural soil	80	

The Sludge Decree regulates which waste types can be used for agricultural purposes and is partly based on the Animal Byproduct Regulation. The quality requirements of the waste, including sewage sludge, are strict in terms of hygiene and treatment (Miljøministeriet, 2006), and heavy metal limit values are much more stringent than EU provisions (Kelessidis & Stasinakis, 2012). The quality criteria compares with those for organic and inorganic fertilizers (Miljøstyrelsen, 2015). Obligations for sewage sludge treatment and specific requirements along with treatment methods in Denmark are presented by Kelessidis & Stasinakis (2012).

The Decree includes waste from households, institutions, and companies. Garden/park waste, manure and silage, and animal byproducts are excluded (§2). Waste intended for soil utilization (fertilizing/soil improvement)[8] or which is fed into manure based biogas or processing plants must comply with the limit values in sludge represented in Table 15 (§6).

Table 15. Danish national limit values of different heavy metals in soil or in waste (per dry matter or per total P) for use in agriculture (Miljøministeriet, 2006)

Parameter	In soil (mg/kg DM)	In sludge (mg/kg DM)	In sludge (mg/kg P)
Cd	0.5	0.8	100
Cu	40	1,000	–
Ni	15	30	2.5
Pb	40	120	10,000
Zn	100	4,000	–
Hg	0.5	0.8	200
Cr	30	100	–

Substrates to be mixed must be individually analysed (§7, §15) (Miljøministeriet, 2006). If limit values of particularly sewage sludge are exceeded, it must be diverted to destruction plants, as dilution is not allowed (Kristensen, 2015). If the share of processed animal products in bioresidual is >25% DM of input, spreading on agricultural land is regulated by the Sludge Decree (Table 16), otherwise by the Livestock Manure Decree (§10). Prior to AD quality control by waste producers includes heavy metal, organic pollutants, and physical impurities. After AD the bioresidual quality control in addition includes pathogens and declaration of NPK (Holm-Nielsen, et al., 2009). Compost is covered by the Sludge Decree and raw materials in principle must meet heavy metal and organic compound requirements before processing. In that case the final product does not need to be analysed by authorities (ORBIT, 2008).

Areas covered with waste and manure may not exceed 170 kg N/ha/y and 30 kg P/ha/y (§22). Liquid waste spreading is restricted beyond winter season (§23), and for solid waste it is allowed between harvest and early winter at winter crop areas (§26). The hygiene restrictions for application are complied in Table 16 (§27). Specifications and treatment definitions can be found in Miljøministeriet (2006). Within the Controlled hygienization it is possible to choose between combinations such as treatment in a reactor 70°C/1h, or equivalently 52°C/10h to 55°C/6h for thermophilic AD.

[8] In §4. Definition of "jordbrugsformål" unlike "jordbrug"

Alternatively, the hygienization requirements before or after AD is 55°C/5.5h to 65°C/1h for thermophilic AD and 55°C/7.5h to 65°C/1.5h for mesophilic (Miljøministeriet, 2006).

Table 16. Hygiene restricted application of selected waste types (Miljøministeriet, 2006)

<table>
<tr><th>Behandling/
Affald</th><th>Ikke behandlet</th><th>Stabiliseret</th><th>Kontrolleret kompostering</th><th>Kontrolleret hygiejnisering</th></tr>
<tr><td>Organisk madaffald</td><td>Må ikke anvendes til jordbrugsformål</td><td>Må ikke anvendes til jordbrugsformål</td><td>+
(1) (4)</td><td>+
(4)</td></tr>
<tr><td>Spildevandsslam</td><td>Må ikke anvendes til jordbrugsformål</td><td>Ikke til fortærbare afgrøder eller på rekreative arealer og privat havebrug. Nedbringes inden 6 timer efter tilførsel. (2)</td><td>Ikke til fortærbare afgrøder eller på rekreative arealer og privat havebrug.
(2)</td><td>Må ikke anvendes til jordbrugsformål</td></tr>
<tr><td>Animalske biprodukter, bortset fra organisk dagrenovation og storkøkkenaffald</td><td colspan="4">Skal følge reglerne, der fremgår af EuropaParlamentes og Rådets forordning (EF) Nr. 1774/2002 af 3. oktober 2002 om sundhedsbestemmelser for animalske biprodukter, som ikke er bestemt til konsum</td></tr>
<tr><td colspan="5">+ Can be used without further hygienic restrictions
(1) On use with cloven animals spreading of compost must happen before sowing
(2) Within 1 year after spreading only grass and corn or seed crops may be grown, and growth of edible crop directly in soil is not allowed. Also restrictions apply when spreading the compost to in forest
(4) For use on grasslands additional rules from the Animal Byproduct Regulation apply</td></tr>
</table>

6.2.2 Energy

According to IEA (2011) Denmark is a benchmarking country within energy and is advised to give more attention to natural gas as a flexible source in power sector to balance fluctuating renewables. In 2012 Denmark initiated a major Energy Agreement with a strategy for constructing new biogas plants by 2020. A Biogas Taskforce group investigates and supports possibilities for realizing the project. The report from Energistyrelsen (2014a) has resulted in several feasibility studies, with measures aiming to increase the availability of biomass, and to promote utilization of biogas for heat and power and for the gas grid and transport. The strategy of this project is to lie within a certain legislative framework for biogas production in Denmark, on an EU and a national level (Table 17).

The Energy Strategy 2050 (Klima- og Energiministeriet, 2011) plans to reach the overall 20% renewables in the energy sector and 10% in transport sector (Table 18). It additionally sets even higher ambitions to reach a 100% renewable energy system by 2050, and having out phased fossil fuels (coal, gas, and oil) from the power and heat sector by 2035. This will result in an 80-95% GHG reduction in 2050 from 1990 level (Klima- og Energiministeriet, 2011) in line with the EU climate agreement (EC, 2015c). Realizing the projected energy policy goals the energy system should reach 33% renewable energy by 2020, which is above the committed level (IEA, 2011). This will be achieved through several measures, such as overall:

- Increasing energy efficiency
- Decreasing gross consumption
- Introducing new green technological solutions
- Providing intelligent European energy system with flexible sources
- Repealing fossil fuels out of the energy system by introducing more renewables

Table 17. Overview of relevant laws, rules etc. constituting the framework for biogas production in Denmark. Extracted from (Energistyrelsen, 2014a)

EU	Miljøministeriet	Klima-, Energi- og Bygningsministeriet	Fødevareministeriet
Affaldsrammedirektiv Bioproduktforordning VE-direktiv VVM-direktiv Habitat-direktiv Nitrat-direktiv Vandrammedirektiv	Planlov VVM-bekendtgørelse Habitatbekendtgørelse Miljøbeskyttelseslov Slambekendtgørelse Affaldsbekendtgørelse Ressourcestrategi Naturstyrelsens biogasrejsehold	Lov om VE Elforsyningslov Varmeforsyningslov Naturgaslov Klimaplan Strategisk energiplan-lægning VE i proces	Landbrugsarealstøtte Landdistriktprogram Bekendtgørelser om gødning og plante-dække mv.

The latter is highly contributed by biomass and wind energy, as well as biogas, and biofuel. The agriculture[9] has highly unexploited biogas resources important for the transition. E.g. wind energy alone is expected to cover 50% of the power demand in 2020. Today it represents 39% (Norre, 2015). Also heat pumps will play a significant role. The Green Growth Agreement proposes that 50% of all livestock manure should by 2020 be utilized for energy purposes, and principally 100% in the long term (Foged, 2010), specifically for production of biogas which can repeal natural gas, oil, and coal (Klima- og Energiministeriet, 2011), and calls for support of biogas injection to the natural gas grid on an equal level as to de-central CHP plants. Enhanced manure utilization is recognized to benefit the aquatic environment and lower methane emissions from agriculture.

Table 18. National 2020 target and estimated, weighted share from renewable energy sources in the total (projected) energy consumption (Klima- og Energiministeriet, 2010)

Sector	2005	2010	2015	2020
RE sources for heating and cooling (%)	23.2	30.8	36.0	39.8
RE sources for electricity (%)	26.8	34.3	45.7	51.9
RE sources for transport (%)	0.2	1.0	6.7	10.1
Overall share of RE sources (%)	16.5	21.9	22.6	30.0

The National Action Plan 2010 (Klima- og Energiministeriet, 2011) describes aspects of transposing EU Directive 2009/28/EC (RES) into national context including biogas integration into the natural gas network (Article 16) as well as biomass supply. The general agreement is to utilize

[9] 16% of the total GHG emissions in Denmark stem from agriculture. It is expected that this sector contributes significantly for reaching the energy and emission goals (Fødevareministeriet, 2008)

biogas for CHP where possible or consider "downgrading" of natural gas, more cost-efficient than upgrading biogas. Expansion of pipelines from biogas plants to de-central CHP plants is being considered. Environmental potential exists in replacing natural gas with district heat. In the Green Growth CHP plants and natural gas grid are both expected to receive high contributions of biomass fuel. Regulation is to ensure a large part of district heating to be based on CHP. The projected share of renewables to meet the binding 2020 targets is divided by energy sectors (heating and cooling, electricity, and transport). Those are displayed in Appendix A: National energy data.

6.2.3 Manure management

The Fertilizing Decree regulates the use of natural fertilizers (Tybirk & Jensen, 2013). The N-norm prescribes the maximal kg total-N/ha to spread on a field, which has been estimated with the Farm-N so as to provide sufficient plant nutrient and simultaneously reduce losses to environment (Aarhus Universitet, 2015). This value depends on the crop and soil type and is annually updated (Foged, 2010). An extensive list of values for different soil and crop types is presented by the Decree annex (Fødevareministeriet, 2010), and the most recent table including values for P (Landbrugsinfo, 2014).

Mandatory utilization rates of N content in different waste types are estimated in law (§24, §25), i.e. the minimum share of N that must be taken up by plants (compiled in Table 19). For mixtures of the above types, including bioresidual, the N share that must be utilized is established by estimating the N consumption following §12 (§24). In Denmark farmers are allowed to supplement bioresidual with inorganic fertilizer, practically decreasing the mix N substitution rate (Møller, et al., 2009) and these represent the substitution of inorganic fertilizer (Wesnæs, et al., 2009). However, as not all N is converted to a mineralized form available for the plants (see Figure 2), the substitution is less than the relative 100% which the mineral fertilizer is considered to have (Hamelin, et al., 2014).

Table 19. Calculated Danish utilization rates for relevant wastes with total share of N taken up by plants (Fødevareministeriet, 2010)

Fertilizers from:	N utilization (%)
Livestock manure and urea	
Pig slurry	75
Cattle slurry	70
Biogas slurry	58
Solid livestock manure	65
Liquid fraction from separation, when solid fraction is incinerated	85
Other organics	
Sewage sludge	45
Household waste (composted)	20
Garden/park waste	0
Other types	40

If the share of co-digested processed animal products is <25% DM of treated biomass the bioresidual is perceived as processed manure and can be spread on agricultural land as regulated by the Livestock Manure Decree (cf. the Sludge Decree). The bioresidual is comprised by the fertilizer definition in the Nitrate Directive and thereby spreading and N balance requirements (Horsens Kommune, 2012). Bioresidual from manure does not have limit values for heavy metal but the ash from incinerated manure must be spread according to the Sludge Decree and not Bio-ash Decree (Hjort-Gregersen & Petersen, 2011).

Manure spreading is basically regulated within general rules from the Environmental Protection Law and Agricultural Law, comprising the Fertilization Decree and Sludge Decree, and some parts of the Livestock Manure Decree are found in the Livestock Law (Horsens Kommune, 2012). The Environmental Protection Law holds provisions on floor quality in barns (slatted floors) e.g. to reduce NH_3 emissions. A technology list (BAT) prohibits[10] broad spread slurry in Denmark, and it has to be applied with band laying systems or injected in case of bare soil/grassland (Foged, 2010). The current regulation also allows slurry acidification during field application as injection alternative (Sindhöj & Rodhe, 2013a; Tybirk & Jensen, 2013).

The VMP III requires a P and N leaching reductions of 50% and 13%, respectively (Foged, 2010). Requirements in regarding distribution of surplus P in slurry will result in the need of transporting large amounts of slurry from P surplus areas (due to high livestock density) to P deficient areas (Wesnæs, et al., 2009).

In the Manure Decree limit values are set for P surplus and N emission to aquifers (§16), as well as a framework for BAT and best practice considerations (§14, §34). Emissions are to be mitigated by several means including a requirement for disposing livestock manure by incineration or biogas production (§3-6, §43). In the Livestock Manure Decree (Miljøministeriet, 2015) leakage from stable and bioresidual tanks are prohibited in Denmark (Wesnæs, et al., 2009), and manure (solid and liquid) storage must follow §11-21 (Miljøministeriet, 2015). Pig slurry tank storage must as minimum have a floating straw layer, and with PVC roof for new tanks (Wesnæs, et al., 2009). Compost or solid manure kept on field must contain min. 30% DM prior to spreading (§13).

Wesnæs, et al. (2009) states that a maximal transport distance of manure and bioresidual with trailer tractor (spreading tanker vehicle) from storage to field is 10 km, and if longer (trailer) trucks must be used. This used to be in the Agriculture Law §18. However, this part of the law was repealed in 2010 and the allowed transport distance is to be evaluated by local authorities within project applications (Miljøministeriet, 2010).

[10] Broad cast spreading was prohibited in 2001 due to NH_3 and health issues. Slurry is presently spread during spring with band spreading with a boom equipped with trailing hoses. The European Regional Development Fund tightened the rules and slurry injection must happen on bare fields or fodder grasslands (Tybirk & Jensen, 2013)

6.3 Practice in Denmark

6.3.1 Organic waste management

Miljøministeriet (2014) analyzed the overall amount of organic waste and residues to investigate the level of current incineration and potential for nutrient utilization through anaerobic digestion or composting. The much larger agricultural sector waste/residue amounts are excluded from the analysis (see also 5.1 Biogas potential in Denmark). Treatment is summarized in Table 20 and the explanations below are based on information from Miljøministeriet (2014). It is believed that some values composed by several sources (particularly for industrial waste) have changed since 2008 with the introduction of new waste and energy strategies.

Table 20. Treatments of organic residuals and waste (in kt wet weight), 2008 (Miljøministeriet, 2014)

Source sector	Potential	Reprocessing (recycling)				Other treatment			Share %
		In situ	Central						
		Use on site	COMP	AD	Other external recycling	INC	LF	Other	
Household	784	21	33	5	–	725	–	–	8
- garden waste	697	139	523		–	–	–	35	6
Service	206	–	18	24	–	165	–	–	2
Park surplus	553	300	248	–	–	2	4	–	5
Industry	5,917	3,514	57	287	2,010	27	7	–	55
Sewage sludge	2,572	–	–	–	2,133	207	3	54	24
All sectors	10,731	3,975	878	316	4,143	1,126	14	88	100

The household sector includes primarily food waste of which more than 92% is collected with the residual waste stream for incineration. A minor part is recovered through municipal source-separating collection schemes for composting (4%) and AD treatment (0.6%), with the remainder being home-composted. In 2011, 87% garden waste was recycled and 4% incinerated and 4% landfilled (Miljøministeriet, 2013a).

The service sector includes catering, wholesale, restaurants etc. mainly as food residuals which is separately collected for composting (9%) and AD treatment (12%). Most waste is incinerated (80%) as a part of the mixed waste collected. Green waste collected from public areas parks is mostly composted (97%).

The industrial sector constituted by industrial sludge and food waste has the highest mass potential. Its byproducts are almost entirely used as substitution for own material (59%) as feed in agriculture or in other process industries. Other byproducts are used on land (34%), or supplied to AD (5%) and composting (2%) mostly from the food branch and industrial sludge. Accounting in DM 79% would be raw material substitution and 16% for soil improvement. Sludge and organic mixed waste is incinerated.

Sludge treated in different ways in municipal and private WWTP has the second largest quantity. After collection 24% is directly spread on agricultural land including bioresidual from AD, in addition to 58% matter mineralized after storage. 8% is incinerated and 7% exported. Miljøministeriet (2013b) reports the three former treatments as 52%, 12%, 24%, and 7%. In 2011, 995 WWTP produced sludge containing 140,000 t DM and 5,000 t of P. Along with livestock manure and organic waste generated the P amount is tenfold (Miljøministeriet, 2013a). 47% of the sludge is anaerobically treated, and 51% WWTP use centrifuge for dewatering (Jensen & Jepsen, 2005). 30% of the WWTP in Denmark produce biogas (Niero, et al., 2013).

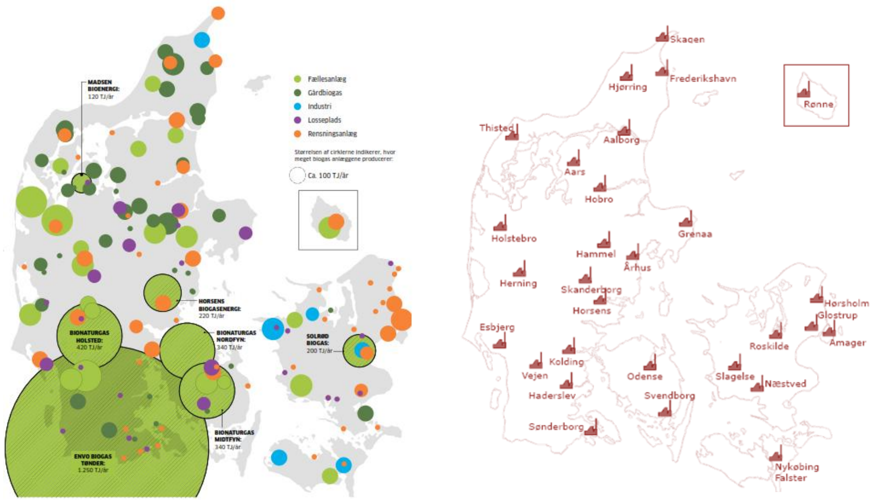

Figure 9. Map of biogas plants (left) (Wittrup, 2014) and incineration plants (right) (Energistyrelsen, 2014b)

Figure 9 localizes biogas and incineration plants in Denmark. Today 29 waste incinerators operate (Energistyrelsen, 2014b) and MSWI contributes significantly to the Danish district heating (20%) and power (5%) production (Fruergaard & Astrup, 2011). Most biogas plants currently produce CHP. The biogas production is displayed in Table 21. In 2013, 79% of the total was utilized for electricity, 20% for heat, and <1% for vehicles and flaring (IEA, 2015). Seven new centralized biogas plants are to increase production from 4.5 PJ to 7 PJ, and six of them are to upgrade the biogas (Wittrup, 2014). The total number of biogas producing plants was 176 (EBA, 2012) of which 80 treat manure (60 small and 20 large reactors) receiving around 2.5 of 34.4 Mt of manure generated in Denmark, of which ca. 88% is slurry and 10% cattle solid (Luostarinen, 2013). Only 5-7% manure is subjected to industrial waste co-digestion (Hamelin, et al., 2011; IEA, 2015) in an integrated treatment system (Holm-Nielsen, et al., 2009). Common practice at centralized biogas plants is namely that several farms deliver manure for free and in turn receive nutrient-equivalent amounts of bioresidual (Lemvig Biogas, 2014).

Biogas produced exclusively from slurry input is not yet the most common practice in Denmark, but is likely to become with the limited availability of energy rich co-substrates (Hamelin, et al., 2010). 25% of the slurry is expected to be separated for fibre incineration by 2020 (Hjort-Gregersen &

Petersen, 2011). The preferred bioresidual post-treatment technology in Denmark is decanter centrifuge rather than screw press (Frandsen, et al., 2011). Today in some cases the fibre fraction from pretreatment or post-treatment is incinerated with CHP recovery (Münster & Lund, 2010; Hjort-Gregersen & Petersen, 2011)

Table 21. Biogas production in Denmark, 2012 (IEA, 2015)

Substrate/Plant type	Number of plants	Production (GWh/year)
Sewage sludge	57	250
Biowaste	–	–
Agriculture	67	861
Industrial	5	51
Landfills	25	65
Total	154	1,218

6.3.2 Manure management

The conventional manure management system in Denmark for both fattening pig and dairy cow (cattle) slurry can be divided in the three main management stages/processes most relevant for the waste in the scope of this study, based on the references in (Hamelin, et al. (2010) and Wesnæs, et al. (2009):

- Inhouse storage
- Outdoor storage
- Transport and field process

These processes are illustrated in Figure 10. Hjort-Gregersen & Petersen (2011) also illustrated a conventional pig and cattle manure management system, along with an incineration and AD alternative, but these are not displayed here.

The Danish system is based on slurry (88%) and two optional technologies are in question: acidification and separation into two fractions (Tyrbirk & Jensen, 2013). In order to considerably reduce NH_3 emissions from storage and application of pig slurry, the use of SyreN systems is becoming common in Denmark. This involves addition of H_2SO4 sulphuric acid into the inhouse slurry pit, usually 4-8.5 L per m^3 slurry (Sindhöj & Rodhe, 2013a) or alternatively in outdoor tank or during field application (Tyrbirk & Jensen, 2013). Slurry separation is performed on-farm with decanter centrifuge, as screw press is less common in Denmark (Frandsen, et al., 2011).

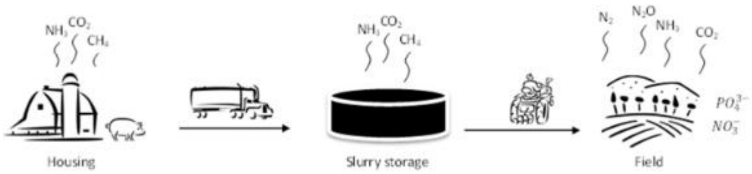

Figure 10: Conventional manure management in Denmark, with process specific emissions (Vega, 2012)

Pig farms: The reference system is based on a housing type with fully slatted floor, as the most common housing system for fattening pigs (50%). Once excreted, slurry is generated and stored inhouse in the pit below the animals. The pits are regularly emptied by flushing (or gravitationally) from the slurry space under the column to an outdoor pre-tank for temporal storage, typically 2-6 weeks. The slurry is then stirred and pumped to an outdoor storage tank.

The pig slurry is stored outdoor in concrete tanks and covered by a floating layer by the farmer, because a natural floating layer is less likely to occur. Straw is usually stirred into the slurry, as it is the cheapest cover (English & Fleming, 2006). PVC cover is also becoming more common. The retention time in the outdoor storage can be for months before application to fields. The slurry is again stirred before being pumped to the transport tank.

Different ways of handling the animal waste from farms exist, but the most common method of manure disposal is spreading on agricultural fields (Vega, 2012). Transport distances from farm to application soil are difficult to estimate. For slurry spreading on own farm fields distance is usually below 5 km, using tractor with trailer. If the distance to field exceeds 10 km, transport by truck is used. The pig slurry is then applied with trail hose tanker (band spreading) as the most common method today (68%). Pig slurry is applied to all crops in with a farm average of 140 kg N/ha/y. The most common crop type for Danish slurries is winter wheat (36.4%), spring barley (19.2%), and wheat (19.2%). 60% of all slurry was applied with trailing hoses during spring (Wesnæs, et al., 2009).

Cattle farms: The practice on cattle farms to a large extent resembles pig farm practice, but there are differences. The inhouse system is based on cubicle housing system with 1.2 m channel slatted floor (50%). Once excreted, the slurry is pumped from the pre-tank connected to the housing system to the outdoor storage. The cattle slurry is stored outdoor in concrete tanks and a natural floating layer crust is created from the straw bedding in the manure, and is regarded as a sufficient cover unlike adding additional straw. The transportation distances may be similar as for pig manure but there is no common distance. The range of 0.4 km up to 32 km has been reported. The slurry is applied with trail hose tankers to the field with an average 140 kg N/ha/y. A part is also applied by injection. The most common crop type for cattle farm is grass (20.6%), spring barley (17.2%), and maize (15.3%). The spreading also occurs during spring.

Manure treatment techniques: The major applications of different manure types in the Baltic Sea Region involve incineration or thermal gasification of solid manure and separated manure fibres. However, most the most widespread technique is AD of slurry, solid manure, and fibre fractions supported by co-substrates (Hjort-Gregersen & Petersen, 2011; Tyrbirk & Jensen, 2013).

To optimize usage the livestock urine fraction should be separately stored in covered tanks and applied as crop fertilizer while the solid fraction (10-30% DM) should undergo AD treatment. In Denmark this is achieved by separating slurry into a solid part before AD, rich on organic matter and a liquid part with partly dissolved organic matter lost from AD. Alternatively separation can be done after AD where the solid fraction is recirculated to maximize degradation of organics or simply treated into a solid fertilizer (composting and pelletizing) (Tyrbirk & Jensen, 2013).

6.4 Legislation in Poland

This subchapter is divided into three sections of Polish national legislation, in a form similar to the one on EU legislation. The segmentation and overview of national strategies and legislation which is in accordance to, but more specific than of the overall EU law, is presented in Table 13.

6.4.1 Waste

The Polish National Waste Management Plan 2020 (Ministry of Environment, 2014) is consistent with national and EU environmental policy. It analyses the current waste sector situation in Poland and suggests actions for improvements to be achieved by 2020 in terms of the main points:

- Decoupled growth of economy from waste generation
- Reducing waste generation including reuse, especially more recycling
- Treating waste as close as possible to source
- Developing implementation methods (technology, legislation, economics)

The National Waste Management Plan 2020 recognizes lacks of processing capacity of regional MSWI facilities and a too slow progress in separate collection of MSW. There is a focus on enhancing the development of systems for recovery and disposal, particularly for biodegradable waste. Analysis of management opportunities of sewage sludge flows are lacking and a high share is landfilled. The National Environmental Policy 2009-2012 action plan seeks to meet requirements specified in directives and in the National Waste Management Plan. Lines of action for preventing waste generation and developing the waste management system are summarized in Table 23.

Table 22. Overview of relevant Polish national strategy and law for the waste segment

National Strategies and Law
Waste
National Environmental Policy 2009-2012 (2016 outlook)
National Waste Management Plan, 2014-2020 (Anenx to Dz.U.Nr. 185, 2010.)
Sludge Decree (Rozporządzenie ministra środowiska w sprawie komunalnych osadów ściekowych. Dz.U.Nr.137, 2010.)
Energy
Energy Policy of Poland until 2030 (EPP 2030)
Renewable Energy Act
New RES Act (Parliament, 20/2/2015)
National Action Plan for Renewable Energy (Krajowy plan działania w zakresie energii ze źródeł odnawialnych, 2010.)
Directions of development for agricultural biogas plants in Poland between 2010-2020
Poland's Climate Policy – the strategies for greenhouse gas emission reductions in Poland until 2020
Poland's energy policy until 2030 (Document adopted by the Council of Ministers on 10/11/2009.)
Long-term Programme for Promotion of Biofuels or Other Renewable Fuels in Transport for the years 2008–2014
Innovative Energy - Energy Agriculture (Ministries of Economy and of Agriculture and Rural Development), 2009
Manure management
Water Law Act and Environmental Protection Act
The National Law on Fertilisers and Fertilization. 26.07.2000. Dz. U. Nr 89, poz. 991
Ministry of Agriculture Decree on application of fertilizers and education in fertilisation
Ministerial Decree on projects likely to have significant environmental effects (Dz.U.Nr.213, 2010.)
Code of Good Agricultural Practice from Ministries of Agriculture and Environment

A fundamental line of action is the increased use of biological and incineration treatments for mixed MSW. In order to reduce organic waste landfilling it is a precondition to establish facilities for composting, digestion, and incineration – i.e. an integrated system of regional solutions, complying with BAT criteria (Ministry of Environment, 2014). Supporting (optionally separate) collection and composting of kitchen and green waste is imperative. The Plan sees MSWI as a preferred option in regional areas.

It is necessary to expand the technical infrastructure, recovery, and recycling of organic waste through e.g. actions in the Council of Minister (13/7/2010). The Plan report partly finishes off by recommending the use of LCA as a useful tool for change oriented analyses in a waste management perspective.

Table 23. Compilation of relevant national 2010-2022 targets on management of generated organic waste. From the text in (Ministry of Environment, 2014)

Initiative	2015 Level (%)	2020 Level (%)	2022 Level (%)
Organic MSW			
Gradual reduction of deposited biodegradable MSW from 75% (2010), to	50	35	
Decrease mass of generated landfilled MSW	60		
Prepare for recovery and recycling several household waste fractions, including organic (garden/park waste etc.)		50	
Environmentally safe increase of energy recovery from MSW		x	
Sewage sludge			
Restrict storage			x
Enhance treatment and thermal processing before release to environment			x
Maximize the use of biogenic material in sludge while meeting sanitary, chemical, and environmental requirements			x

According to the national Sludge Decree the Polish standards on heavy metals are based on the lower EU limit values (Kelessidis & Stasinakis, 2012; Oleszczuk, 2006). Obligations for treatment of sewage sludge and specific requirements along with common treatment methods in Poland are presented by Kelessidis & Stasinakis (2012). Table 24 provides heavy metal limit values in sewage sludge that may not be exceeded for certain applications, reflecting the quality of use. Also limit values tested for different soil qualities are designated.

Laws on waste materials and fertilizers allow use of sewage sludge and mixed waste for composting or organic fertilizer production if the final product meets the heavy metal standards. Waste from animal origin must be approved and industrial organic waste must be excluded from fertilizer production. Regulations specify limits for compost amounts to be applied on soil but for N content there are only limits for manure spreading (ORBIT, 2008).

Table 24. Polish national limit values of heavy metals (mg/kg DM) in sewage sludge and soils for different application (ISAP, 2013)

Parameter	Reclamation of/use on agricultural land	Reclamation of/use on non-agricultural land	Cultivation of plants for composting and non-consumer crops (human and animal)	In soil for agricultural purposes	In soil for non-agricultural purposes
Cd	20	25	50	1 – 3	3 – 5
Cu	1000	1200	2000	25 – 75	50 – 100
Ni	300	400	500	20 – 50	30 – 60
Pb	750	1000	1500	40 – 80	50 – 100
Zn	2500	3500	5000	80 – 180	150 – 300
Hg	16	20	25	0.8 –1.5	1 – 2
Cr	500	1000	2500	50 – 100	100 – 200

6.4.2 Energy

The New RES Act 2015 on renewable energy aims for increasing energy sector development and possibility of self-sufficiency. According to (IEA, 2011) Poland is characterized by currently domestic coal abundance, where it accounts for about 55% of its primary energy supply and 92% for electricity production. At the same 80% natural gas and almost all oil is imported. Poland is dedicated to address climate issues and energy security by improving self sufficiency from domestic gas supply storages and new LNG terminals, and diversify energy mix by increased share of high efficiency renewables supported by CCS technologies. Another challenge is the ageing of CHP plants and decarbonising the power sector.

The projected share of renewables to meet the binding 2020 targets is divided by energy sectors (heating and cooling, electricity, and transport). Those are detailed by resource in

Appendix A: National energy data Appendix A: National energy data.

Table 25. National 2020 target and estimated, weighted share from renewable energy sources in the total (projected) energy consumption. Only few years of the series in (Ministry of Economy, 2010a) are displayed.

Sector	2005	2010	2015	2020
RE sources for heating and cooling (%)	–	12.29	13.71	17.05
RE sources for electricity (%)	–	7.53	13.00	19.13
RE sources for transport (%)	–	5.84	7.73	10.14
Overall share of RE sources (%)	7.2	9.58	11.90	15.50

The EPP 2030 sectoral strategy summarizes the socio-economical activities for achieving the EU goal commitment (Ministry of Economy, 2009). Biogas energy will largely contribute to the transition, and the forecasted effects are described in Ministry of Economy (2010b) and Ministry of Economy (2010a) such as:

- Improving energy security through increased supply of domestic renewables
- Gas supply, electricity, heat, and transportation can be based on agricultural biogas e.g. to deliver natural gas quality energy to dwellings and industry
- CHP efficiency improvement
- Generation of power and heat from raw materials not competing with food industry, categorized as by-products from agriculture, food industry waste and manure
- Obtaining large amounts of high quality granulated fertilizer from post-fermentation.
- Using organic wastes which does [not] emit GHG for energy production

Also mechanisms are set to extend the network of agricultural biogas plants. Biogas introduction to gas grids must follow limiting processing and distribution losses, as one aspect of the EU Climate Package is the technological development. These including improvements of:

- Biogas manufacturing from different types of agriculturally based substrates
- Methods of obtaining post-fermentation products
- Technology for conversion of biogas into CHP
- Purification process into biomethane

The ambition in the Innovative Energy - Energy Agriculture program of installing 2-3 GW biogas plant capacity by 2020 (one for each of 2,173 municipalities) will help reaching the EU goals (Chodkowska-Miszczuk & Szymańska, 2013). Ministry of Economy (2010a) signifies biogas integration into the natural gas network. The Act of 8 January 2010 amending the Energy Law Act contains several new regulations on agricultural biogas. Poland has not yet introduced biogas into the natural gas network due to higher profitability of power generation, but is expected in the near future according to the Polish Energy Law.

6.4.3 Manure management

The legal framework comprising and regulating on-farm manure management in order to protect environmental compartments is described below, based on both Sindhöj & Rodhe (2013b) and

Foged (2010). The laws from Table 13 describe safe manure disposal from the Fertilizer and Fertilisation Act, meeting sanitation requirements (not spreading on fields growing crops for human consumption), homogeneous spreading, injection depth, and groundwater table depth.

The Water Law requires prevention of N chemical compound discharge to water from agricultural activities (Article 47) and Regulation of Minister of Environment sets criteria describing water to classify as in danger of, or being polluted with nitrates, and describes thresholds for eutrophication initiation.

According to the Polish Act on Fertilizers and Fertilizing the dose of manure applied on land may not exceed 170 kg N/ha/y in the period of 1.3-30.11 (Article 17.3). In larger pig farms min. 70% of manure must be used on farm arable land and the remainder can be sold (Article 18.1). Manure and slurry must be stored in sealed containers of at least a 4 month capacity (Article 25.1). Fertilizers other than manure and slurry must be stored on impervious, hard surface pit secured against leaks into soil, with full leakage collection (Article 25.2). The Act prohibits use of liquid manure on soils without plant cover, and on vegetation intended for human consumption (Article 20.1.2a), and during the growing season of plants for human consumption (Article 20.1.2b).

The Act requires making a plan of fertilization, comprising law and the Good Agricultural Practice guidelines/recommendations which is based on environmental protection law (mainly soil and water). This includes ensuring proper use and storage of natural and artificial fertilizers, and to apply nutrient management regulations as well as equipment maintenance.

6.5 Practice in Poland

6.5.1 Organic waste management

CSOP (2014) accounted the waste management sector for 2013 in terms of collection of MSW fractions in each municipality, disaggregated by sectors and by treatment. However, it has not been possible to break down the data into treatment of organic waste from different sectors. Therefore it is necessary to provide an as detailed as possible overview given in Table 26 and Table 27.

Table 26. Treatments of MSW and belonging organic matter (in kt wet weight), 2013 (CSOP, 2014a)

<table>
<tr><th rowspan="2">Source sector</th><th rowspan="2">Collected</th><th rowspan="2">Of which separately collected</th><th rowspan="2">Of which organic waste</th><th colspan="3">Recovery operations</th><th colspan="2">Disposal operations</th></tr>
<tr><th>RECY</th><th>COMP or AD</th><th>INC (energy recovery)</th><th>INC (no energy recovery)</th><th>LF</th></tr>
<tr><td>Households</td><td>7,139</td><td>1,028</td><td>227</td><td rowspan="4">1,499</td><td rowspan="4">1,231</td><td rowspan="4">563</td><td rowspan="4">203</td><td rowspan="4">5,979</td></tr>
<tr><td>Services</td><td>1,952</td><td>172</td><td>35</td></tr>
<tr><td>Municipal service</td><td>383</td><td>75</td><td>51</td></tr>
<tr><td>All sectors</td><td>9,474</td><td>1,275</td><td>313</td></tr>
</table>

Table 27. Treatments of sewage sludge from municipal WWTP (in kt dry matter), 2013 (CSOP, 2014b)

Source sector	Collected	Applied			LF	Temp. stored	Thermal treatment	Other
		Land reclamation	Agriculture	Plant cultivation				
Sewage sludge	547	29	105	33	31	70	73	199

The service sector includes commerce, small businesses, offices and institutions, the household includes kitchen and green waste, and municipal service is public and green area maintenance/cleaning. 75% MSW was collected from households in total (Table 26), while similarly for mixed MSW- and 82% for separate collection, where also 75% of all the organic waste fraction was separately collected, as opposed to 50% in 2012 (CSOP, 2014a). The breakdown by fraction is displayed in Figure 11.

Table 26 also displays that at least around 13% of all collected waste is organic, as that share is subjected to composting or fermentation. This includes mainly wastes from garden and park, market places, kitchen and gastronomy (CSOP, 2014a). However, a large part of the organic waste is expected to have ended up in the landfilling, as this treatment option is the most common in Poland. There are very few MSWI plants in Poland, which is also suggested by the waste amount thermally treated, ca. 6% of all waste.

Sewage sludge generated in Poland is accounted in dry matter potential, where the DM of stabilized (mainly digested) sludge and dried sludge is usually 20% (Ministry of Environment, 2014). The second most common application is on agricultural land. This suggests that the heavy metal content in almost 20% complies with legislation, while only 5% is applied for remediating land. Around 13% is both stored for mineralization (maturing) and later application, and for thermal treatment including industrial co-incineration. It is unclear whether “other” includes AD or composting.

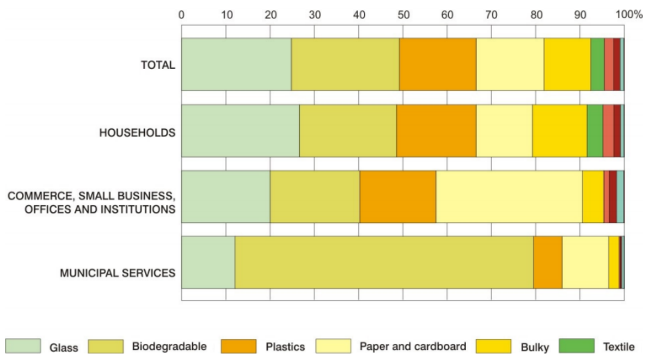

Figure 11. MSW collected separately by fractions and sectors, 2013 (CSOP, 2014a)

Figure 12 shows treatment facilities in each region of Poland. There is only one (non-hazardous) MSWI plant and numerous composting and sorting facilities for separately collected and mixed MSW, and numerous landfills. Incineration has not yet found a traditional practice in Poland but is planned to expand in the midterm (Ministry of Environment, 2014). 72% of the Polish biogas production in 2008 came from sewage sludge. Today Poland has 186 biogas CHP plants with a capacity of ca. 32 GWE and 33 GWT (EBA, 2012). 29 agricultural biogas plants were in operation in 2013 and are displayed in Figure 12. The brown plants are based on liquid manure.

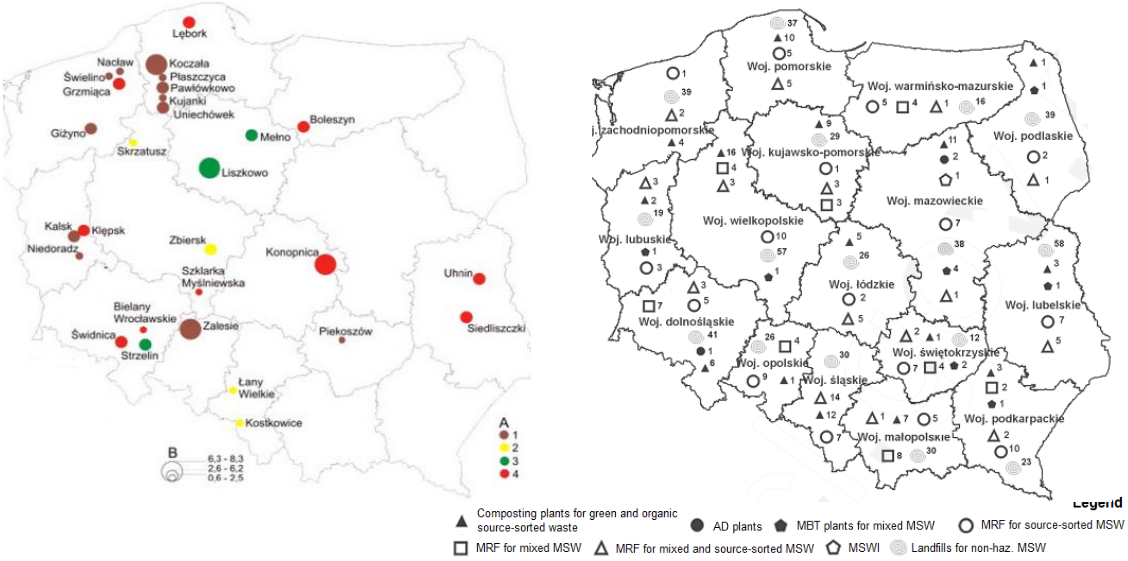

Figure 12. Map of agricultural biogas plants (left) (Chodkowska-Miszczuk & Szymańska, 2013) and types and number of municipal waste management facilities, 2009 (right) (Ministry of Environment, 2014)

Table 28. Biogas production in Poland, 2010 (Baum, et al., 2013).

Substrate/Plant type	Number of plants
Sewage sludge (WWTP)	46
Biowaste	–
Agriculture (co-digestion)	6
Landfills	73
Total	125

6.5.2 Manure management

Sindhöj and Rodhe 2013 describe manure handling techniques currently in practice on large-scale animal production farms (two cattle, and two similar pig farms) in Poland. The entire manure handling chain from housing system to storage and on to land application to crops is analysed. An illustration of the value chain from the analyzed farms is displayed in Figure 15. Otherwise, the most typical manure utilization pathway is direct spreading on land (Sindhöj & Rodhe, 2013b).

Figure 13. Manure management with biogas production (Vega, 2012)

Pig farms: Farm 1 and Farm 3 have a housing system with pens with slatted floor to produce liquid manure only. The manure falls below the pens into an underground channel from where it flows to a concrete manure pit (see Figure 14). Water is used for rinsing the floor. From the pit the manure is pumped to a covered temporary storage pound 300 m away (weekly).

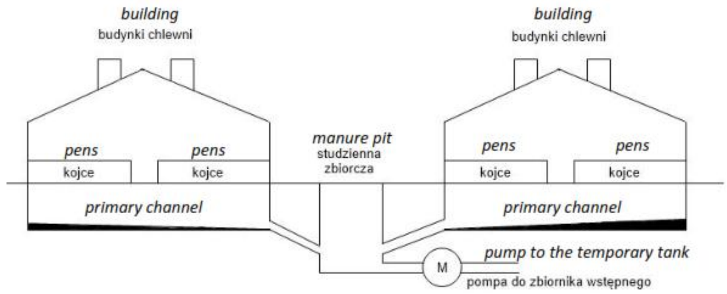

Figure 14. Cross section of the pig housing unit in Farm 1 and Farm 2

From here all manure is subjected to AD and is co-digested with slaughter waste and glycerine or green waste and plant processing waste, in addition to corn silage, to produce CHP. The digestate is transferred to covered storage ponds. Digestate is applied on corn and triticale (and barley) fields (<5 km away) by an umbilical system with trailing hose applicators or closed slot injection.

Cattle farms: All the barns are of open type (loose housing) and with straw beddings. On Farm 1 there are three housing systems in two barns (Figure 15). The first has a concrete base, from where manure is removed manually (daily) to a container outside the barn, and transported to a concrete pad (weekly). The second (hard floor) and third (slatted floor on top of pit) systems have cubicles and are connected. Manure is scrapped mechanically (daily) to an outdoor, concrete solid manure pit, and moved to the concrete pad by a mobile unit (monthly). The liquid part is collected from drainage system and pumped11 by a slurry tank into the liquid manure pit inside the barn (every 6 weeks). The liquid is sent to the pad by a mobile slurry tank during winter when spreading is forbidden. The leachate is directed into sewer.

[11] Along with collected rainwater and grey-water

Fertilizers from the pad (solid) and pit (liquid) are spread on respectively maize fields 25 km from farm, and on grasslands for feed production <1 km from farm. The respective spreaders are broadcaster and slurry tank.

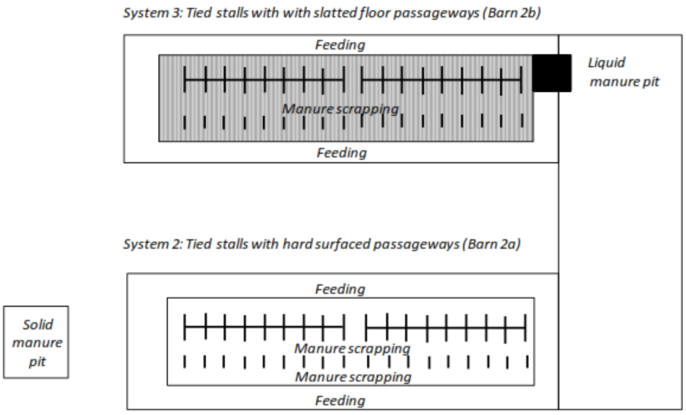

Figure 15. Cattle housing systems in second barn. Retrieved from Sindhöj and Rodhe 2013

On Farm 2 there are two housing systems. The first has an open concrete pen from which manure is manually removed (twice a week) directly to outdoor concrete pad. The liquid fraction is collected with a mobile slurry tanker and discharge into the temporary slurry pit in the other barn. From the other system manure is scraped (twice a day) with tractor to an outdoor concrete manure pit with slatted floor. Here it is mixed with the other liquid fraction. The slurry is moved (weekly) to an uncovered outdoor storage tank 20-100 m from farm. The solid manure is stored in both the concrete pad and in the field heap 2 km away.

Slurry is spread on rape (250 m away) and maize (7 km away) fields, while solid manure is spread only on maize fields. Respectively band spreaders with trailing hose and broadcaster is used. Spreading occurs on summer (August) on rape and in spring (April/May) on maize fields. In autumn (October) some manure is spread on grassland.

The farm has recently acquired a manure storage tank (under-floor pit) and an outside press-separator for the manure. Depending on the efficiency, a part of the solid fraction can be recycled as bedding with remainder exported as quality P fertilizer, while the liquid fraction is pumped directly to a roofed storage tank nearby.

7 Methodology

This chapter describes the materials and methods used and important assumptions made to conduct the study and framing the system definition.

7.1 Life Cycle Assessment (LCA)

LCA is a tool for modelling environmental impacts from products and systems through lifecycle thinking (LCT) and can be used for decision making. The application follows the ISO 14000/44 standards defined in (EC, 2010). LCA can be comparative and based on partly attributional and consequential methodology. Those are using average data for each process and seeking to describe a decision consequence by applying marginal data. Allocation is avoided by system expansion, where equivalent marginal systems are replaced. The framework and possible midpoint and end-point impact categories are displayed in Figure 16. The case-specific framework stages are described in the different sections of the present report.

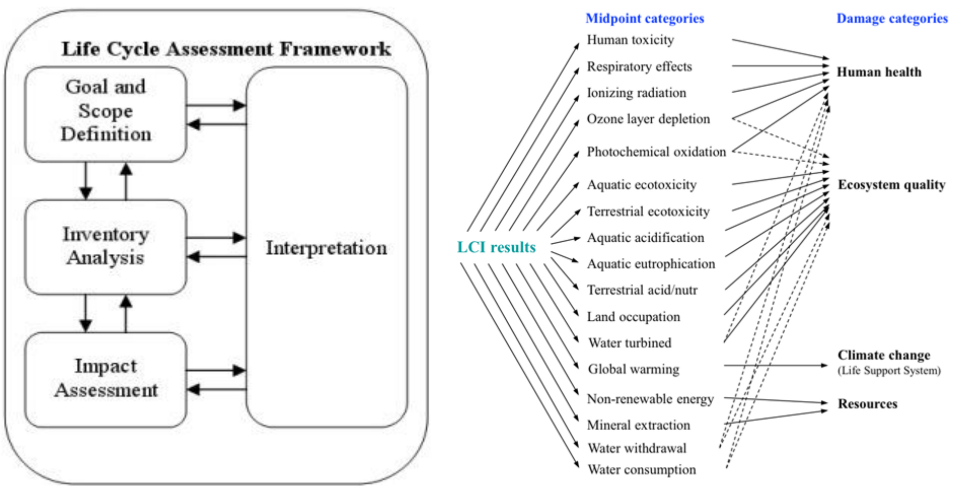

Figure 16. Iterative stages in the LCA framework (left). Disaggregated and aggregated impact categories (right)

7.2 Goal and Scope Definition

7.2.1 Intended application

This project is a part of the BIOTENMARE research project between universities in Norway and Poland and is regarded as a student contribution of joint new knowledge. The report will be disclosed to public. The aim is to perform a comparative LCA evaluating the overall performance of two management systems treating organic waste. System 1 reflects the currently most common practice and System 2 practice is in expansion:

- Reference System (REF): Conventional manure management and incineration in MSWI
- Alternative System (ALT): Anaerobic co-digestion in centralized biogas plants

This study will assist in identifying the optimal handling of organic waste with respect to the waste hierarchy. A key statement is: “In the future, waste incineration will play a less important role and there will be focus on the material resources in parallel with energy recovery. We must recycle more and incinerate less” (Miljøministeriet, 2013a). The purpose is to compare the systems in the context of Denmark (DK) and of Poland (PL) independently within and between each other (Figure 17) in the light of the research questions. This is done through several scenario variants.

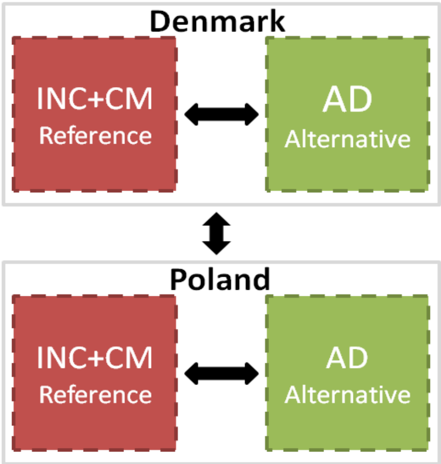

Figure 17. Internal and external comparison of the system performances of Denmark and Poland. The LCA approach is rather comparative (static) than consequential (dynamic)

This observation sets a larger scale decision context of situation B as the study outcome may contribute to the decision on “spreading renewable technologies or not” (EeBGuide, 2012) as a part of the EU 2020 goal.

7.2.2 Functional unit

The primary function of the systems is to dispose of waste and deliver recovered energy (heat, power, and fuel) and material (nutrients). The functional unit is Treatment of 1000 kg of dry matter (DM) organic waste. The reference flow is 1000 kg of DM, constituted by the organic waste types from: (1) pig and cattle manure, (2) household, (3) sewage sludge, (4) slaughterhouse intestines, (5) food from wholesale and (6) frying fat.

7.2.3 System boundaries

Based on principles of consequential LCA, only processes that would react on a change in demand should be included in the system. Any system generating waste would thus be unaffected by the use of waste and therefore these upstream systems are excluded from the system boundaries. This “zero waste approach” assumes that the waste substrates are already generated and are to be collected and treated in the respective waste management systems. This also regards emissions from farming before manure excretion. Furthermore, being a comparative LCA induction of change from diverting waste substrates from other existing treatment systems (lost alternative) is not considered.

Concerning marginals from system expansion, both the REF (Figure 18) and ALT (Figure 19) systems are tested for variants of possible marginal energies (for heat and power). These appear in the form of a present average mix, a future mix, and a single marginal technology characteristic for DK and PL. The systems also include marginal fertilizer assumed to be a mineral fertilizer consisting of N and P, either as a NPK mixture or individually.

In ALT the sorted fraction contains impurities to be removed before processing, but remaining impurities are disregarded from the flow according to the reference flow (1000 kg organic). In reality in the REF system household waste is collected as residual waste (after sorting out main waste fractions). It is assumed that collection takes place in municipalities where household waste is not segregated but incinerated with the residual waste. The REF modelling assumes partitioning principle where only the biodegradable fraction flow is allocated throughout the value chain. This assumption has implications on the physical composition of downstream MSWI residuals, which in reality may be suitable for utilization as e.g. construction material. This way allocation and system expansion is avoided, as the residuals are assumed to consist mainly of organic ashes (assuming complete combustion). Similarly, the chemical use in APC is only attributed to the sewage sludge flow which is potentially rich on heavy metals.

Direct emissions within the system boundary consist of CH_4, N_2O, NH_3, and CO_2 (fossil and biogenic) where $CO_{2,\ biogenic}$ is disregarded. Only emissions from manure collection and storage are identical for both processes but are maintained in both systems for transparency.

Garden/park waste treatment is excluded from the modelling and is solely displayed for comparison. The cut-off criterion is set to 3% of the total resulting impacts. Other relevant aspects left out of the defined system boundaries and modelling concern:

- Carbon (C) faith (tracing and sequestration)
- Potassium (K) in fertilizer
- Organic decomposition efficiency from C/N ratio
- Straw in manure flow from bedding and storage cover
- Change of manure composition due to emissions (mass balance)
- Electricity for composting
- Acid used for storage NH_3 reduction
- Prior dewatering of sewage sludge at WWTP
- Wastewater subjected to WWTP
- Collection bags for household waste (plastic bags govern reject amount)
- Soil and geochemical considerations including pH e.g. for N transformation
- The composting process itself
- Impurities in biogas such as H_2S

- Emissions from landfilled MSWI residues
- Inhouse NH_3 leakage from manure management
- Utilization of APC residues from MSWI (bottom ash, fly ash, gypsum)
- Lifecycle impact from capital (infrastructure: plants, vehicles etc.)

The comparative LCA for DK and PL is a system analysis thus identical processes are not omitted.

7.2.4 The waste management systems

Most assumptions are based on data from the background chapters of this report. The systems are briefly described below. The data used in the various processes and detailed explanation are referred to in 7.4 Inventory Analysis (LCI). The systems are displayed in Figure 18 and Figure 19 as simplified sketch for overview and differs slightly from model design. It includes selected variant flows and processes of which some are tested. The setup of scenarios to analyze in a DK and PL context is tabulated in Table 30 and explained below.

REF system: This system (Figure 18) can be described as the current practice in DK. It consists of two waste flows treating the waste differently. The raw livestock manure flow is generated on farm and stored in-house and outdoor for some time before being spread by a tractor on local agricultural land. The nutrients in manure replace equivalent nutrient amounts in commercial mineral fertilizer. The by-products and waste flow is generated and collected from source by trucks tankers. Household waste is collected by waste collection trucks in urban areas and delivers directly to plant. A given waste mix is transported to thermal treatment in a regional waste-to-energy MSWI. It is modelled as a “black box”.

Utilization 0: Energy is recovered as CHP where the heat and power outputs are to substitute equivalent energy from resources defined by the allocation method. APC residues (fly ash) are hazardous due to the partial treatment of process emissions occurring from particularly sewage sludge, and are transported by truck and ship to landfilling in Langøya, Norway. The bottom ashes are landfilled locally.

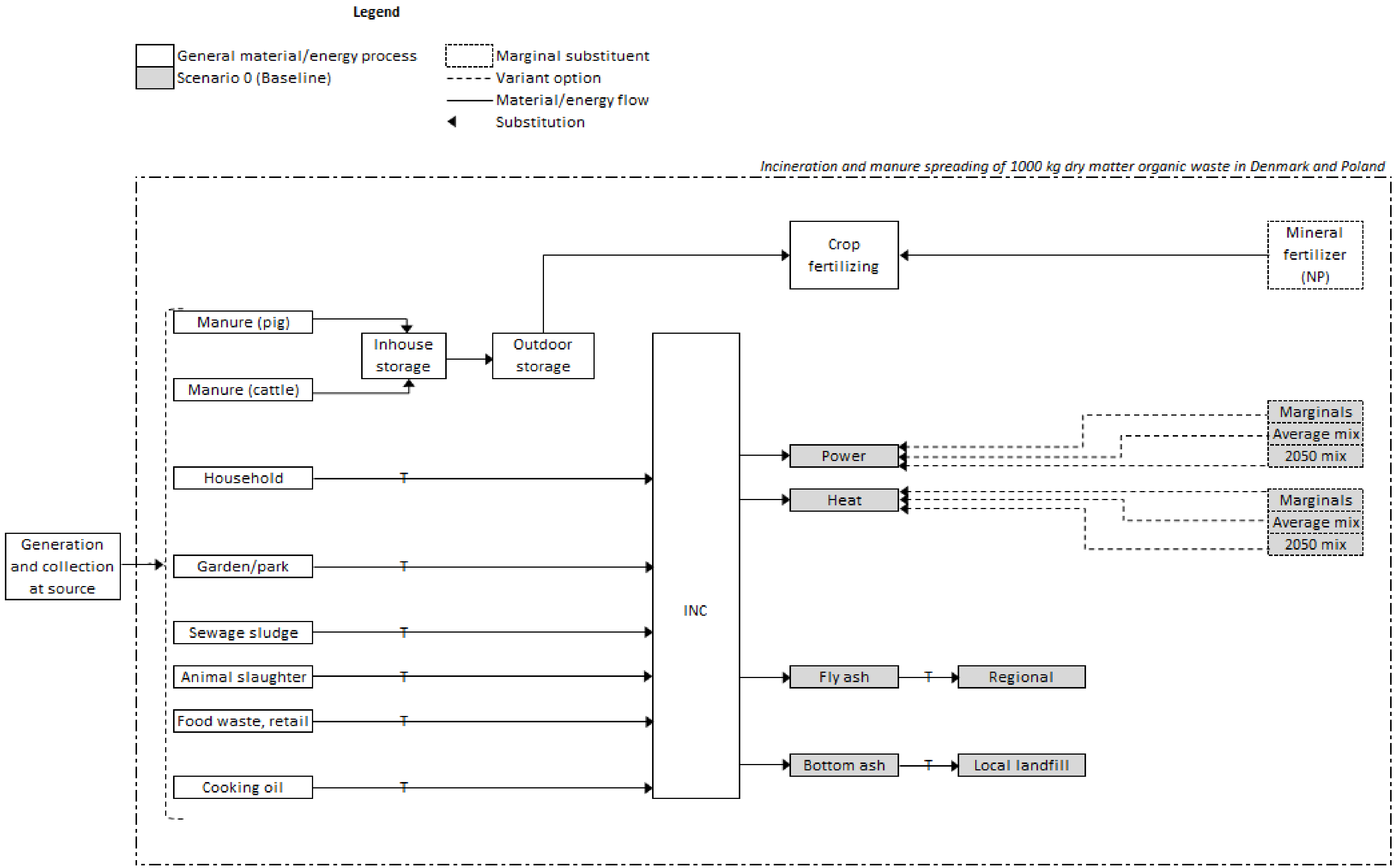

Figure 18. The REF system. Transport (T), nitrogen-phosphorous (NP). Flows without "T" happen in vicinity and transportation is cut-off. Processes outside boundaries are not displayed

ALT system: This system (Figure 19) can be described as the current and emerging practice in DK and PL and it resembles the system previously modelled by Lyng, et al. (2012). As the baseline setup it consists of a given mix of organic wastes substrates from manure and selected non-agricultural organic wastes types collected for co-digestion. The AD plant is strategically placed closer to the source of manure which normally has the largest share in the AD mixture (wet weight).

The manure is generated and stored at source before being transported to AD. The remaining wastes are transported from source to the AD plant. The household organic waste fraction is assumed segregated into paper bags to allow for use of shredder and magnet pretreatment. Impurities are mechanically sorted out at AD and reject is incinerated with recovery. All waste is regulated on site by dewatering or watering to obtain 10% DM. The feedstock mix is then heated for pasteurization and cooled down to thermophilic conditions. The AD plant is modelled as a "black box".

Bioresidual is created as a co-product spread on agricultural land or used as soil improver, both with or without dewatering, and applying only solid or both fractions on land. Raw biogas considered in three utilization pathways. All of them employ prior cleaning from trace impurities:

Utilization 1: The raw biogas recovered is directly combusted in gas turbines on site to produce heat and power which are utilized in respectively the national and local grids, substituting equivalent energy utilities.

Utilization 2: The raw biogas recovered is purified by CO_2 removal on site to produce highly calorific biomethane. Recovered biomethane is injected in the nearby natural gas grid, substituting equivalent energy utilities.

Utilization 3: The raw biogas recovered is purified by CO_2 removal on site to produce highly calorific biomethane. Recovered biomethane is compressed and transported to tank stations or liquefied and distributed to filling stations, substituting equivalent transportation fuel.

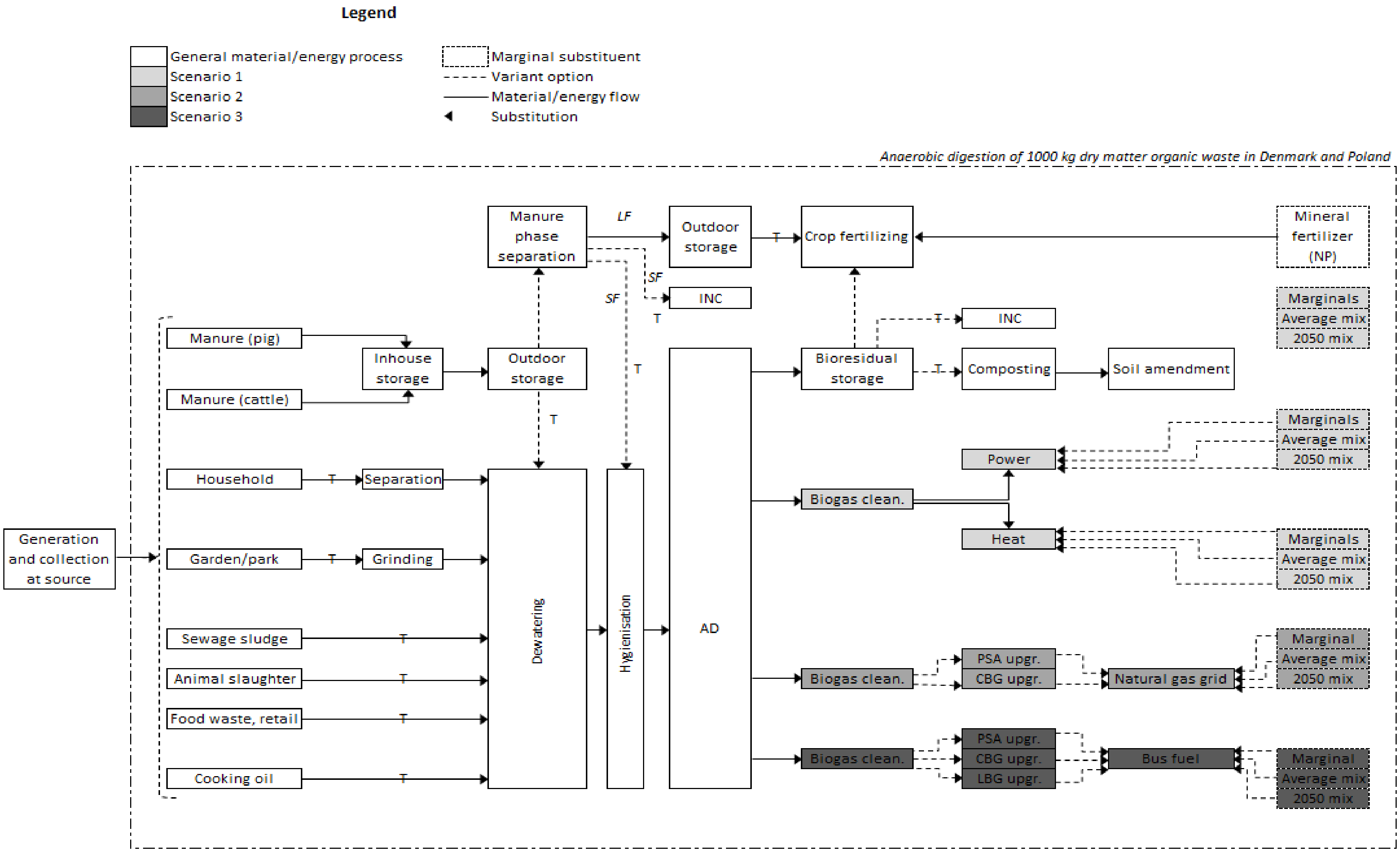

Figure 19. The ALT system. Transport (T), nitrogen-phosphorous (NP), solid fraction (SF), liquid fraction (LF). Flows without "T" happen in vicinity and transportation is cut-off. Processes outside boundaries are not displayed

7.2.5 CASE biogas plants

Both plants employ CSTR technology where the bioresidual does not recirculate.

Lemvig: The largest thermophilic AD plant in Denmark operates since 1992 as a centralized biogas plant (Nort-west Jutland). It receives most types of mixed organic waste. Bioresidual is delivered back to farms which supplied manure fibres and biogas is currently utilized in CHP with a perspective of being upgraded by MeGa-stoRE technology for direct injection in natural gas grid. AD happens at 52°C. Operational data and specifications are published (Lemvig Biogas, 2014). Also, since thermophilic reactors are considered emerging (Christensen, 2011) only this type is modelled.

Horsens: The newest mesophilic AD plant in Denmark operates by Bigadan since 2014 as a centralized biogas plant (South-east Jutland). Located near Danish Crown pig slaughterhouse, it receives (in % DM): cattle manure (16), pig manure (32), slaughter intestinal (21), flotation sludge (4), and bedding and poultry manure (27), representing a feedstock base of 75% farm waste. Since Horsens plant is privately owned, scarce accounting data and operational data are not to be disclosed to public. It is only known that AD happens at 38°C after 70°C pasteurization (Horsens Kommune, 2012). Upgrading is done by DONG Energy but the technology is not specified, only that the removed CO_2 is emitted to atmosphere.

7.2.6 Scenario analysis

Variants can be used to test different factors within the system keeping certain other factors equal (Pöschl, et al., 2010). They can be sorted by sets of variables: within substrate mixture, manure management, upgrading technologies, bioresidual application, and energy substitution. A general setup is in Table 29. Given the waste management systems, principally several modelling approaches can be taken, depending on the perspective of investigation (particularly for ALT):

- Combinations of different fixed variables are tested with reference in each biogas utilization pathway, providing a comparison base between them
- Different scenario variants are created and tested in comparison to a main management pathway (combining most common variables)
- Using legislation as criteria for choosing concrete scenarios

All approaches carry advantages and drawbacks in terms of complexity, flexibility (freedom of choice), and degree of practicality/realism. A combined approach of the above alternatives is found suitable with respect to the research questions, obtaining a balance between being realistic and testing as many different variant combinations as rationally viable.

Table 29 presents a general overview of grouped system variables intended to be tested in various combinations. Table 30 provides a general screening and the groupings are visually illustrated in

Appendix B: LCI data tables. M2, M3, and B3 are not modelled in this project but are displayed as possible variants. Thus all M are assumed to be M1.

Table 29. Chosen possible variations for the REF and ALT systems. Highlighted variables are not modelled

Variant category	Variant code	Description
Feedstock substrate mix (% DM)	F1 [a)]	Pig manure (75), household waste (15), slaughter (10)
	F2 [b)]	Pig manure (50), cattle manure (25), slaughter (21), sludge (4)
	F3 [c)]	Pig manure (10), household (5), sludge (40), commercial (25), cooking oil (20)
Manure management d)	M1	Directly to AD
	M2	Phase separation: liquid (crop fertilizing) and solid (to AD)
	M3	Phase separation: liquid (crop fertilizing) and solid (to INC)
Bioresidual application	B1	Crop/grassland fertilizing
	B2	Composting for soil amendment use
	B3	Incineration
Upgrading technologies and utilization	U1	WS (CGB)
	U2	AS (CBG)
	U3	CS (LBG)
Energy substituent [e)]	E1	Marginal
	E2	Average mix
	E3	2020 mix

a) An example of a likely distribution in DK, manure based and boosted with energy rich co-substrates. Substrates already fully exploited e.g. from food and oil industries are not included
b) Using Horsens feedstock. The straw bedding is assumed proportionally distributed over the manures as it is not a part of the scope. The sludge is slaughter flotation but assumes same management rules as the assumed use of urban WWTP
c) Using a random Lemvig feedstock with emphasis on sewage sludge and cooking oil.
d) Only M1 fulfils the reference flow of 1000 t DM to AD. M2 and m^3 split the manure flow
e) Is mostly different for each biogas utilization pathway (see Appendix B: LCI data table)

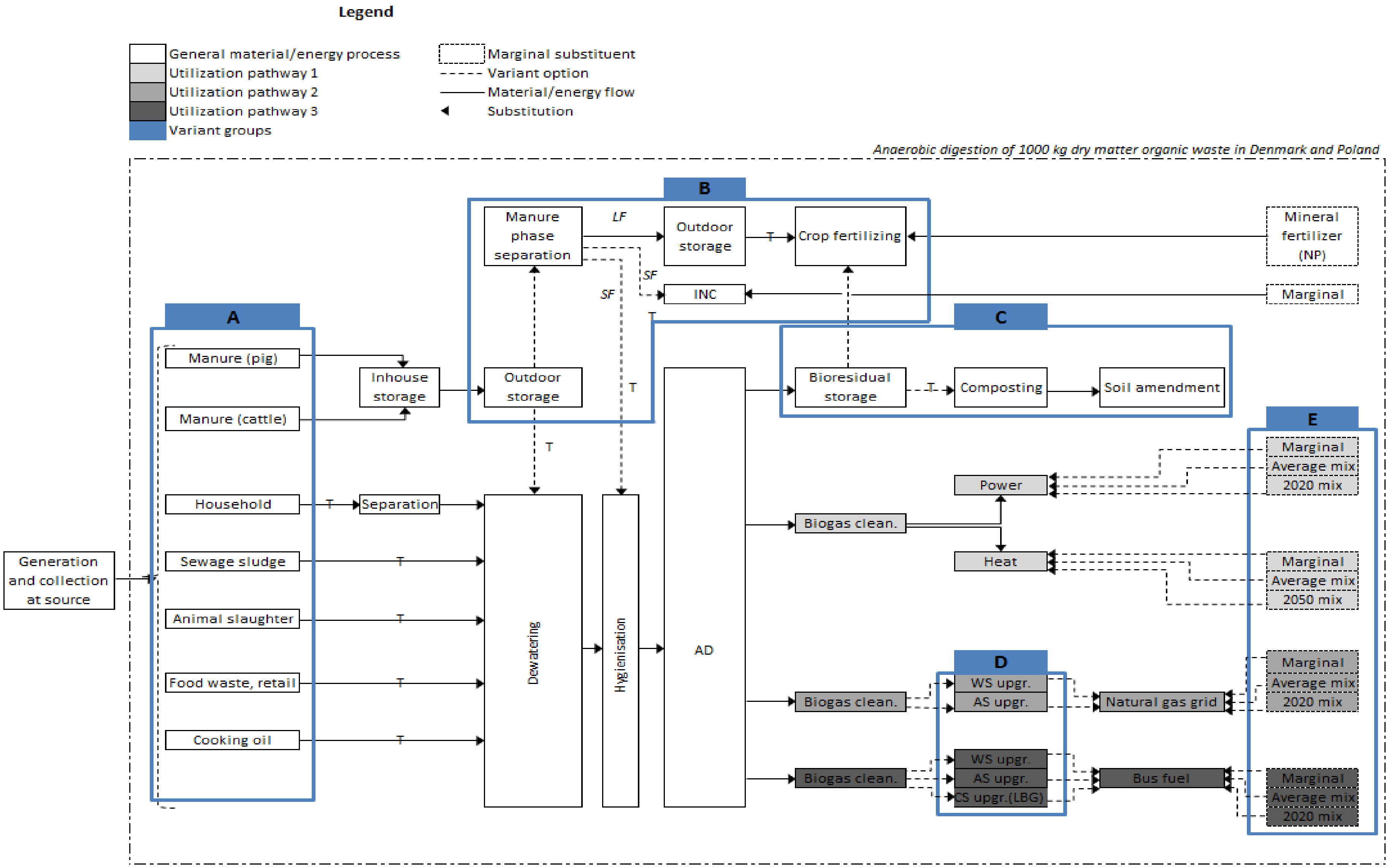

Figure 20. Overview of variants grouped in ALT

7.2.7 Basis for scenario development

Change of material and energy flow pathways is induced (chain reaction) by variant alteration and is primarily governed by legislation[12]:

- F1 and F2 composition decides whether bioresidual is allowed to B1 (nutrient substitution) or B2 (no nutrient substitution). F3 must undergo B2.
- M1 and M2 allow F1 and F2 to preserve the 75:25 rule but not m^3, as the solid part of manure is diverted from AD and bioresidual must undergo B2.
- Using m^3 changes the liquid fraction N-substitution rate from 75% to 85%
- No U means direct CHP recovery after cleaning
- U1, U2, and U3 respectively represent biogas upgrading for natural gas injection, CBG bus fuel injected to grid, and LBG bus fuel transported
- The choice of U thus decides which E composition will be substituted
- All scenarios assume use of VS_{ED} for AD except the incineration scenarios 0a, 0b, 2c using $VS_{ED}+VS_{SD}$ (initially planned)

It is assumed that the sewage sludge from the given WWTP is suitable for AD with respect to its heavy metal content.

Table 30. Summarization of REF and ALT system variables tested in scenario variants. Highlighted variables are not modelled

System	REF		ALT								
Variant	Scenario 0		Scenario 1 TYPICAL			Scenario 2 HORSENS			Scenario 3 LEMVIG		
	a	b	a	b	c	a	b	c	a	b	c
F1			x	x	x				x		
F2	x					x	x	x		x	
F3		x									x
M1	x	x	x	x	x	x			x	x	
M2	–	–					x				x
M3	–	–						x			
B1	–	–	x	x	x	x	x		x		
B2	–	–						x		x	x
B3	–	–									
U1	–	–				x		x			
U2	–	–					x				x
U3	–	–								x	
E1	x			x		x	x	x		x	x
E2	x	x	x						x		
E3		x			x						

[12] Byproduct Regulation, Livestock Manure Decree, and Sludge Decree

7.2.8 Scenario description

The scenario variants are constituted based on the variant codes tabulated in Table 30 and explained below. They are independent from the utilization pathways highlighted in Figure 20. The complete list of processes and parameters chosen specific for each scenario variant is detailed in Appendix B: LCI data tables. Some parameters of current practice and BAT are included as variants and others are tested in sensitivity analysis.

- Scenario 0: The reference system is tested by altering feedstock mix and energy mix. Average electricity is substituted while in (a) marginal heat and (b) 2020 heat mix combined (due to model constraint)
- Scenario 1: Common AD treatment of a waste mix with direct biogas utilization CHP and raw bioresidual spreading. All other variants are fixed so as to test the influence of different energy substitution methods for heat and power
- Scenario 2: Case of Horsens feedstock use. Feedstock mix and energy mix are fixed, testing the importance of biomethane utilization, bioresidual management (and manure management). Different biomethane utilization pathways are tested, substituting marginal natural gas (a) and diesel (b and c). In (a) and (b) the bioresidual is separated in two fractions (a) for later use and (b) liquid is sent to WWTP. Thus the P and N runoffs are assumed 50% less. (c) differs by composting (no nutrient credit).
- Scenario 3: Case of Lemvig feedstock uses all the given feedstock mixes. (a) produces CHP from manure and substitutes the current heat and power energy mix. (b) assumes such a great DM variation resulting in requisite composting while the bus LBG substitutes diesel. Biogas is liquefied for vehicle fuel. (c) is also composted and the bus CBG replaces the diesel marginal.

7.2.9 Allocation method

Energy: According to the research questions the LCA is partly attributional but takes a consequential approach. The system interacts with markets outside the boundary inducing a change in demand by increased output of end products, here energy and material flows. By system expansion equivalent processes delivering same service are substituted. Thereby its impacts are avoided and credited by subtraction original system impacts. There are numerous practices of defining energy marginals in LCA and the mechanisms are not yet well understood (Hamelin, 2013c). Few examples can be:

- The base load resource
- The resource mix reacting to a market change
- The resource mix affected by a particular (local) energy facility
- The single resource first to be out-phased according to the Merit Order
- Average mix of current or future technologies (Fruergaard & Astrup, 2009)

Similarly system energy input from the upstream background process can be composed of different marginal mixes. It is possible to choose among numerous custom made or from database system energy inputs and substitutions, and these can be identical. However, in order to preserve a comparison basis in all baseline scenarios, this model assumes system energy input fixed to *average* electricity mix of DK/PL and *average* heat being central heat from wood pellet (CH) in all scenarios (see Table 35 and Table 37). Initially the “high voltage” transmission type in Ecoinvent is assumed. Not all energy resources offer this choice and it is important to note that there will be energy losses and crediting during transmission depending on high, medium, or low voltage type. It is possible to select separate Ecoinvent “voltage transformation” processes but these observations are not further tested within the scope of this study.

Mineral fertilizer: when the demand on commercial fertilizer market is declining the most costly one is likely to be phased out. There are several types of mineral fertilizers that can be chosen in Ecoinvent, as N and P separately or as NPK mixes (Wesnæs, et al., 2009). Equivalent amounts of N and P will be substituted. The P fertilizer is applied as P_2O_5 and thus 1 kg P substitutes an equivalent molar mass of 2.29 kg P_2O_5 (Wesnæs, et al., 2009).

7.3 Life Cycle Impact Assessment (LCIA)

7.3.1 LCIA methodology

ReCiPe Midpoint (H) methodology comprises 18 midpoint categories. The five ones most relevant from literature and expected most impacted are selected:

- Climate change (kg $CO_{2\ eq}$)
- Fossil depletion (kg oil $_{eq}$)
- Marine eutrophication (kg N $_{eq}$)
- Terrestrial acidification (kg $SO_{2\ eq}$)
- Human toxicity (kg 1,4-DB $_{eq}$)

This methodology accounts Global warming impacts in a 100 year time frame perspective. According to the adopted principles of LCA, biogenic CO_2 emitted is accounted as a having a neutral global warming potential, unlike CH_4 and N_2O which have respectively a GWP of about 25 and 210 $CO_{2\ eq}$ (EC, 2010).

The characterized impact results remain in the form of midpoint score so as to preserve information availability and enable transparent interpretation. Therefore results will not be normalized to personal equivalences nor aggregated into endpoint categories (areas of protection).

7.3.2 About the model

The generic LCA model is developed in SimaPro 8.0.1 software and applies foreground data from research and generic background LCI from the Ecoinvent v3.0 database. The model is based on material flow analysis (MFA) principles including mass balance and substrate flows of dry matter and nutrients. The individual waste substrate inputs are setup as partial flows based on the reference flow (1000 kg DM). These are derived from an equation for biogas production according to characteristics chosen as input. The analytical solutions include interconnected parameters and variables and it is possible to enable or disable the use of alternative technological and managerial processes. These allow for using transfer coefficients for substance flows and testing alternatives. The numerous processes are aggregated in few main processes displayed in LCIA. Therefore cut-off criteria can only be regulated in the LCA model software. A detailed parameter description list is referred to in Saxegård (2015). The model has been subsequently upgraded to include energy consumption on-farm (stirring and pumping), CHP electricity substituting processes, and energy mixes.

7.3.3 MFA models

A simple substrate flow analysis (SFA) of nutrients and energy is illustrated in Appendix E: MFA (Modified scenarios). This is in order to demonstrate the principal influence of incinerating the bioresidual (and of adding garden waste to the feedstock (Table 31) which is omitted from modelling (Table 29). The nutrient and energy transfer during separation and AD conversion are based on literature. Figure 28 shows that the undegraded matter from AD, residual energy potential in VS_{SD}, can be recovered by incineration but with all nutrients lost to ashes which cannot be readily recovered. Oppositely, if directing the bioresidual to agricultural fertilizing, all nutrients can substitute additional nutrients besides from liquid fraction, but the energy potential will be lost (if disregarding C benefits in soil). This could be solved by e.g. recirculating bioresidual to AD. Similarly, incineration of solely solid manure fraction would imply nutrient losses and modest energy recovery. It is also noticed that only a negligible part of VS is lost in soil and the flow distribution will depend on the separation efficiency.

7.4 Inventory Analysis (LCI)

7.4.1 Data provision

Collection focused mainly on quality in terms of accuracy and country specificity for DK and PL. Foreground data obtained in this project are based on primary and secondary research in the form of site-specific measurements, interviews, theoretical-technical specifications, experimental values, and default estimations (Clavreul, et al., 2012). Background data are generic process data from Ecoinvent database. It is chosen to present mostly data of factors found to be important by literature. Two sections describe substrate characteristics and important processes within the categories in Table 29. LCI for modelling has been well organized and sorted with parameters displayed for each scenario variant, worth seeing in Appendix B: LCI data tables. No quantitative uncertainty analysis (e.g. Monte Carlo and error bars) is performed but the following uncertainty analysis is necessary to provide qualitative discussion and serves as base for the sensitivity analysis. Important data quality aspects are eventually summarized.

7.4.2 Representativeness

The three types of scopes are mutually interconnected e.g. technological efficiency improvement depends on time and technology depends on region.

Temporal: The age of reviewed references span 1986-2015 at the time of publishing. A majority is relatively novel (2005-2014). Most data applied in the model originate from studies 2009-2014. The temporal scope is limited to 2020 with an outlook to 2050. Average temperatures are based on recent historical data and do not account for long term trends due to global warming. The chosen modelling methodology considers impact time frame of 100 years.

Geographical: Most foreground data applied are from DK literature with some European data on technologies and few from PL. These comprise national data spanning over the entire country.

Technological: Primary data for conversion technology at a novel DK MSWI plant and the largest thermophilic DK AD plant are used and considered representative owing to long lifetimes of e.g. CHP. Generally newest data from common practice and BAT technologies is applied to ensure timeliness and accordance with the study goal and scope. Average values are applied for other process data. Long term projections on efficiency development as in Fruergaard & Astrup (2011) are not made.

7.4.3 Modelling Denmark and Poland

Being EU member states PL and DK compare but DK is a “benchmarking” country in terms of more sustainable waste and energy practices. It is expected from such well prepared countries to have a simulative role (Behrendt, 2014). Thus in a long term perspective the scope of PL is assumed to be based on the technological scope of DK. Mainly three elements distinguish regional characteristic of DK and PL:

- Substrate composition (biogas potential)
- Climate (governing direct and indirect emissions)
- Energy mix (emissions and saving potential)

In the sections for PL only data specifically found for PL will be described. The remaining is assumed to be for Danish conditions.

7.4.4 Denmark: Description of substrates

Crude datasets for each substrate are compiled in Table 31. The degradability factor and BMP of most substrates are very similar to those given in Swedish Carlsson & Uldal (2009) and Danish Christensen (2011), respectively. Despite the slightly lower VS/DM values in all Danish substrates, the data seems to reveal similar regional characteristics of the waste. Table 31 shows that most datasets are consistent with its source of origin. The degradability is defined by VS_{ED} while the remainder is not degraded in the AD reactor.

Table 31. Dataset for different DK substrate components (per 1000 kg ww). Exceptions from main references are indicated next to the particular values. Highlighted areas are omited from modelling

Category	Waste substrate	DM (kg)	VS (kg)	VS/DM (%)	VS_{ED}/VS (%)	BMP (m^3 CH_4/kg VS)	CH_4 (%)	HHV (MJ/kg)	N (kg)	P (kg)
Manures	Pig manure a)	74.8	60.7	81	60	0.26 d)	65 i)	0.55 l)*	6.0	1.21
	Cattle manure b)	125.7	104.2	83	37	0.21 d)	65 i)	0.52 l)*	6.87	1.02
MSW	Household a)	315	259.8	82	64	0.33	65 i)	3.04 n)	8.79	1.29
–	Garden a)	609	517	85	68	0.203	60 e)		3.41	0.67
–	Sewage sludge c)	250 j)	225	90	19	0.25 g)	65 i)	4.5 c)*	10.0 h)	6.5 h)
IOW	Slaughterhouse d)	152	140.6	92.5	93 f)	0.375	63 e)	1.3 l)*	5.15	1.1
	Food, wholesale a)	244.1	228.4	94	57	0.277	63 e)	3.6 m)	8.06	1.12
–	Cooking oil e)	900	900	100	100	0.757	68 e)	38.3 k)	0	0.015

a) (Hamelin, et al., 2014) (Supporting Information)
b) (Hamelin, et al., 2011) (Supporting Information)
c) (Christensen, 2011)
d) (Jensen, 2015)
e) (Carlsson & Uldal, 2009).
f) (Jørgensen, et al., 1986)
g) (Miljøministeriet, 2014)
h) (Niero, et al., 2013), estimated
i) (Jørgensen, 2009)
j) (Miljøministeriet, 2013b)
k) (Khalisanni, et al., 2008)
l) (Miljøministeriet, 2004)½
m) (NIRAS, 2004)
n) (Jørgensen, 2003)
* LHV, multiplied by 1.3 in model

Table 32 compiles substrate heavy metal content. Primary focus is on sewage sludge having the highest heavy metal concentrations being imperative for its application. The unit of slaughter waste and sludge are identical showing the contrast in concentration. The unit of remaining substrates indicates the negligible amount of heavy metals per FU. Datasets for a small scale and large scale WWTP sewage sludge are presented for comparison. These concentrations are provided as criteria for further bioresidual treatment and are also included in the model.

Table 32. Dataset of heavy metals for sewage sludge and intestinal (mg/kg DM), and others (kg/1000 kg ww). Exceptions from main references are indicated next to the particular values. Minus indicates "not available"

Waste substrate	Cu	Zn	Pb	Cd	Hg	Ni
Pig manure a)	0.0304	0.0891	–	–	–	–
Cattle manure b)	0.0116	0.0224	–	–	–	–
Household a)	0.009	0.022	–	–	–	–
Garden a)	0.008	0.039	–	–	–	–
Sewage sludge c)	183.0	620.0	32.7	0.972	0.587	21.4
	238.0	835.0	44.9	1.36	1.18	28.0
Slaughterhouse d)	1.2	90 e)	< 1.0	< 0.25	< 0.01	< 1.0
Food, wholesale a)	0.002	0.009	–	–	–	–
Cooking oil f)	?	–	0	0	?	0

a) (Hamelin, et al., 2014) (Supporting Information)
b) (Hamelin, et al., 2011) (Supporting Information)
c) (Niero, et al., 2013)
d) (Lukehurst, et al., 2010)
e) (Jørgensen, et al., 1986)
f) (Khalisanni, et al., 2008)

A note on HHV: Higher heating value (HHV) is applied due to the technical nature of the LCA model. All data except for cooking oil are DK. It is unknown whether some calorific values found are HHV or LHV. It is possible to estimate HHV from LHV and vice versa by several methods (Christensen, et al., 2003; Christensen, 2011) but require chemical composition data and resources are very limited. Generally the difference can vary more or less making them fairly uncertain. The found LHV values are increased a factor 1.3 in the model and some will undergo sensitivity analysis for Scenario 0. Especially sewage sludge is uncertain due to different treatment practice (Niero, et al., 2013).

Pig manure: Ex-animal data is from Wesnæs, et al. (2013) is based on updated DK manure standards (Hamelin, et al., 2014) and thus are slightly higher than in e.g. Wesnæs, et al. (2009), Hamelin, et al. (2010) and Hamelin, et al. (2011). VS degradability ranges from 46% (Miljøministeriet, 2014) to 60% (Hamelin, et al., 2014). VS/DM ratio is fairly the same in all three sources, as well as is N ranging between 4.9 kg (Horsens Kommune, 2012) and 5.4 kg (Hjort-Gregersen & Petersen, 2011), and P ranging between 0.8 and 1.1 in the same references. Thus N and P values from Hamelin, et al. (2014) are considered representative averages. Horsens Kommune (2012) reports a DM value of 5.5% but according to Jensen (2015)[13] incoming manure varies between 1-16% DM with a mean value of 5.9%, thus a representative value is assumed.

[13] All data are published in Horsens Kommune (2012) but Jensen (2015) additionally reports the BMP and VS/TS values, by which VS can be estimated. Here DM, N and P values are slightly changed due to analyses performed by Bigadan itself. The method is DM at 105 °C and VS after 505 °C (Jensen, 2015)

Cattle manure: Wesnæs, et al. (2009) and Hamelin, et al. (2010) reported 10.3% DM ex-animal. Horsens Kommune (2012) reports 9% DM, and N and P of 6 and 1 kg/t. Jensen (2015) reports 7.9% DM with 80% VS/TS (6.3% VS), and N and P of 4.6 and 0.6 kg/t. Similarly, N and P values are 4.3 and 0.9 kg/t in Hjort-Gregersen & Petersen (2011). VS degradability is also in very good accordance across references, being 37% (Deublein & Steinhauser, 2008) and 35% (Miljøministeriet, 2014) from which BMP in the range 100-300 m^3 CH_4/t VS covers the applied value.

Household: Jørgensen (2003) found source-sorted household waste to 27% DM and 86% VS/DM. The fraction is assumed to be impurity separated. BMP is calculated as an average from three Danish studies. This is in agreement with (Davidsson, et al., 2007) who found a BMP of 300-400 m^3 CH_4/t VS from sorted household waste in several Danish cities. On the other hand Hansen, et al. (2007) found an average BMP of 459±6% m^3 CH_4/t VS. If collected in plastic bags the plastic content reaches 10%, sometimes up to 20% ww. However, paper bag collection is assumed initially.

Sewage sludge: Heavy metal and nutrition dataset of the smallest and largest DK WWTP[14] (Niero, et al., 2013) were chosen as contrasting candidates (Table 32) for comparison in relations to DK heavy metal law on individually collected wastes. In the small WWTP only Cd slightly exceeds the allowed 0.8 mg/kg DM while the remainders stay well below. Commonly the mean values of Cd and Hg in DK sludge exceed (Jensen & Jepsen, 2005) but the geometric standard deviation of Cr is 1.66 implies a probability of limit value compliance among several samples (Niero, et al., 2013). Average sludge DM can vary 3.5-5.5% (Christensen, 2011; Hamelin, et al., 2011) and is dewatered to around 25% DM (Miljøministeriet, 2013b) and assuming no loss, VS is estimated from VS/DM. BMP ranges 160-350 and 250 m^3 CH_4/t VS is chosen as a mean value, also staying within 220-430 m^3 CH_4/t VS (Luostarinen, et al., 2011). Average N and P values in the small and large WWTP are respectively 40 and 26 against 42 and 37 g/kg DM (Niero, et al., 2013).

Slaughterhouse: Fattening pigs (30-100 kg) constitute 70% of the total amounts of pigs in Denmark (Wesnæs, et al., 2009). Thus the intestinal data (offal) from the 90 kg pigs are used (Jørgensen, et al., 1986) though it may have been more representative to use weighted average of both the 20 kg and 90 kg pig intestinal data provided. Total P is estimated from DM and is similar to Jensen (2015). (Palatsi, et al., 2011) found DM, VS, and N from triplicates in pig stomach (in g/kg ca. 183±8, 180±8, and 12.4±0.7) with overall BMP from animal byproducts in the range of 270-300 L m^3/kg COD. The pig by-product data is thus in reported ranges. Horsens Kommune (2012) reports DM as 15.2% from DC Horsens pig slaughterhouse. DM also varies but not as much as manure (Jensen, 2015). BMP though is somewhat different from other sources e.g. 700 m^3/t VS (Miljøministeriet, 2014).

[14] Treating respectively by aerobic stabilization with agriculture application, and anaerobic digestion following incineration

Food, wholesale: Catering food is reported with a DM of 220 kg/t and VS of 198 kg/t (NIRAS, 2004). Also DM and VS/DM for catering, supermarkets and detail are found as respectively 32 and 79, 29 and 94, and 13 and 90 % (Jørgensen, 2003). All the DK data are in fair accordance.

Cooking oil: The cooking oil is already exploited for biodiesel production and economically more rentable(Olesen, 2014). However, it is interesting to test its influence in this LCA. The calorific value is realistic compared to other substrates as the fuel oil value is about 40 MJ/kg (Miljøministeriet, 2004). Used cooking oil is assumed to have no regional difference in composition and only depends on the raw material and use.

7.4.5 Denmark: Description of other parameters

Transportation: Manure delivery of 8 km is based on legislative aspects and other literature (Wesnæs, et al., 2009; Vega, et al., 2014) while compost distance to the nearest forest/park is assumed 16 km. The same applies for fresh bioresidual which is assumed spread on the same fields of the manure origin. Slaughterhouse distance to AD is 60 km (Horsens Kommune, 2012). All organic waste is based on this distance to enable comparison of load per tkm. Bottom ash is assumed landfilled locally while fly ash is shipped to Langøya, Norway (about 2500 km from central DK and PL). Transport of LBG to a filling station is around 100 km from Lemvig (Stenkjær, 2012). No transportation and pipeline injection losses of CBG are considered (Poeschl, et al., 2012a; Rehl & Müller, 2013). The same distances in ALT apply for REF.

Material emissions: CH_4, N_2O, and NH_3 occurring during manure and bioresidual storage and application on land are estimated by co-student Saxegård (2015) mostly from Amon, et al. (2006). These data are applied in the modelling of DK and PL due to the technical configuration of model. There is a relative high uncertainty on the data from Amon, et al. (2006) due to origin of data and estimations. Otherwise, detailed LCI for DK and PL can be found in literature (Hamelin, et al., 2014; Wesnæs, et al., 2013; Vega, 2012; Skura, et al., 2013; Frandsen, et al., 2011). In Hamelin, et al. (2014) Supporting Information shows very low emissions of N-compounds during storage and field application (below 0.7 kg per t manure), while CH_4-C is 0.54 and 1.80 during storage. Also P leaching is 0.060 kg/t manure estimated from the P concentration which is almost twofold compared to Skura, et al. (2013).

Incinerator efficiency: The newest incinerator from 2012 (ARC) next to the existing one at Amager, Copenhagen Region, has an installed energy efficiency of 72% heat and 28% power (Meyer, 2014). Regarding internal energy use, specific heat has been reported (70 kWh/t waste) but is somewhat uncertain because of its waste specificity (McDougall, et al., 2008).

Biogas plant efficiency: Lemvig has two biogas engines, from 2005 and 2013, with CHP efficiencies of power (39.9% and 42.7%) and heat (44.4% and 44.0%). However, the energy is delivered to the municipality CHP with engine efficiencies of 43% and 57% (Kristensen, 2015). These are applied.

Separation technology: Decanter centrifuge is the preferred option in Denmark is used on farm for separating manure into a liquid and solid fraction. It has been acknowledged by farmers for several benefits, such as being able to increase BMP a factor 5-6 in solid compared to raw slurry and enable better N/P ratio and N concentration in liquid resembling mineral fertilizer (Baltic Compass, 2015). Only decanter centrifuge is modelled as dewatering technology for bioresidual.

Table 33. Energy and efficiency characteristics of selected manure separation technologies (Bauer, et al., 2013). Highlighted areas are omitted from modelling

Technology	Energy use (kWh/t input)	DM in solid (%)	N in solid (%)	P in solid (%)	Polymer (kg/t input)
Decanter centrifuge (PAM) a)	2.18	87.2	41.9	90	0.9
Screw press a), c)	1.45	29.6	6.8	9.1	–
Rotating belt conveyor b)	1.2	72	42	79	–

a) (Hamelin, et al., 2010)
b) (Wesnæs, et al., 2013)
c) (Hamelin, et al., 2011)

Biogas upgrading technology: According to Bauer, et al. (2013) DK possessed only one biogas upgrading facility as of 2012. The WS (40%), PSA (25%) and the AS (25%) technologies used to dominate the market but an increasing number of MS (4%) and CS are emerging (Niesner, et al., 2013; Bauer, et al., 2013). The energy consumption and purification efficiency of upgrading technologies is collected in Table 35. Most options consume in a range of 0.20-0.30 kWh/m^3 raw biogas. Only the amine scrubber has a high heat demand in addition. Average values are applied from given ranges, which are fairly narrow (Bauer, et al., 2013). Niesner, et al. (2013) also studied academic and industrial literature (age 2006-2012) for most of the upgrading technologies. The values seem to be in fair accordance between the two sources, also regarding biomethane purity. CS has scarce reliability in reality but assumes recovers and substitution of food grade CO_2 for industrial purpose.

Table 34. Energy and efficiency characteristics of selected biogas upgrading technologies (Bauer, et al., 2013). Highlighted areas are omitted from modelling

Technology	Energy consumption (kWh/m^3 biogas)	CH_4 slip (%)	Water consumption (m^3/m^3 biogas)	Chemicals (kg/m^3 biogas)
Water scrubber (WS)	0.27	1	0.00022	–
Pressure swing (PSA)	0.25	2	0	–
Cryogenic (CS)	0.22	5	0	–
Amine scrubber (AS)	0.13 + 0.55	0.1	0.00003	0.00003
Membrane (MS)	0.23	0.5	0	–

Energy substitution: The choice of complex marginal types potentially creates high uncertainty when modelling (Cherubini & Strømman, 2011). Marginals are selected in Table 35. Short-term marginal assumes the local energy resource which is first to be outphased. Average mix is defined by the newest Ecoinvent database. The 2020 mix is estimated from data in different sources (Appendix A: National energy data) assuming the same distribution of fossils as currently used. The transportation sector assumes the market for diesel bus vehicles thus gasoline-replacing renewables are left out. In 2020 half of the heating/cooling energy is used for district heating which is assumed here assumed as heat. The projected 2020 mix dataset is fairly uncertain in a decision-making context. Also, the timeliness of DK data is in reality slightly distorted as DK is already ahead of planned renewable shares (Brix, 2015).

Table 35. Three energy allocation types applied for DK for different energy sectors. 2020 mix H and P is roughly estimated by normalization based on given sources. Highlighted areas are omitted from modelling due to the specific model configuration

Allocation type	CHP			
	Power	Heat	Natural gas grid	Transport
Marginal [a)]	100% coal (condensing power)	100% natural gas	100% natural gas	100% diesel
Average mix	Current electricity mix is applied from Ecoinvent database	Assuming heat from (CH) biomass plant	–	–
2020 mix [b), c), e)]	1% oil 14% natural gas 33% coal 22% biomass 30% wind	31% coal 29% natural gas 40% biomass	–	90% diesel 10% bio-diesel

a) (Hamelin, et al., 2013b)
b) (Klima- og Energiministeriet, 2010)
c) (IEA, 2011a)
e) (Danish Energy Agency, 2014)

Mineral fertilizer substitution: Many different mineral fertilizer types exist and legislative substitution rates and plant uptake rates regulate the use (Hamelin, et al., 2011). The N and P content in manure and bioresidual can replace production of equivalent amounts mineral fertilizer. In DK the marginal is considered ammonium nitrate and diammonium phosphate (Hamelin, et al., 2014) but are modelled as background provision of single N and P_2O_5.

Temperature: The temperature may have influence in some processes such as heating of feedstock in AD and GHG formation and emission from manure and bioresidual management. The average DK temperature is reported as 8°C in DK (Hamelin, et al., 2014). The official mean for 2010-2012 is 8.3°C and reaches 8.8°C for the past decade, showing some fluctuations (DMI, 2012). The outdoor temperatures are used to model the initial temperature of feedstock prior to preheating, and also to determine GHG emission rates from manure and bioresidual management as given in the IPCC methodology (IPCC, 2006). DK and PL temperatures are both assumed to be 8°C in baseline regarding input feedstock and storage CH_4 emission. Sensitivity is analysed.

Poland: Description of substrates

Some substrates are PL specific while others are assumed to be based on DK values. An overview is displayed in Table 36.

Table 36. Dataset for different PL substrate components (per 1000 kg ww). Exceptions from main references are indicated next to the particular values. Data assumed same as DK is highlighted

Category	Waste substrate	DM (kg)	VS (kg)	VS/DM (%)	VS_{ED}/VS (%)	BMP (m^3 CH_4/kg VS)	CH_4 rate (%)	HHV (MJ/kg)	N (kg)	P (kg)
Manure	Pig manure [a)]	71.8	57.3	80	76	0.26	65	0.55*	5.97	1.55
	Cattle manure [c)]	85 b)	82.4	80	37	0.21	65	0.52*	2 b)	1 b)
MSW	Household [d)]	270	235	87	64	0.33	60	3.04	8.79	1.29
–	Sewage sludge [f)]	250	155 [e)]	62 [e)]	19	0.25	65	4.5*	4.06	0.035
IOW	Slaughterhouse [c)]	152	140.6	92.5	93	0.375	63	1.3*	5.15	1.1
	Food, wholesale [c)]	244.1	228.4	94	57	0.277	63	3.6	8.06	1.12
–	Cooking oil [c)]	900	900	100	100	0.757	68	38.3	0	0.015

a) (Skura, et al., 2013)
b) (Sindhöj, et al., 2013c)
c) DK data (Table 31)
d) (Pawlowski, et al., 2013)
e)(Sosnowski, et al., 2003), VS/DM estimated from given data
f) (Oleszczuk, 2006)
* LHV, multiplied by 1.3 in model

Pig manure: PL specific data considers physical properties of ex-animal manure. Due to practice differences much higher DM after on-farm storage is reported (Skura, et al., 2013). Ex-animal data is used as reference to DM and nutrient losses from emissions. Furthermore, manure composition significantly depends on the feed (Møller, 2012; Hamelin, et al., 2010; Olesen, 2011) but is not further considered.

Cattle manure: PL specific data considers physical properties of ex-animal manure. Only ex-storage data is reported in Sindhöj, et al. (2013c). Data from two farms are given and with seasonal error bars and the mean for slurry in first farm is used. It is initially assumed that the material losses from emissions do not influence considerably on the results.

Household: PL has significantly increased separate collection of organic MSW. It is perceived that Polish household waste is relatively wet (den Boer, et al., 2010). Thus the physical composition of source-segregated organic MSW is assumed from Pawlowski, et al. (2013) as average values.

Sewage sludge: Sampling of nine PL municipal WWTP sludge (after MBT) shows that heavy metal concentrations vary considerably but all stay within the PL regulation limits (Oleszczuk, 2006). Pb ranges 16.2-38.5 and Cd ranges 1.08-9.50 mg/kg DM. The nutrient and heacy metal composition is also reported to be similar to other European studies. Average N is found to 40.1 g/kg (28.0-68.6) and P is 24.2 mg/kg (14.6-35.4). The sludge flow with highest P content is selected. Since the sludge has been MBT treated the VS can be assumed of the sludge mixture in Snosowski, et al. (2003).

Slaughterhouse: Assumed same as DK data. Uncertainties are expected to be on same level as pig intestinal composition is influenced by livestock feed which is similar in DK and PL (Sindhöj & Rodhe, 2013b).

Food, wholesale: Assumed same as DK data. Relatively high uncertainty is expected here similar to the difference in household waste.

Cooking oil: Assumed the same as DK data. Regional differences are considered negligible.

7.4.6 Poland: Description of other parameters

The only parameters for PL that differs from DK system boundaries are individually presented.

Energy substitution: Principally the considerations for DK apply for PL. Only the 2020 energy mixes of power and heat are being distinguished (). These are based on recent data from distribution of fossils but assuming that natural gas expansion has reduced coal share. The nuclear energy expansion strategy is more long-term than to 2020 and is not considered.

Temperature: The average temperature in PL reached 6-11°C in PL between 2012-2014 (IMGW, 2015). The DK temperature lies within this range resembling its climate.

Table 37. Three energy allocation types applied for PL for different energy sectors. 2020 mix H and P is roughly estimated by normalization based on given sources. Grey areas are omitted from modelling due to the specific model configuration. Light blue areas is assumed the same as for DK

Allocation type	CHP			
	Power	Heat	Natural gas grid	Transport
Marginal a)	100% coal (condensing power)	100% natural gas	100% natural gas	100% diesel
Average mix	Current electricity mix is applied from Ecoinvent database	Assuming heat from (CH) biomass plant	–	–
2020 mix b), c)	75% coal 6% natural gas 2% hydro 9% wind 8% biomass	14% oil 37% coal 32% natural gas 15% biomass 2% solar	–	90% diesel 10% bio-diesel

a) (Hamelin, et al., 2013a)
b) (Ministry of Economy, 2010a)
c) (IEA, 2011b)
d) (Luostarinen, 2013)

7.4.7 Data quality evaluation

The three segments individually characteristic for DK and PL are analyzed with respect to data quality in order to facilitate the choice of parameters for the sensitivity analysis for PL only, as most of its system definition is already based on DK. Taking the heating value uncertainty aspect out of consideration for substrates, the following quality index criteria for data ranking are given:

- Data source (primary, secondary, assumptions)
- Data quantity (number of sources)
- Data quality (country specificity)
- Data accuracy (deviation)
- Data consistency (agreement with other sources)

Ideally, datasets from single primary sources combined with similar replicates from other sources would be most accurate. The summarization of results is displayed in Table 38.

Summarization: The substrate data generally seem to be of fairly high quality, disregarding the three ones more or less justified assumed for PL from DK. Both average temperature data set slightly deviate but DK and PL are similar why the mean will be initially applied in modelling for both countries. The energy substituent data is commonly accepted for “marginal”, however the natural gas substitution process is based on data from “Rest of World”. It is not known whether the individual “average mix” are updated in the newest Ecoinent v.3.0 database. The forecasted 2020 mix is based on estimations from several sources but is uncertain due to its nature of forecasting and because “2020 heat mix” for PL and DK lacks most of the country specific processes in Ecoinvent which have been based on Switzerland.

Table 38. Data quality divided in three categories characteristic for DK and PL (5 is high certainty)

Substrate (composition)	**DK**	**PL**
Manure, pig	5	4
Manure, cattle	5	4
Household waste	5	2
Sewage sludge	4	4
Slaughterhouse	5	*)
Commercial food	5	*)
Used cooking oil	4	*)
Climate		
Temperature	4	4
Energy mix type		
Marginal	4	4
Average mix, present	4	4
2020 mix	3	3

*) Assumed entirely the same as DK

8 Results

This chapter presents the environmental impact results for the various scenario variants of the waste management systems of DK and PL. These are displayed for five chosen impact categories followed by a sensitivity analysis. The uncertainty aspect inter alia is discussed later. Detailed results for each graph can be found in Appendix C: Raw data results.[15]

Figure 21 shows that both for DK and PL global warming potential is generally caused by the manure storage and application process. CH_4 and N_2O emission are both strong GHG which happen respectively the most during in-house and outdoor storage, and after being spread on fields. In the ALT scenarios bioresidual post-treatment has a significant influence on climate change as well. The only scenario with LBG production (S3b) witnesses that this process is very GHG intensive compared to the production of CBG. In the REF scenarios the highest energy savings stem from substitution of power and heat when the feedstock is rich on a high calorific and less wet mixture. On the other hand, a FU of organic waste dominated by manure (S0a) is able to avoid some mineral fertilizer. However, energy substitution from incinerating high-calorific waste more than offsets the savings from fertilizer.

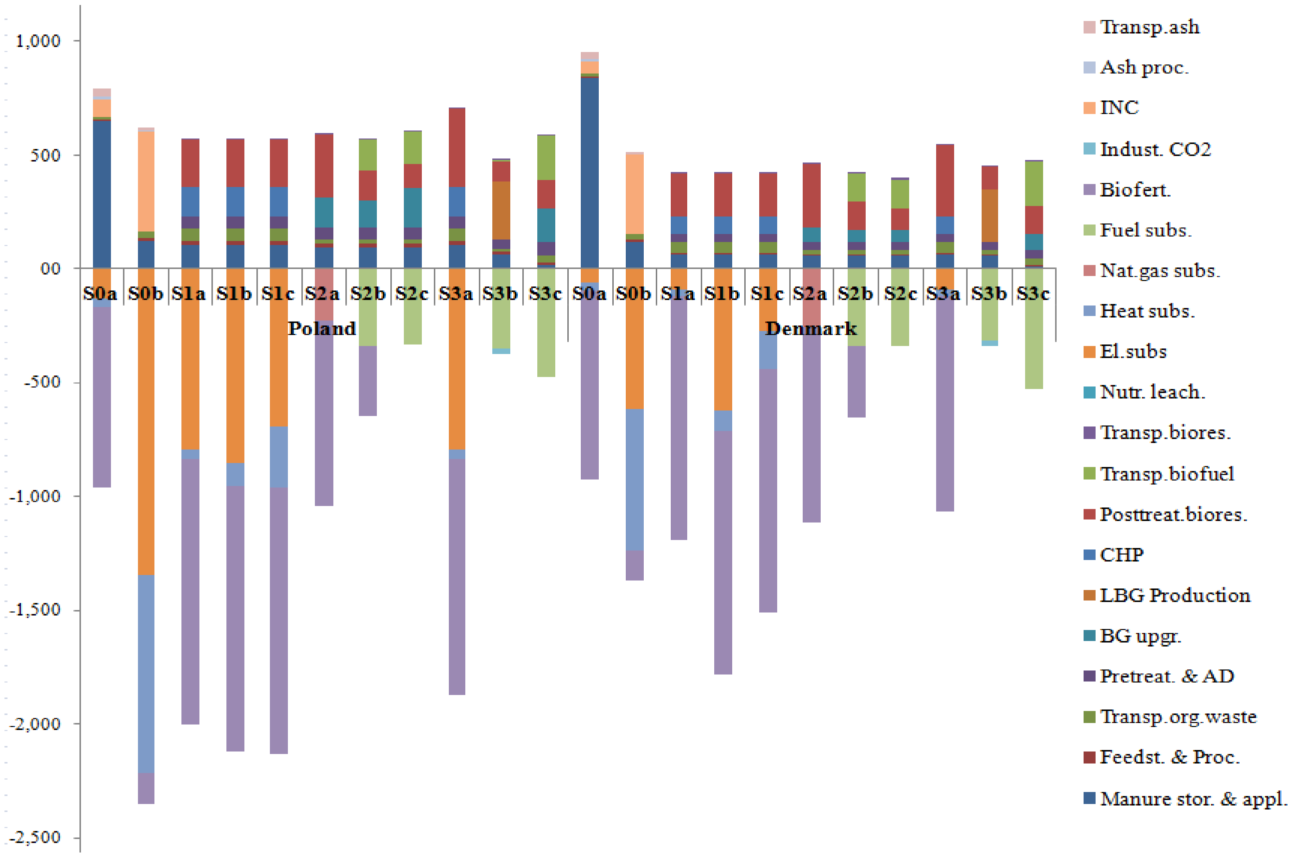

Figure 21. Climate change impact (kg $CO_{2\ eq}$) for PL and DK scenarios

[15] NB: A mistake was untimely discovered regarding input of heavy metal data from feedstock (sewage sludge), due to a misunderstanding of the model function. Therefore the Human toxicity impact category is not displayed in this chapter

Savings from substituting diesel fuel does not seem to be considerable compared to the production load in ALT scenarios while AD with CHP utilization is both able to produce net electricity savings and savings from utilizing bioresidual, given the agricultural substitution rates. Similarly, the substitution potential for grid natural gas is low and more benefit is achieved from bioresidual.

Interestingly, impacts from PL S0a are higher than in DK but vice versa in S0b. Here it requires more electricity input to heat up the wet feedstock unlike S0a where most of it (manure) is diverted to direct spreading. In all scenarios higher electricity savings are obtained by PL because the replaced mixtures are more CO_2 intensive for PL, as clearly seen in S0b. Even when replacing 100% marginal coal the usual provision of these resources are more CO_2 intensive for PL (S1b), thereby a higher GHG saving. This also applies for 2020 energy mixes (S1c) when PL is expected still to have a high share of fossils in the energy system. Comparing S1a-c heat savings it seems like the 2020 mix is more CO_2 intensive (S1c) than e.g. 100% natural gas while the opposite applies for electricity, especially for DK. A summation of net climate impacts is in Figure 22 revealing that CHP based energy systems (S0a-b, S1a-c, S3a) in most cases individually save the most GHG impacts.

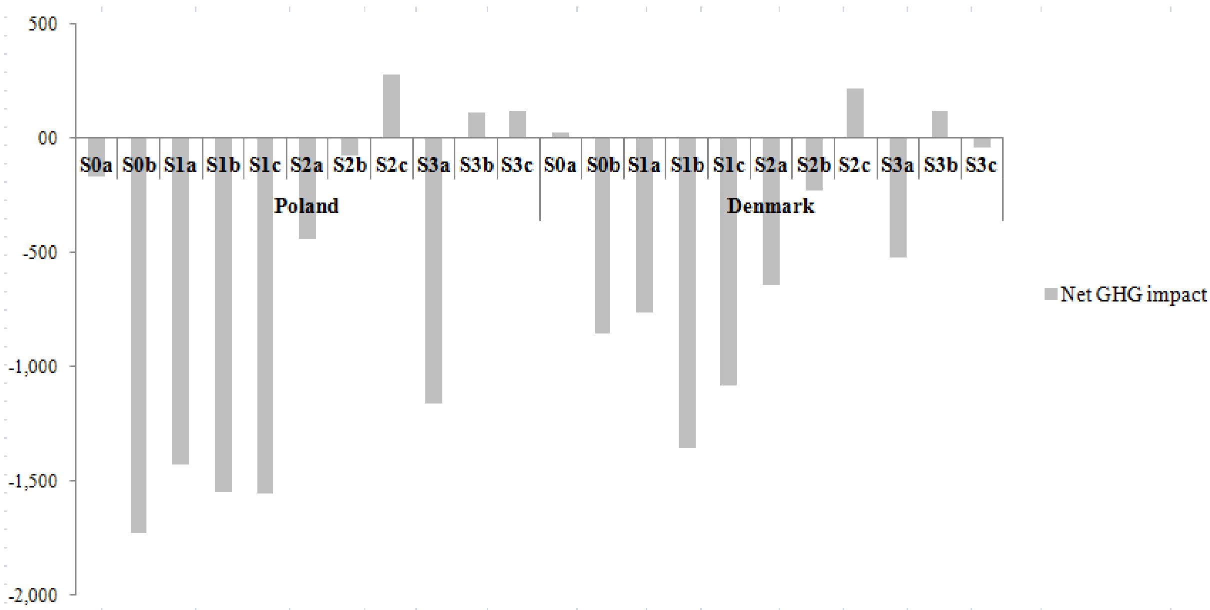

Figure 22. Net climate change impact (kg $CO_{2\ eq}$) for PL and DK scenarios

The terrestrial acidification potential (Figure 23) is most expressed in scenarios with post treatment of bioresidual, especially when it is separated in a liquid and solid fraction and stored prior to spreading (S2a and S3a) unlike direct spreading of bioresidual (S1a-c). The latter does not even apply NH_3 reduction measures during storage, so these can be a cause of the high impacts.

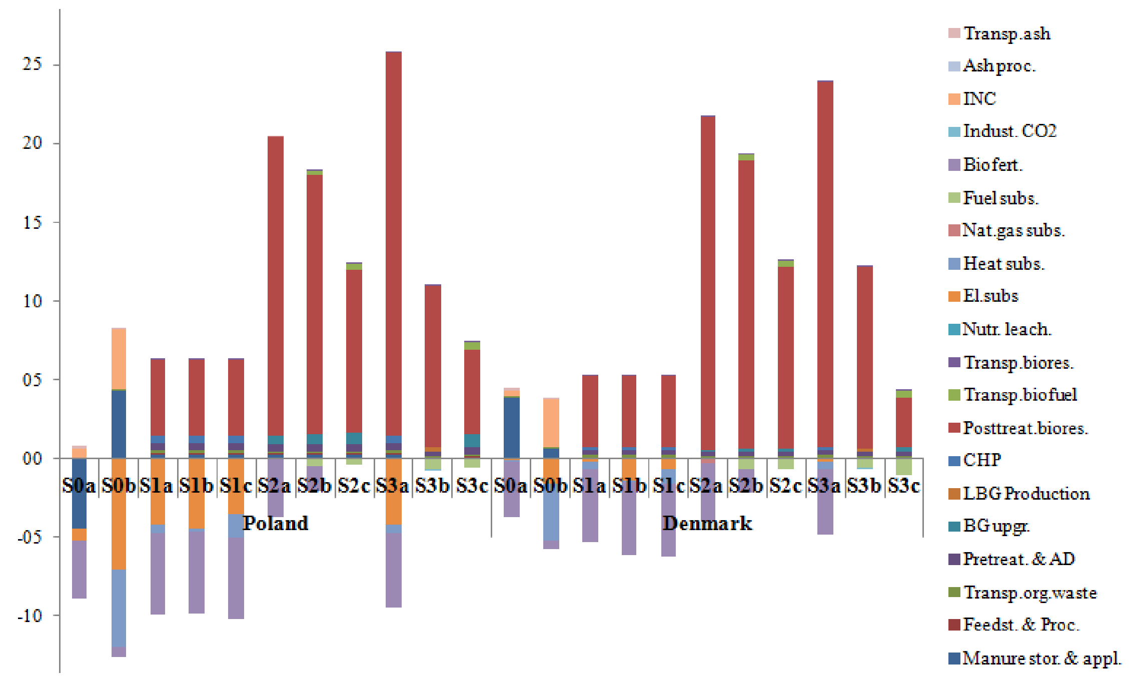

Figure 23. Terrestrial acidification (kg $SO_{2\ eq}$)

Marine ecotoxicity is caused by processes that leach N compounds (Figure 24). The relatively highest impacts stems from scenarios utilizing the bioresidual as raw (S1a-c), or separated (S3a and S2a). Bioresidual post-treatment activities pose about 2/3 impacts. N leaching to aquifer also happens with nutrients during application which are not taken up by plants having a transfer coefficient of 34.2% (Saxegård, 2015). However, this does not appear to be the case when observing REF with conventional manure management due to a lack in the model setup.

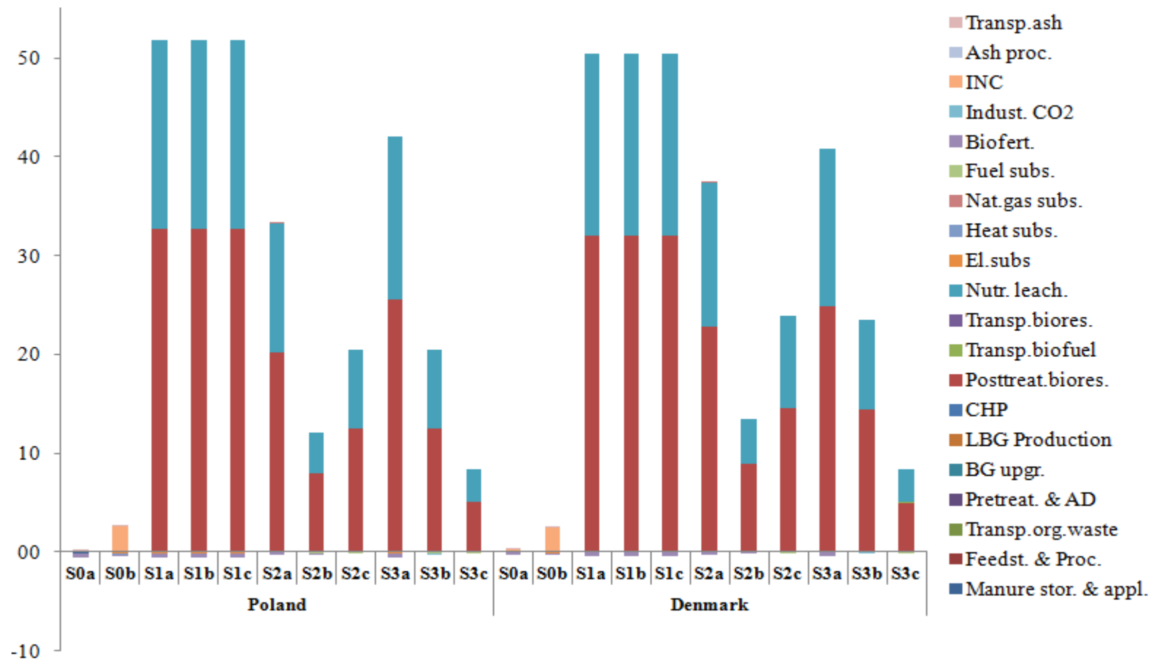

Figure 24. Marine ecotoxicity (kg N_{eq})

Extraction and use of abiotic resources (Figure 25) is correlated with the GHG emissions in Figure 21. The pattern is similar at least concerning avoided CO_2 emissions from different energy resource mixtures containing fossils. For example the highest fossil fuel savings occur from incinerating highly calorific feedstock with recovery of energy to substitute an average electricity mix and marginal heat mix in PL (S0b). This chart indicates the degree of fossil share in the energy mixes, evidently being lower for DK. Overall the savings from replacing diesel for transportation and natural gas from grid are sufficient to offset the impact from producing the substitutive biogas co-product. The other co-product (bioresidual) also provides a significant saving of fossil fuel because the manufacturing of equivalent mineral fertilizers is energy intensive and seems to be based mainly on fossil fuel.

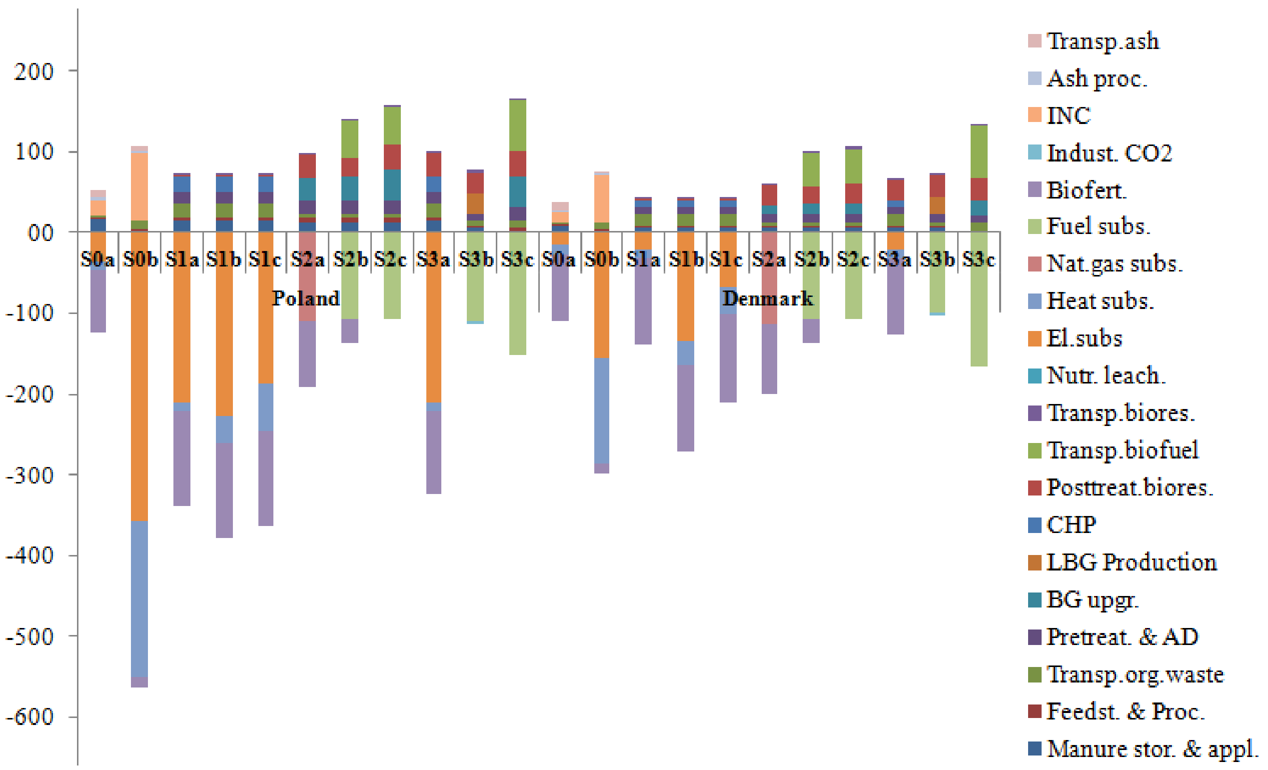

Figure 25. Fossil depletion (kg oil $_{eq}$)

8.1 Sensitivity analysis

Uncertainty check is caused by the need to test parameters that vary considerably in literature, to identify hot-spots in system, and tests the robustness of model results influenced by uncertainty (Clavreul, et al., 2012). A simple sensitivity analysis can be done by changing input parameters and evaluating the reaction of e.g. impact categories. It is also possible to test change of technological parameters to a reference result but combinations of these are expressed in the scenario analysis. Only factors which can be influenced upon in decision making are tested for sensitivity. The choice for testing is based on:

- Result indications
- Data uncertainty and considerably wide ranges
- Literature findings for comparison

The analysis is only performed for the context of PL as data is already based on the DK system and as some PL data are regarded fairly uncertain. Scenarios with their given parameter setup are evaluated in Table 39. Illustrative flow charts for both scenarios are presented in Appendix D: MFA (Sensitivity results).

8.1.1 Scenario 0a

- CHP efficiency: instead of BAT data, applying average of 19.5% power and 65.4% heat (Fruergaard & Astrup, 2011). Power and heat efficiency is not individually tested
- HHV: this is considered fluctuating particularly for sewage sludge assumed to change from 23.4 to 30 GJ/t DM
- Power use: 70 kWh/t is probably the start-up energy, also reported as 77.8 kWh/t for MSW (Zaman, 2010), so the running auxiliary energy is assumed to be 7 kWh/t, though it depends highly on the water content.
- Transport: hazardous fly ash is assumed landfilled locally, 25 km from site
- N availability: For winter crops (spring) is 65% with trailing shoe but drops to 45% for spreading on grass (winter) (Lukehurst, et al., 2010), assuming it is manure

8.1.2 Scenario 3a

- Substrate DM: pig manure is assumed being separated on farm to increase DM from 7.2% to 33.4% (Skura, et al., 2013) using this solid fraction in AD
- Specific heat: more precise is to partition for the 10% DM content of 3.00 MJ/(t·°C), yielding a weighted 4.1 MJ/(t·°C)
- Temperature: outdoor temperature reaches level of feedstock input of 16.8% (Lemvig Biogas, 2014) having implications both on AD and CH_4 emission from manure storage raising from 3% to 4% (IPCC, 2006)

- CHP efficiency: instead of BAT data, applying average of 40% power and 46% heat (Hamelin, et al., 2014). Power and heat efficiency is not individually tested
- Loss of CH_4 from AD: increased from 0% to 10% (Møller, et al., 2009)
- Loss of P from AD: is reduced from 10% to 1% assuming almost conservation
- Synergy effect: factor of 1.1 assumes positive synergy in substrate mix (Pöschl, et al., 2010)
- Loss of NH_3 from spreading: the emission reduction rate decreased in effect from 90 to 30%
- N content: due to novel management practice N is assumed to increase by 10% in substrates
- Separation efficiency: fixing mass ratio, VS, and N and P distribution, the DM distribution in solid decreases from 69.4% to 40.2% (Wesnæs, et al., 2013)

Table 39. Relative sensitivity due to a change in parameter (the change in impacts). Increase (+), decrease (–). Values 5-9% are green and ≥10% red. Values are rounded. Human toxicity is not discussed

Impact category	Climate change (kg $CO_{2\ eq}$)	Terrestrial acidification (kg SO_2)	Marine ecotox-icity (kg N_{eq})	Human toxicity (kg 1,4 DB_{eq})	Fossil depletion (kg oil_{eq})
Scenario 0a (baseline)	-168	-8.1	-0.4	-9.0	-71.0
CHP efficiency	-26%	-3%	-1%	-8%	-16%
HHV	17%	2%	1%	5%	11%
Power use	23%	3%	1%	9%	16%
Transport (ash)	5%	0%	0%	1%	4%
N availability	-114%	-8%	-9%	-44%	-29%
Scenario 3a (baseline)	-1162	16	42	429	-223
Substrate DM	3%	6%	0%	0%	5%
Specific heat	0%	8%	-1%	0%	0%
Outdoor temperature	-4%	8%	-1%	0%	-1%
CHP efficiency	-6%	10%	-1%	1%	-8%
Loss of CH_4 from AD	-29%	10%	-1%	1%	-9%
Loss of P from AD	7%	12%	10%	-1%	5%
Synergy effect	6%	5%	-1%	-2%	9%
Loss of NH_3	0%	5%	0%	0%	0%
N content	5%	13%	10%	-1%	4%
Separation efficiency	0%	0%	-8%	-46%	0%

9 Discussion

9.1 Main findings

The results between DK and PL scenario variants show a clear difference in impacts though different in terms of impact categories. As observed in the result analysis substitution of fossil-rich energy mixtures is beneficial for climate change avoidance. This can be achieved by incinerating dry and highly calorific waste combined with controlled manure spreading, replacing high carbon footprint mineral fertilizers. On the other hand, provision of upgraded biofuel for the natural gas grid or vehicles does not have the same substitution potential as CHP replacing marginal electricity and heat. However, an important aspect not qualified in this LCA is the storability of biogas and especially biofuel for transportation, unlike real-time consumption of CHP energy from the power and district heat grid (Fruergaard & Astrup, 2009).

It is acknowledged that the DM content and heating value of waste in S0b (90% DM) is considerably higher than in S0a (25% DM). More electricity is required to heat up the greater mass but the savings from energy recovery are also great, especially for heat which substitutes the 2020 mix, which is fairly CO_2 intensive for PL. Conversely, S0a electricity input is low due to the low mass treated and the low energy savings can be explained by extensive use of heat for evaporating the water content in feedstock to an optimal level (from higher heating value), combined with replacement of relatively CO_2 low natural gas heat marginal. In S0a the most impacts are caused by emissions from handling the large mass of manure.

The CHP efficiency with the given assumptions in REF has greater savings than in ALT for PL whereas the GHG savings are more closely ranked in DK. In all scenarios of bioresidual utilization this process contributes the most to savings while the energy saving potential (at least for heat) is decreased compared to incineration. Feedstock processing had a negligible overall system impact in all categories as well as has bioresidual transport the nearby farms. Transport of organic waste only contributes to 1-11% GHG of net impacts in all scenarios and is individually lower for DK in scenar-ios with considerable shares of household waste, being more wet in case of PL.

Generally, the terrestrial acidification potentials are slightly higher for DK than PL caused by the higher N-content in feedstock. This is especially noticed in S1a-c where the N transfer coefficient for separated bioresidual is assumed fairly high thereby creating acidic NH_3, especially during storage and somewhat during spreading (Saxegård, 2015). Only these scenarios do not apply methods for dissolving 65% NH_3 by acid addition. As was the opposite case for GHG comparing S0a and S0b, unlike DK, applying PL manure (75%) directly on land actually saves on acidification impact due to less nutrient content. Applying biofertilizer instead of conventional fertilizer (or compost) avoids $SO_{2\,eq}$ emissions as does processes which use electricity that replaces SO_x intensive resources such as coal and wood in the current (S0b, S1a) and 2020 (S1c) electricity mixes for especially PL.

The choice of bioresidual handling method has also proved significant for impacts on aquifer due to N and P leakage. Direct spreading of raw bioresidual poses the highest marine ecotoxicity as the only scenario variants assuming no NH_3 reduction measures during storage. Separation into a liquid and solid fraction impact relatively high while using only the dry fraction and even compost, both dewatered to 30% DM with discharge to WWTP, as only 33.5% N remains in the solid fraction thereby reducing the leakage potential.

Assumptions of difference in feedstock properties and energy system of PL and DK can have considerable implications on the LCA results. This can be seen comparing the net energy savings for both countries in S0b and S3a where average electricity mix is substituted. Electricity consuming processes cause higher impacts for PL because the power input to system was assumed to be national average electricity mix. The fuel substitution is higher for DK because of the generally higher VS/DM ratio in feedstock. Downstream indirect GHG emissions are influenced by CHP operation while for DK the share is dominated by post-treatment of bioresidual emitting mainly CH_4 storage and N_2O during application. Upgrading to LBG (S3b) causes high indirect and direct GHG emissions mainly from CH_4 loss just enough to offset the replaced diesel. Capture and substitution of 20% biogenic with industrial CO_2 during upgrading has an insignificant effect similar to distribution of biofuel. Management and storage of pig manure in S1a-c accounts for 75% DM and having slight less DM in PL manure and slightly higher N content makes the difference.

In the sensitivity analysis for PL, despite having a high DM share in feedstock mixture, pig manure with an individually much higher DM content does not contribute significantly to GHG and acidification savings (3-6%) in S3a. This can be explained mainly by the low content of VS_{ED}. The VS/DM is 50% higher of DK sludge compared to PL sludge, and along with use of energy type for biogas upgrading this can contribute to shifting the performance rankings as seen in S3c (Figure 21).

CHP efficiency combined with marginal energy type substitution is crucial for both climate and fossil depletion whose net impact (savings) drops by 16-26% when the overall CHP efficiency drops 15% in S0a. Similarly, a drop by 14% CHP efficiency in S3a only drops the same two impact categories by 6-8%. However, these are not comparable between S0a and S3a as the heat marginals and heat/power efficiencies are different. In addition, the absolute sensitivity in S3a is higher losing a net saving of -69 kg $CO_{2\ eq}$ compared to -44 kg $CO_{2\ eq}$ in S0a. All in all, the choice of applying BAT or average CHP may be decisive for comparative assessments of results.

A 10% loss of CH_4 from AD decreases GHG savings by 29% due to both loss of energy substituent and enhanced global warming. P loss reduction induces fossil and GHG savings due to replaced mineral fertilizer but due to leakage it impacts terrestrial acidification and eutrophication and similarly with increase of feedstock N-content. Also a 10% increase in synergy effect, solely caused by a proper feedstock mixture and ambient reactor conditions, can lead to GHG savings.

Transportation of MSWI ash did not prove considerably sensitive to longer distances as the minor unconverted organic residual is dry. An example of low parameter robustness is observed from very uncertain data, i.e. when increasing the calorific value (HHV) of dewatered sewage sludge by 28% increases the GHG avoidance by 17% and fossil savings by 11%. This suggests a direct link with the energy production and further depending on factors such as efficiency, system expansion, and internal energy consumption. The latter in terms of power use is almost entirely correlated to efficiency. The importance of preservation of N nutrients is demonstrated when manure plant uptake decreases from 65% to 45% where net GHG savings are more than halved. This is due to the lost potential of CO_2 intensive mineral fertilizer which would otherwise have been substituted. Fossils fuels are not depleted equivalently high as less CO_2 intensive fossil fuels are probably used as manufacturing energy. Also terrestrial acidification and eutrophication are affected due to more leaking.

The same applies for N availability for plants (at least in manure), as a maximum utilization both avoids more mineral fertilizer and marine eutrophication. CH_4 fugitive loss from AD also has a twofold negative effect: not only it enhances climate change it also decreases the substitution potential for other more polluting energies. Power use for incineration also impacts climate but is expected to be less after start-up of plant. The heating value is relatively sensitive to change in incineration, as all the downstream processes and crediting relies on this upstream feedstock energy potential. Other particularly critical variables are P which loses the ability to replace mineral fertilizers if lost in the system.

If the results were to be used for decision making, uncertainties regarding marginal energy type allocation must be considered. A requisite assumption of particularly average heat input (biomass) and marginal heat (natural gas) based on Swiss data (from LCI database) for both DK and PL makes these incomparable. However, the energy type profiles reveal the contribution to the overall system performance. Sensitivity analysis showed that minor fluctuations in manure DM do not seem significant for the system performance confirming the robustness of this parameter.

9.2 Agreement with literature

A far as allowed by the applied methodology, certain findings from related lifecycle studies is compared. The methodological differences of LCA itself seem to limit the possibilities of cross-comparing study findings (Cherubini & Strømman, 2011).

The pretreatment of feedstock containing organic MSW has shown negligible in the holistic system performance of the relevant ALT scenarios but differences agree with Poeschl, et al. (2012a) that organic wastes containing animal byproducts must undergo energy consuming hygienization. They also found no significant impact savings from upgrading biogas to biomethane with diesel or natural gas substitution.

Comparing biogas production with incineration, Hamelin, et al. (2014) and Fruergaard & Astrup (2009) state that storability of biogas in general is an asset. Lyng, et al. (2012) finds that biogas upgrade for diesel substitution combined with all bioresidual spreading yields highest GHG savings, which corresponds to S2b among all biofuel scenarios.

Clavreul, et al. (2012) and Rehl & Müller (2013) stating that the largest savings in AD and INC stems from power substitution, but more heat in INC, which applies for the most scenarios, especially S0b and S1b. This is due to less calorific value of biogas than of waste and due to the different energy recovery efficiencies. Further, Clavreul, et al. (2012) finds water content in waste to be the most sensitive factor for AD and INC, followed by BMP and electricity recovery, which were not individually tested for household waste in this study.

Numerous studies emphasize on CHP energy efficiencies having implications on Climate change and Fossil depletion (Quiros, et al., 2014; Münster & Lund, 2010), verified by present sensitivity analysis. Börjesson & Berglund (2006b) also state that collection and transport of MSW and CH_4 emissions from storage and upgrade are insignificant, although the latter is not transparent from the present results. On contrary, as also found by other studies, fugitive CH_4 emissions from AD can vary considerably and thus have even threefold negative effects in terms of increased GWP, and decreased CO_2 and fossil energy saving potential (Hamelin, et al., 2014; Vega, et al., 2014; Møller, et al., 2009; Rehl & Müller, 2013).

Bioresidual separation efficiencies do not compare with Pöschl, et al. (2010) because they tried out different separation technologies unlike separation methods. The present sensitivity analysis however showed increased impacts on marine eutrophication. DM separation efficiencies show significant sensitivity to terrestrial acidification and fossil depletion, but again due to methodological differences (assuming proportional distribution of nutrients and VS and aggregating this process in result display) it hardens comparison.

When lowering the NH_3 emission factor Vega, et al. (2014) finds that terrestrial acidification is reduced but marine eutrophication is impacted because more N is available for leaching. The first observation is comparable with the present test of lowering field spreading emission reduction, leading to a 5% impact increase. However, marine eutrophication does not seem to be considerably decreased (-0.4%) probably due to the lower N emission factors of separated bioresidual fractions.

9.3 Strength and weakness of method

As mentioned in 7.2 Goal and Scope Definition, there is a trade-off regarding the scenario setup. The present method enables good possibilities for testing scenarios based on realistic assumptions and with multiple variants. This method takes considerably longer time to generate results compared to a setup of few or a single scenario where individual technological parameters are altered. There is a balance of choosing combinations of several variables and defining main scenarios where individual variables are altered, as in Poeschl, et al. (2012a) and Figure 20. The drawback is that generally too many variables are tested in each scenario variant, which does not ease comparison between them. However, good comparison opportunity appears from the resulted figures. The allocation method has shown useful to test different system expansions for energy substitution, which there is no clear consensus on in literature (Hamelin, 2013c) and scientifically defined marginals can also vary between countries (Hamelin, et al., 2013b). Impact savings seem fairly high because of the energy substitution method. This requires further detailed investigation as part of decision making, and consensus about energy crediting must be established and standardized. The particular choice of FU as dry matter substrates for treatment is not often observed in literature. However, since DM is the reference flow it embeds the wet weight for testing of transportation effect and simultaneously DM is the functional parameter along with the criteria for optimal AD (10% DM in total).

The LCA model is generic and allows extensive manipulation with an array of more variables than was tested in this project. There is a trade-off between aggregating processes to be displayed simply and loss of detailed information. Also categorization of processes will have implications on interpretation and comparability. Nevertheless the model may still need to be refined to ease its application. This concerns separating processes of heat and power substitution from system heat and power energy input. Also for future work, to enable INC electricity substitution from certain (custom defined) marginals instead of only letting the allocation being fixed to the system input energy type. Currently the Human toxicity impact category results are misleading because of wrong modelling. The model has possibility for enabling different criteria for heavy metal limit values and the vales entered will represent the Human toxicity impact according to assumed management of bioresidual. The legislation in Norway differs from that in Denmark and Poland and would have implications on a realistic scenario setup as described in 7.2 Goal and Scope Definition. For more convenience it is suggested to enable a function which automatically calculates the heavy metal content based on entered concentrations, especially for sewage sludge, and coupling this with a criteria function that determines suitability of treatment with respect to legislation. Thus the law aspect is integrated in the model.

9.4 Recommendation for further work

Apart from upgrading mentioned aspects in the model, this thesis has set the frames for including an aspect in LCA modelling which is worth testing in a DK context (Hjort-Gregersen & Petersen, 2011; Prapaspongsa, et al., 2010) in terms of literature research and methodology setup. The scenarios are based on reality taking into account how legal mechanisms govern managerial decisions. It was planned to set up scenarios where pretreatment of manure takes place in the form of separation into a solid fraction for incineration and liquid for digestion. In many scenarios during the present modelling feedstock dewatering was necessary after all. Manure separation would be able to avoid this dewatering and possible nutrient loss. However one main aspect would be to test the trade-off between recovering full energy from the fibres against loosing the nutrients to ash, as displayed in Appendix E: MFA (Modified scenarios). Manure separation is also expected to have other qualitative benefits such as better nutrient management where needed. For further work it is also recommended to include considerations of residual VS i.e. C sequestration in soil to investigate if the bioresidual management process will contribute with more benefits for ALT in an expanded system (Fødevareministeriet, 2008).

More extensive research on PL data to obtain a full picture of regional difference is recommended as well and applying more realistic technology efficiencies for PL. It is advised to additionally test parameters which barely can be controlled such as N_2O emission from land. These emissions are caused by plant uptake mechanisms (Christensen, 2011) and can pose the highest GHG impacts and simultaneously have high uncertainty (Møller, et al., 2009; Lyng, et al., 2012; Vega, et al., 2014; Meyer-Aurich, et al., 2012). Also other feedstock types can be introduced such as garden waste, or even extending the study to a consequential LCA ("dynamic" unlike "static") where choices in one scenario will induce changes in another interconnected scenario (Hamelin, et al., 2014). Furthermore, it could be interesting to bring the LCA methodology one step further from midpoint to endpoint categories with scores to facilitate decision making. Finally, it is recommended to critically investigate the qualitative aspect of biofuel storability against higher benefits from direct CHP production.

In order to cope with testing parameter combinations in numerous different scenarios which can possibly take place, more advanced software would be necessary to apply. This can for example take approach in matrix algebra for which the Arda software is suitable. If uncertainty intervals of data are possessed Monte Carlo simulation and error propagation can test the overall confidence for selecting the environmentally best performing scenario.

The trade-off between GHG impacts from energy conversion processes, and soil and aquatic loads from manure and bioresidual management will also depend on political decisions when attributing weighted scores to the characterized impacts, thus this can be the long term step in LCA modelling.

10 Conclusion

This thesis investigated environmental impacts and important parameters and methodological considerations in a lifecycle perspective from the treatment of 1 tonne dry organic matter in a Danish and Polish context. Comparison was made between typical waste management system consisting of incineration and conventional manure spreading, and an alternative anaerobic digestion producing biogas for multifunctional use and bioresidual fertilizer.

The following key points can be drawn:

- Environmental impacts from GHG and fossil depletion are the lowest for REF systems which combust high DM and calorific value waste
- Emissions from manure in agriculture are considerable
- Optimal REF performance depends on the combination of high calorific organic waste for INC and manure per FU supported by on-farm emission reduction technologies. The performance is limited by manure handling emissions
- GHG and fossil impacts are higher for PL than DK due to use of more fossil rich electricity and heat, and similarly savings are higher even for 2020 mix which is more fossil rich
- Technological efficiency of CHP is crucial for both INC and AD scenarios
- An optimal combination of individual heat and power efficiencies can replace a maximum of affected marginals, given the most polluting energy types
- AD systems produce less net GHG impact savings than INC
- AD with CHP is in all cases more efficient than biomethane/biofuel utilization, but the latter is an asset for energy systems regarding storability
- Net GHG and fossil savings from PL AD are larger than DK due to substitution of more CO_2 intensive electricity
- Natural gas substitution yields the lowest net GHG savings among AD scenarios
- GHG savings from bioresidual production directly depends on substrate properties and nutrient recovery
- Fugitive CH_4 emissions from AD process can highly impact GHG emission, fossil depletion, and loss of energy saving potential
- There is a trade-off between LBG and CBG benefits in terms of transportation
- Direct use of bioresidual without soil injection (NH_3 reduction) poses the highest marine ecotoxicity potential. It is also among the highest in terrestrial acidification
- High uncertainty is associated with different inconsistencies between HHV and VS content
- More research on GHG and leachate emissions from fertilizer spreading on land is needed

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Appendix A: National energy data

Table 40. Estimation of renewable energy technology share in total final energy consumption (heating and cooling) in Denmark (ktoe) (Klima- og Energiministeriet, 2010)

Renewable energy technologies	2005	2010	2015	2020
Solar	10	10	14	16
Biomass :	1759	2245	2526	2643
Solid	1714	2178	2426	2470
Biogas	45	59	92	165
Bioliquid	0	8	8	8
RE from heat pumps	100	210	301	370
Total	1869	2466	2841	3028
of which district heating	854	1053	1341	1486
of which biomass in households	700	976	973	948

Table 41. Estimation of renewable energy technology share in total gross electricity generation in Denmark (GWh) (Klima- og Energiministeriet, 2010)

Renewable energy technologies	2005	2010	2015	2020
Hydro	23	31	31	31
Solar	2	2	3	4
Wind	6614	8606	11242	11713
Biomass :	3243	3772	6035	8856
Solid	2969	991	5312	6345
Biogas	283	194	721	2493
Bioliquid	0	0	1	8
Total	9881	12412	17312	20595
of which in CHP	3243	3772	6033	8838

Table 42. Estimation of renewable energy technology share in total in the transport sector in Denmark (ktoe) (Klima- og Energiministeriet, 2010)

Renewable energy technologies	2005	2010	2015	2020
Bioethanol/bio-ETEBE (imported) of which biofuels from Article 21 §2[16]	0	13	95	94
Biodiesel (imported) of which biofuels from Article 21 §2	0	18	152	167
Electrical energy from renewable source	9	11	19	29
Total	9	42	266	291

Table 43. Estimation of domestic biomass energy supply in primary energy production (heating and cooling) in Denmark (TJ). The relevant sources are highlighted (Klima- og Energiministeriet, 2010)

Sector of biomass origin		2015	2020
Forestry	Direct supply (forest)	61,000	77,000
	Indirect supply	7,192	7,192
Agriculture and fisheries	Crops, fishery products, grass	n/a	n/a
	Agricultural byproducts, processed residues	26,000	29,500
Waste	Organic MSW (garden/park, food, kitchen, households, restaurants, caterers/retails, food processing plants)	25,600	28,600
	Biodegradable industrial waste (including paper)	0	0
	Sewage sludge	879	879

[16] biofuels produced from “wastes, residues, non-food cellulosic material, and ligno-cellulosic material”

Table 44. Estimation of renewable energy technology share in total final energy consumption (heating and cooling) in Poland (ktoe) (Ministry of Economy, 2010a)

Renewable energy technologies	2010	2015	2020
Geothermal	23	57	178
Solar	21	176	506
Biomass :	3911	4393	5089
Solid	3846	4118	4636
Biogas	65	275	453
Bioliquid	-	-	-
Heat pumps	25	72	148
Total	3980	4532	5921

Table 45. Estimation of renewable energy technology share in total gross electricity generation in Poland (GWh) (Ministry of Economy, 2010a)

Renewable energy technologies	2005	2010	2015	2020
Hydro	915	2279	2439	2969
Geothermal	0	0	0	0
Solar	0	1	2	3
Wind	136	2310	7541	15120
Biomass :	1451	6028	9893	14218
Solid	1340	5700	8950	10200
Biogas	111	328	943	4018
Bioliquid	0	0	0	0
Total	3787	10618	19875	32400
of which in CHP	1441	1874	3156.5	5069

Table 46. Estimation of renewable energy technology share in total in the transport sector in Poland (ktoe) (Ministry of Economy, 2010a)

Renewable energy technologies	2005	2010	2015	2020
Bioethanol/bio-ETEBE	28	279	334	451
Biodiesel :	15	687	993	1451
of which biofuels from Article 21 §2	0	0	88	132
Electrical energy from renewable source	0	15	23	50
Others (biogas, veg.oil etc.)	-	-	26	66
of which biofuels from Article 21 §2	-	-	26	66
Total	43	981	1376	2018

Appendix B: LCI data tables

Table 47. LCI data table with baseline input values of processes and parameters applied in model. Green indicates parameters for sensitivity analysis; orange are processes varied directly in model. CH4, N2O and NH3 emission factors from manure and bioresidual are not shown

DENMARK and POLAND				REF		ALT								
				S 0		S 1			S 2			S 3		
Process	**Value**	**Unit**	**Reference**	**a**	**b**	**a**	**b**	**c**	**a**	**b**	**c**	**a**	**b**	**c**
General transport														
Transport, truck diesel EURO5	1.056	MJ/tkm	Ecoinvent	x	x	x	x	x	x	x	x	x	x	x
Transport, household collection	16.12	MJ/tkm	Ecoinvent	x	x	x	x	x	x	x	x	x	x	x
Consumption, diesel	0.022	kg/tkm	Ecoinvent est.	x	x	x	x	x	x	x	x	x	x	x
Transport, manure to farm	1	km	(Sindhöj & Rodhe, 2013b)	x	x									
Transport, manure to AD	8	km	(Vega, et al., 2014)			x	x	x	x	x	x	x	x	x
Transport, household	60	km	Assumption	x	x	x	x	x	x	x	x	x	x	x
Transport, sewage sludge	60	km	Assumption	x	x	x	x	x	x	x	x	x	x	x
Transport, slaughterhouse	60	km	(Horsens Kommune, 2012)	x	x	x	x	x	x	x	x	x	x	x
Transport, commercial	60	km	Assumption	x	x	x	x	x	x	x	x	x	x	x
Transport, cooking oil	60	km	Assumption	x	x	x	x	x	x	x	x	x	x	x
Manure management														
Direct spreading, manure	–	–	SimaPro model alternative	x	x									
Energy(e), pumping and stirring	4.6	kWh	(Wesnæs, et al., 2009)	x	x	x	x	x	x	x	x	x	x	x
Loss, CH_4 from manure storage	3.0	%	(Hamelin, et al., 2014)	x	x	x	x	x	x	x	x	x	x	x
Loss, N runoff	34.2	%	(Børgesen, et al., 2013) est.	x	x	x	x	x	x	x	x	x	x	x
Loss, P runoff	4.95	%	(Hamelin, et al., 2014) est.	x	x	x	x	x	x	x	x	x	x	x
Incineration processes														
Optimal water content in INC	60	%	(Christensen, 2011)	x	x									
Temperature sorbed by water	120	°C	(Chang & Huang, 2001)	x	x									
Evaporation energy, water	2260	MJ/t	(Christensen, 2011)	x	x									
Energy(e) use	70	kWh/t	(McDougall, et al., 2008)	x	x									
CHP, heat efficiency	72	%	(Meyer, 2014)	x	x									
CHP, power efficiency	28	%	(Meyer, 2014)	x	x									
Share to fly ash, DM	10	%	(Christensen, 2011)	x	x									
Share to bottom ash, DM	90	%	(Christensen, 2011)	x	x									
Share to fly ash, heavy metal	30	%	(Christensen, 2011)	x	x									
Share to bottom ash, heavy metal	70	%	(Christensen, 2011)	x	x									
Transportation, bottom ash	15	km	Assumption	x	x									
Transportation, fly ash (ship)	2500	km	Assumption	x	x									
AD processes														
Energy(e), sorting and crushing	11.6	kWh/t output	(NIRAS, 2004)			x	x	x				x		x
Sorting, inorganics in household	4	%	(Hamelin, et al., 2014)			x	x	x				x		x
Sorting, inorganics in commercial	4	%	Assumption											x
Sorting, loss of organics	0	%	(Hansen, et al., 2007)			x	x	x	x	x	x	x	x	x
Optimal water content in AD	90	%	(Hamelin, et al., 2014)			x	x	x	x	x	x	x	x	x
Specific heat, water	4.20	MJ/(t·°C)	(Hamelin, et al., 2014)			x	x	x	x	x	x	x	x	x
Temperature, input substrate	8	°C	(Hamelin, et al., 2014)			x	x	x	x	x	x	x	x	x
Before pasteur. (heat-exchanged)	39.7	°C	(Lemvig Biogas, 2014)			x	x	x	x	x	x	x	x	x
Pasteurization (5% loss assumed)	75	°C	(Hamelin, et al., 2014)			x	x	x	x	x	x	x	x	x
Loss, during heating up	5	%	(Hamelin, et al., 2014)			x	x	x	x	x	x	x	x	x
Evaporation energy, water	2260	MJ/t	(Christensen, 2011)			x	x	x	x	x	x	x	x	x
Energy(e) use, pasteurizing waste	19.66	kWh/t	(Lemvig Biogas, 2014), est.[17]			x	x	x	x	x	x	x	x	x

[17] Heat and power consumed by Lemvig Biogas plant (in 2012) per t organic waste is calculated based on the following informations given in Lemvig Biogas (2014):
- 0.01 MWh/L fuel oil
- 68,217 L fuel oil equivalent to 1% produced energy
- 235,975 t treated waste
- 2.2% power consumption of total energy output
- 6.8% heat consumption of total energy output

Energy(e) use, AD of waste	6.36	kWh/t	(Lemvig Biogas, 2014), est.			x	x	x	x	x	x	x	x	x
Co-digestion benefit	0	%	Assumption			x	x	x	x	x	x	x	x	x
Loss, CH_4 from AD	0	%	(Møller, et al., 2009)			x	x	x	x	x	x	x	x	x
Loss, CH_4 from torch	0.7	%	(Lemvig Biogas, 2014)			x	x	x	x	x	x	x	x	x
Loss, P from AD	10	%	(Möller & Müller, 2012)			x	x	x	x	x	x	x	x	x
Biogas utilization														
Cleaning, biogas	0	kWh	(Bauer, et al., 2013)			x	x	x				x		
Loss, CH_4 from biogas cleaning	2	%	(Rehl & Müller, 2013)			x	x	x				x		
Loss, CO_2 from biogas cleaning	2	%	Assuming same WS loss			x	x	x				x		
CHP, heat efficiency	57	%	(Kristensen, 2015)			x	x	x				x		
CHP, power efficiency	43	%	(Kristensen, 2015)			x	x	x				x		
LHV, CH_4	35.88	MJ/Nm^3	(Lemvig Biogas, 2014)			x	x	x				x		
Density, CH_4	0.668	kg/Nm^3	(TETB, 2014)			x	x	x				x		
Density, CO_2	1.842	kg/Nm^3	(TETB, 2014)			x	x	x				x		
Energy use, CHP (of input power)	2.2	%	(Lemvig Biogas, 2014)			x	x	x				x		
Biomethane utilization														
Upgrading (WS)	–	–	SimaPro model alternative						x		x			
Upgrading (WS), CH_4 loss	2	%	(Bauer, et al., 2013)						x		x			
Upgrading (WS), CO_2 loss	98	%	(Bauer, et al., 2013)						x		x			
Upgrading (WS), energy use	0.27	kWh/m^3 BG_{in}	(Bauer, et al., 2013)						x		x			
Upgrading (AS)	–	–	SimaPro model alternative							x				x
Upgrading (AS), CH_4 loss	0.1	%	(Bauer, et al., 2013)							x				x
Upgrading (AS), CO_2 loss	99.8	%	(Bauer, et al., 2013)							x				x
Upgrading (AS), energy use	0.68	kWh/m^3 BG_{in}	(Bauer, et al., 2013)							x				x
Upgrading (CS)	–	–	SimaPro model alternative										x	
Upgrading (CS), CH_4 loss	5	%	(Bauer, et al., 2013)										x	
Upgrading (CS), CO_2 loss	100	%	(Bauer, et al., 2013)										x	
Upgrading (CS), energy use	0.22	kWh/m^3 BG_{in}	(Bauer, et al., 2013)										x	
Upgrading (CS), CO_2 capture	20	%	(Acrion, 2011)										x	
For natural gas grid														
Compression, 45-55 bar	0.16	kWh/m^3 BM	(Bauer, et al., 2013)						x					
For bus fuel (CBG)														
Compression, 200 bar	0.21	kWh/m^3 BM	(Bauer, et al., 2013)											x
Compression, 300 bar	0.25	kWh/m^3 BM	(Bauer, et al., 2013)							x	x			
Energy use, bus	15.18	MJ/vkm	(Hung & Solli, 2012)							x	x			x
Transport, to filling station	100	km	(Stenkjær, 2012)							x	x			x
For bus fuel (LBG)														
LBG per BM	1.7	L/Nm^3	(Bauer, et al., 2013)										x	
Loss, CH_4 (BM to LBG conv.)	1.8	%	(Bauer, et al., 2013)										x	
Transport, to filling station	100	km	(Stenkjær, 2012)										x	
Tanking, LBG	0.16	kWh/m^3	(Hung & Solli, 2012)										x	
Energy use, bus	15.18	MJ/vkm	(Hung & Solli, 2012)										x	
Bioresidual management														
Reduction, CH_4 (tight storage cover)	68	%	(Anderson-Glenna & Morken, 2013)			x	x	x	x	x	x	x	x	x
Reduction, storage NH_3 (acid add	65	%	(Wesnæs, et al., 2009)						x	x	x	x	x	x
Utilizing raw wet bioresidual	–	–	SimaPro model alternative			x	x	x						
Utilizing separated fractions	–	–	SimaPro model alternative						x			x		
Utilizing solid fraction only	–	–	SimaPro model alternative							x				
Utilizing composted bioresidual	–	–	SimaPro model alternative								x		x	x
Loss, DM dewatered to liquid fr.	30.6	%	(Wesnæs, et al., 2013)						x	x	x	x	x	x
N in dewatered bioresidual solid	33.5	%	(Wesnæs, et al., 2013)						x	x	x	x	x	x
P in dewatered bioresidual solid	82.2	%	(Wesnæs, et al., 2013)						x	x	x	x	x	x
Energy, dewatering	2.3	kWh/m^3	(Hamelin, et al., 2010)						x	x	x	x	x	x
Energy, cleaned water to WWTP	0.4289	kWh/m^3	Ecoinvent							x				
Energy, waste water to WWTP	0.3997	kWh/m^3	Ecoinvent							x				
Dewatered bioresidual, DM	30	%	(Christensen, 2011)						x	x		x		
Composted bioresidual, DM	30	%	(Christensen, 2011)								x		x	x
Transport, bioresidual	8	km	(Vega, et al., 2014)			x	x	x	x	x		x		
Transport, compost	16	km	Assumption								x		x	x

= 6.36 kWh power / t waste AND 19.66 kWh heat / t waste = 70.77 MJ heat / t waste

Spreading, diesel, >25% H_2O	0.8	kWh/t	Ecoinvent			x	x	x	x	x	x	x	x	x
Spreading, diesel, <75% H_2O	0.16	kWh/t	Ecoinvent			x	x	x	x	x	x	x	x	x
Reduction, NH_3 by soil injection	90	%	(Sindhöj & Rodhe, 2013a)			x	x	x	x	x	x	x	x	x
Mineral fertilizer substitution														
Plant availability (P), bioresidual	100	%	(Fødevareministeriet, 2014)			x	x	x	x	x		x		
Plant availability (N), bioresidual	70	%	(Lukehurst, et al., 2010)			x	x	x	x	x		x		
Plant availability (P), manure	100	%	(Fødevareministeriet, 2014)	x	x									
Plant availability (N), manure	65	%	(Fødevareministeriet, 2014)	x	x									
Plant availability (P), compost	0	%	(Fødevareministeriet, 2014)								x		x	x
Plant availability (N), compost	0	%	(Fødevareministeriet, 2014)								x		x	x
Energy substitution														
Marginal, power (coal)	100	%	SimaPro model alternative				x							
Marginal, heat (natural gas)	100	%	SimaPro model alternative				x							
Marginal, gas grid (natural gas)	100	%	SimaPro model alternative						x					
Marginal, transport (diesel)	100	%	SimaPro model alternative							x	x		x	x
Average mix, heat	100	%	SimaPro model alternative	x	x	x						x		
Average mix, power	100	%	SimaPro model alternative	x	x	x						x		
2020 mix, power	100	%	SimaPro model alternative					x						
2020 mix, heat	100	%	SimaPro model alternative					x						
Energy input source to system				**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**
Average mix, power	100	%												
Average mix, heat (biomass)	100	%												
Substrates (different mixtures)				**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**
DM, pig	7.48	%	See Table 31											
DM, cattle	12.57	%	See Table 31											
DM, household	31.5	%	See Table 31											
DM, sewage sludge	25	%	See Table 31											
DM, slaughterhouse	15.2	%	See Table 31											
DM, commercial	24.4	%	See Table 31											
DM, cooking oil	90	%	See Table 31											
VS/DM, pig	81	%	See Table 31											
VS/DM, cattle	83	%	See Table 31											
VS/DM, household	82	%	See Table 31											
VS/DM, sewage sludge	90	%	See Table 31											
VS/DM, slaughterhouse	92.5	%	See Table 31											
VS/DM, commercial	94	%	See Table 31											
VS/DM, cooking oil	100	%	See Table 31											
CH_4 share, pig	65	%	See Table 31											
CH_4 share, cattle	65	%	See Table 31											
CH_4 share, household	65	%	See Table 31											
CH_4 share, sewage sludge	65	%	See Table 31											
CH_4 share, slaughterhouse	63	%	See Table 31											
CH_4 share, commercial	63	%	See Table 31											
CH_4 share, cooking oil	68	%	See Table 31											
CH_4 yield, pig	260	m^3/t VS	See Table 31											
CH_4 yield, cattle	210	m^3/t VS	See Table 31											
CH_4 yield, household	330	m^3/t VS	See Table 31											
CH_4 yield, sewage sludge	250	m^3/t VS	See Table 31											
CH_4 yield, slaughterhouse	375	m^3/t VS	See Table 31											
CH_4 yield, commercial	277	m^3/t VS	See Table 31											
CH_4 yield, cooking oil	757	m^3/t VS	See Table 31											
Undegraded DM, pig	51.4	%	See Table 31											
Undegraded DM, cattle	69.3	%	See Table 31											
Undegraded DM, household	47.5	%	See Table 31											
Undegraded DM, sewage sludge	82.9	%	See Table 31											
Undegraded DM, slaughterhouse	14.0	%	See Table 31											
Undegraded DM, commercial	46.4	%	See Table 31											
Undegraded DM, cooking oil	0.0	%	See Table 31											
N, pig	80.2	kg/t DM	See Table 31											
N, cattle	55.7	kg/t DM	See Table 31											
N, household	27.9	kg/t DM	See Table 31											
N, sewage sludge	40.0	kg/t DM	See Table 31											
N, slaughterhouse	33.9	kg/t DM	See Table 31											

N, commercial	33.0	kg/t DM	See Table 31											
N, cooking oil	0	kg/t DM	See Table 31											
P, pig	16.2	kg/t DM	See Table 31											
P, cattle	8.1	kg/t DM	See Table 31											
P, household	4.1	kg/t DM	See Table 31											
P, sewage sludge	26.0	kg/t DM	See Table 31											
P, slaughterhouse	7.2	kg/t DM	See Table 31											
P, commercial	4.6	kg/t DM	See Table 31											
P, cooking oil	0.02	kg/t DM	See Table 31											
HHV, pig	10833	MJ/t DM	See Table 31											
HHV, cattle	6563	MJ/t DM	See Table 31											
HHV, household	9651	MJ/t DM	See Table 31											
HHV, sewage sludge	23400	MJ/t DM	See Table 31											
HHV, slaughterhouse	11118	MJ/t DM	See Table 31											
HHV, commercial	14748	MJ/t DM	See Table 31											
HHV, cooking oil	42556	MJ/t DM	See Table 31											
Heavy metal, sludge (Cu)	183	g/t DM	See Table 32											
Heavy metal, sludge (Zn)	620	g/t DM	See Table 32											
Heavy metal, sludge (Pb)	32.7	g/t DM	See Table 32											
Heavy metal, sludge (Cd)	0.972	g/t DM	See Table 32											
Heavy metal, sludge (Hg)	0.587	g/t DM	See Table 32											
Heavy metal, sludge (Ni)	21.4	g/t DM	See Table 32											
Heavy metal, sludge (Cr)	–	g/t DM	See Table 32											

Table 48. LCI data table with baseline input values of processes and parameters used in the model, marked for each scenario variant. Climate and energy allocation aspects are not included because they apply for the same scenario variant setups

POLAND difference				REF		ALT								
				S 0		S 1			S 2			S 3		
Process	Value	Unit	Reference	a	b	a	b	c	a	b	c	a	b	c
Substrates				x	x	x	x	x	x	x	x	x	x	x
DM, pig	7.2	%	See Table 36											
DM, cattle	8.5	%	See Table 36											
DM, household	27	%	See Table 36											
VS/DM, pig	80	%	See Table 36											
VS/DM, cattle	80	%	See Table 36											
VS/DM, household	87	%	See Table 36											
VS/DM, sewage sludge	62	%	See Table 36											
Undegraded DM, pig	39.2	%	See Table 36											
Undegraded DM, cattle	70.4	%	See Table 36											
Undegraded DM, household	44.3	%	See Table 36											
Undegraded DM, sewage sludge	88.2	%	See Table 36											
N, pig	83.1	kg/t DM	See Table 36											
N, cattle	23.5	kg/t DM	See Table 36											
N, household	32.6	kg/t DM	See Table 36											
N, sewage sludge	16.2	kg/t DM	See Table 36											
P, pig	21.6	kg/t DM	See Table 36											
P, cattle	11.8	kg/t DM	See Table 36											
P, household	4.8	kg/t DM	See Table 36											
P, sewage sludge	0.1	kg/t DM	See Table 36											
Heavy metal, sludge (Cu)	161	g/t DM	(Oleszczuk, 2006)											
Heavy metal, sludge (Zn)	1680	g/t DM	(Oleszczuk, 2006)											
Heavy metal, sludge (Pb)	20.1	g/t DM	(Oleszczuk, 2006)											
Heavy metal, sludge (Cd)	2.36	g/t DM	(Oleszczuk, 2006)											
Heavy metal, sludge (Hg)	–	g/t DM	(Oleszczuk, 2006)											
Heavy metal, sludge (Ni)	15.7	g/t DM	(Oleszczuk, 2006)											
Heavy metal, sludge (Cr)	56.9	g/t DM	(Oleszczuk, 2006)											

Appendix C: Raw data results

Full result tables for selected impact categories for all 11 scenario variants (DK and PL) and sensitivities (16) in the following order:

Impact category	Unit
Climate change	kg CO_2 eq
Terrestrial acidification	kg SO_2 eq
Marine eutrophication	kg N eq
Human toxicity	kg 1,4-DB eq
Fossil depletion	kg oil eq

Denmark scenarios

Table 49. Raw data for the REF (upper) and ALT (lower) scenarios for DK

REF DK									
S0a									
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	INC	Ash proc.	Transp.ash	Heat subs.	El.subs	Biofert.
25.8	840.8	5.9	10.9	53.3	30.1	9.8	-32.9	-60.8	-831.4
0.7	3.9	0.0	0.0	0.4	0.1	0.0	-0.1	-0.2	-3.6
0.0	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	-0.3
-10.3	0.7	0.0	0.1	2.2	4.3	0.1	-0.2	-1.4	-16.1
-73.2	7.4	2.0	3.5	11.8	9.2	3.2	-11.2	-15.4	-83.8
S0b									
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	INC	Ash proc.	Transp.ash	Heat subs.	El.subs	Biofert.
-855.5	119.3	7.5	28.4	345.4	7.5	3.4	-623.3	-617.6	-126.1
-1.9	0.6	0.0	0.1	3.1	0.0	0.0	-3.6	-1.6	-0.6
2.3	0.0	0.0	0.0	2.5	0.0	0.0	-0.1	-0.1	0.0
-25.6	0.1	-0.1	0.2	18.9	1.1	0.0	-29.5	-13.8	-2.4
-222.5	1.1	2.4	9.1	59.4	2.3	1.1	-128.8	-156.4	-12.6

ALT DK												
S1a												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El.subs	Heat subs.	Fuel subs.	Biofert.
-765.2	65.4	6.7	47.5	35.7	77.2	189.5	6.2	x	-87.1	-32.4	x	-1074.0
0.0	0.1	0.0	0.2	0.3	0.1	4.6	0.0	x	-0.2	-0.5	x	-4.6
50.0	0.0	0.0	0.0	0.0	0.0	32.0	0.0	18.3	0.0	0.0	x	-0.3
263.8	0.5	0.1	0.3	6.3	0.6	290.2	0.0	x	-1.9	-11.4	x	-20.9
-95.7	5.8	1.7	14.9	9.5	7.4	2.5	2.0	x	-22.1	-9.0	x	-108.5
S1b												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El.subs	Heat subs.	Fuel subs.	Biofert.
-1355.5	65.4	6.7	47.5	35.7	77.2	189.5	6.2	x	-622.3	-87.5	x	-1074.0
-0.8	0.1	0.0	0.2	0.3	0.1	4.6	0.0	x	-1.4	-0.1	x	-4.6
50.0	0.0	0.0	0.0	0.0	0.0	32.0	0.0	18.3	0.0	0.0	x	-0.3
273.0	0.5	0.1	0.3	6.3	0.6	290.2	0.0	x	-3.6	-0.6	x	-20.9
-228.1	5.8	1.7	14.9	9.5	7.4	2.5	2.0	x	-133.9	-29.7	x	-108.5
S1c												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El.subs	Heat subs.	Fuel subs.	Biofert.
-1083.5	65.4	6.7	47.5	35.7	77.2	189.5	6.2	x	-274.7	-163.2	x	-1074.0
-0.9	0.1	0.0	0.2	0.3	0.1	4.6	0.0	x	-0.7	-0.9	x	-4.6
50.0	0.0	0.0	0.0	0.0	0.0	32.0	0.0	18.3	0.0	0.0	x	-0.3
259.6	0.5	0.1	0.3	6.3	0.6	290.2	0.0	x	-9.8	-7.7	x	-20.9
-165.7	5.8	1.7	14.9	9.5	7.4	2.5	2.0	x	-67.4	-33.7	x	-108.5
S2a												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	BG upgr.	Posttreat.biores.	Transp.biores.	Nutr. leach.	Nat.gas subs.	Fuel subs.	Biofert.	
-645.0	55.5	8.3	16.7	35.8	65.0	281.8	6.2	x	-253.9	x	-860.4	
17.7	0.1	0.0	0.0	0.3	0.1	21.1	0.0	x	-0.3	x	-3.7	
37.1	0.0	0.0	0.0	0.0	0.0	22.8	0.0	14.5	0.0	x	-0.3	
194.9	0.4	0.2	0.1	6.4	1.0	203.0	0.0	x	0.4	x	-16.6	
-139.4	5.1	2.1	5.5	9.5	10.9	26.1	2.0	x	-113.9	x	-86.6	
S2b												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	BG upgr.	Transp.biofuel	Posttreat.biores.	Transp.biores.	Nutr. leach.	Fuel subs.	Biofert.	
-230.7	55.5	8.3	16.7	35.8	52.1	126.9	125.1	1.1	x	-340.3	-312.0	

17.1	0.1	0.0	0.0	0.3	0.2	0.3	18.3	0.0	x	-0.7	-1.5	
13.3	0.0	0.0	0.0	0.0	0.0	0.0	8.8	0.0	4.6	0.0	-0.1	
208.8	0.4	0.2	0.1	6.4	2.8	0.9	207.4	0.0	x	-3.7	-5.7	
-39.2	5.1	2.1	5.5	9.5	12.8	42.2	20.7	0.4	x	-107.2	-30.2	
S2c												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	BG upgr.	Transp.biofuel	Posttreat.biores.	Transp.biores.	Nutr. leach.	Fuel subs.		
219.7	55.5	8.3	16.7	35.8	52.1	126.9	96.5	7.5	x	-340.3		
12.0	0.1	0.0	0.0	0.3	0.2	0.3	11.6	0.0	x	-0.7		
23.5	0.0	0.0	0.0	0.0	0.0	0.0	14.6	0.0	9.3	0.0		
210.2	0.4	0.2	0.1	6.4	2.8	0.9	199.3	0.1	x	-3.7		
51.3	5.1	2.1	5.5	9.5	12.8	42.2	26.7	2.5	x	-107.2		
S3a												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El.subs	Heat subs.	Fuel subs.	Biofert.
-520.0	65.4	6.7	47.5	35.7	77.2	311.0	6.2	x	-87.1	-32.4	x	-950.2
19.1	0.1	0.0	0.2	0.3	0.1	23.2	0.0	x	-0.2	-0.5	x	-4.1
40.4	0.0	0.0	0.0	0.0	0.0	24.8	0.0	15.9	0.0	0.0	x	-0.3
177.5	0.5	0.1	0.3	6.3	0.6	201.4	0.0	x	-1.9	-11.4	x	-18.3
-59.0	5.8	1.7	14.9	9.5	7.4	26.1	2.0	x	-22.1	-9.0	x	-95.4
S3b												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	Transp.biofuel	LBG Produc-tion	Posttreat.biores.	Transp.biores.	Nutr. leach.	Fuel subs.	Biofert.	Indust. CO_2
116.0	55.5	8.3	16.7	35.8	0.8	234.0	95.9	7.5	x	-317.7	x	-20.7
11.5	0.1	0.0	0.0	0.3	0.0	0.2	11.5	0.0	x	-0.6	x	0.0
23.5	0.0	0.0	0.0	0.0	0.0	0.0	14.3	0.0	9.2	0.0	x	0.0
208.5	0.4	0.2	0.1	6.4	0.0	1.9	203.1	0.1	x	-3.4	x	-0.2
-30.5	5.1	2.1	5.5	9.5	0.3	22.1	26.7	2.5	x	-100.1	x	-4.2
S3c												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	BG upgr.	Transp.biofuel	Posttreat.biores.	Transp.biores.	Nutr. leach.	Fuel subs.	Biofert.	
-46.5	8.7	6.9	29.3	36.4	74.4	196.1	121.3	6.3	x	-526.0	x	
3.3	0.0	0.0	0.1	0.3	0.3	0.5	3.1	0.0	x	-1.0	x	
8.3	0.0	0.0	0.0	0.0	0.0	0.0	5.0	0.0	3.3	-0.1	x	
178.4	0.1	0.1	0.2	6.5	4.0	1.4	171.8	0.0	x	-5.7	x	
-32.3	0.8	1.8	9.4	9.7	18.2	65.1	26.4	2.1	x	-165.7	x	

REF DK									
S0a									
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	INC	Ash proc.	Transp.ash	Heat subs.	El.subs	Biofert.
25.8	840.8	5.9	10.9	53.3	30.1	9.8	-32.9	-60.8	-831.4
0.7	3.9	0.0	0.0	0.4	0.1	0.0	-0.1	-0.2	-3.6
0.0	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	-0.3
-10.3	0.7	0.0	0.1	2.2	4.3	0.1	-0.2	-1.4	-16.1
-73.2	7.4	2.0	3.5	11.8	9.2	3.2	-11.2	-15.4	-83.8
S0b									
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	INC	Ash proc.	Transp.ash	Heat subs.	El.subs	Biofert.
-855.5	119.3	7.5	28.4	345.4	7.5	3.4	-623.3	-617.6	-126.1
-1.9	0.6	0.0	0.1	3.1	0.0	0.0	-3.6	-1.6	-0.6
2.3	0.0	0.0	0.0	2.5	0.0	0.0	-0.1	-0.1	0.0
-25.6	0.1	-0.1	0.2	18.9	1.1	0.0	-29.5	-13.8	-2.4
-222.5	1.1	2.4	9.1	59.4	2.3	1.1	-128.8	-156.4	-12.6

Poland scenarios

Table 50. Raw data for the REF (upper) and ALT (lower) scenarios for PL

REF PL									
S0a									
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	INC	Ash proc.	Transp.ash	Heat subs.	El.subs	Biofert.
-168.5	652.2	5.9	10.9	79.1	31.9	10.3	-32.9	-132.4	-793.5
-8.1	-4.5	0.0	0.0	0.6	0.1	0.0	-0.1	-0.7	-3.7
-0.4	-0.2	0.0	0.0	0.1	0.0	0.0	0.0	0.0	-0.3
-9.0	1.2	0.0	0.1	2.6	4.5	0.1	-0.2	-2.4	-14.9
-71.1	15.8	2.0	3.5	18.9	9.7	3.4	-11.2	-35.2	-78.1
S0b									
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	INC	Ash proc.	Transp.ash	Heat subs.	El.subs	Biofert.
-1728.6	125.3	8.0	30.3	438.0	15.7	5.7	-868.7	-1345.1	-137.7

-4.3	4.3	0.0	0.1	3.8	0.1	0.0	-5.0	-7.0	-0.6
2.4	0.2	0.0	0.0	2.5	0.0	0.0	-0.1	-0.1	-0.1
-49.8	0.2	-0.1	0.2	20.2	2.2	0.0	-45.8	-24.2	-2.6
-458.4	2.2	2.6	9.7	84.9	4.8	1.9	-193.6	-357.3	-13.6

ALT PL												
S1a												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El.subs	Heat subs.	Fuel subs.	Biofert.
-1428.2	105.1	15.7	53.5	58.1	128.6	206.6	6.2	x	-795.3	-37.7	x	-1169.1
-3.6	0.3	0.1	0.2	0.5	0.4	4.8	0.0	x	-4.2	-0.6	x	-5.2
51.3	0.0	0.0	0.0	0.0	0.0	32.6	0.0	19.2	-0.1	0.0	x	-0.4
636.4	0.9	0.2	0.3	6.6	1.3	677.0	0.0	x	-14.3	-13.3	x	-22.3
-264.1	13.8	4.2	16.7	15.7	19.3	2.5	2.1	x	-211.2	-10.5	x	-116.7
S1b												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El.subs	Heat subs.	Fuel subs.	Biofert.
-1549.3	105.1	15.7	53.5	58.1	128.6	206.6	6.2	x	-852.2	-101.9	x	-1169.1
-3.5	0.3	0.1	0.2	0.5	0.4	4.8	0.0	x	-4.5	-0.2	x	-5.2
51.3	0.0	0.0	0.0	0.0	0.0	32.6	0.0	19.2	-0.1	0.0	x	-0.4
650.8	0.9	0.2	0.3	6.6	1.3	677.0	0.0	x	-12.5	-0.6	x	-22.3
-303.6	13.8	4.2	16.7	15.7	19.3	2.5	2.1	x	-226.7	-34.6	x	-116.7
S1c												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El.subs	Heat subs.	Fuel subs.	Biofert.
-1556.6	105.1	15.7	53.5	58.1	128.6	206.6	6.2	x	-696.6	-264.8	x	-1169.1
-4.0	0.3	0.1	0.2	0.5	0.4	4.8	0.0	x	-3.6	-1.5	x	-5.2
51.3	0.0	0.0	0.0	0.0	0.0	32.6	0.0	19.2	-0.1	0.0	x	-0.4
637.7	0.9	0.2	0.3	6.6	1.3	677.0	0.0	x	-12.3	-14.0	x	-22.3
-289.6	13.8	4.2	16.7	15.7	19.3	2.5	2.1	x	-188.3	-59.0	x	-116.7
S2a												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	BG upgr.	Posttreat.biores.	Transp.biores.	Nutr. leach.	Nat.gas subs.	Fuel subs.	Biofert.	
-443.7	91.9	18.0	17.5	58.1	126.9	280.3	6.2	x	-223.4	x	-819.1	
16.8	0.3	0.1	0.0	0.5	0.5	19.0	0.0	x	0.1	x	-3.7	

32.9	0.0	0.0	0.0	0.0	0.0	20.2	0.0	13.0	0.0	x	-0.3	
479.6	0.9	0.2	0.1	6.6	1.8	484.1	0.0	x	1.3	x	-15.4	
-92.4	13.1	4.8	5.7	15.7	27.2	29.8	2.1	x	-109.8	x	-81.0	
S2b												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	BG upgr.	Transp.biofuel	Posttreat.biores.	Transp.biores.	Nutr. leach.	Fuel subs.	Biofert.	
-76.5	91.9	18.0	17.5	58.1	115.3	139.4	127.9	1.1	x	-338.0	-307.6	
16.3	0.3	0.1	0.0	0.5	0.6	0.3	16.4	0.0	x	-0.5	-1.6	
11.8	0.0	0.0	0.0	0.0	0.0	0.0	7.9	0.0	4.1	0.0	-0.2	
493.1	0.9	0.2	0.1	6.6	3.8	1.0	489.4	0.0	x	-3.5	-5.4	
3.1	13.1	4.8	5.7	15.7	30.0	46.3	23.8	0.4	x	-107.9	-29.0	
S2c												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	BG upgr.	Transp.biofuel	Posttreat.biores.	Transp.biores.	Nutr. leach.	Fuel subs.	Biofert.	
275.1	91.9	18.0	17.5	58.1	170.5	142.2	104.7	7.3	x	-335.0	x	
11.9	0.3	0.1	0.0	0.5	0.8	0.4	10.3	0.0	x	-0.4	x	
20.4	0.0	0.0	0.0	0.0	0.0	0.0	12.4	0.0	8.0	0.0	x	
492.4	0.9	0.2	0.1	6.6	2.6	1.0	484.4	0.1	x	-3.5	x	
51.2	13.1	4.8	5.7	15.7	38.8	47.2	30.3	2.4	x	-106.9	x	
S3a												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El.subs	Heat subs.	Fuel subs.	Biofert.
-1162.2	105.1	15.7	53.5	58.1	128.6	343.9	6.2	x	-795.3	-37.7	x	-1040.4
16.3	0.3	0.1	0.2	0.5	0.4	24.3	0.0	x	-4.2	-0.6	x	-4.7
41.6	0.0	0.0	0.0	0.0	0.0	25.5	0.0	16.6	-0.1	0.0	x	-0.4
428.9	0.9	0.2	0.3	6.6	1.3	466.9	0.0	x	-14.3	-13.3	x	-19.7
-223.1	13.8	4.2	16.7	15.7	19.3	29.8	2.1	x	-211.2	-10.5	x	-103.0
S3b												
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	Transp.biofuel	LBG Production	Posttreat.biores.	Transp.biores.	Nutr. leach.	Fuel subs.	Biofert.	Indust. CO_2
111.4	65.1	8.1	17.5	35.7	0.8	256.9	91.6	7.3	x	-348.9	x	-22.7
10.2	0.1	0.0	0.0	0.3	0.0	0.2	10.3	0.0	x	-0.7	x	0.0
20.4	0.0	0.0	0.0	0.0	0.0	0.0	12.4	0.0	8.0	0.0	x	0.0
489.3	0.5	0.1	0.1	6.3	0.0	2.1	484.2	0.1	x	-3.8	x	-0.3
-37.8	5.8	2.1	5.7	9.5	0.3	24.3	26.6	2.4	x	-109.9	x	-4.6
S3c												
Total	Manure stor. &	Feedst. &	Transp.org.waste	Pretreat. &	BG upgr.	Transp.biofuel	Posttreat.biores.	Transp.biores.	Nutr. leach.	Fuel subs.	Biofert.	

	appl.	Proc.		AD								
119.4	14.0	15.0	30.4	58.9	148.3	194.6	124.1	6.4	x	-472.3	x	
6.8	0.0	0.1	0.1	0.5	0.8	0.5	5.4	0.0	x	-0.6	x	
8.4	0.0	0.0	0.0	0.0	0.0	0.0	5.0	0.0	3.3	-0.1	x	
435.7	0.1	0.3	0.2	6.8	5.0	1.4	426.7	0.0	x	-4.9	x	
16.1	1.8	4.0	9.7	15.9	38.6	64.7	30.1	2.1	x	-150.7	x	

Sensitivity analysis

Table 51. Raw data for the REF (upper) and ALT (lower) scenarios for PL

REF (S0a)									
Baseline									
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	INC	Ash proc.	Transp. ash	Heat subs.	El.subs	Biofert.
-168.5	652.2	5.9	10.9	79.1	31.9	10.3	-32.9	-132.4	-793.5
-8.1	-4.5	0.0	0.0	0.6	0.1	0.0	-0.1	-0.7	-3.7
-0.4	-0.2	0.0	0.0	0.1	0.0	0.0	0.0	0.0	-0.3
-9.0	1.2	0.0	0.1	2.6	4.5	0.1	-0.2	-2.4	-14.9
-71.1	15.8	2.0	3.5	18.9	9.7	3.4	-11.2	-35.2	-78.1
CHP eff.									
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	INC	Ash proc.	Transp. ash	Heat subs.	El.subs	Biofert.
-125.3	652.2	5.9	10.9	79.1	31.9	10.3	-29.9	-92.2	-793.5
-7.9	-4.5	0.0	0.0	0.6	0.1	0.0	0.0	-0.5	-3.7
-0.4	-0.2	0.0	0.0	0.1	0.0	0.0	0.0	0.0	-0.3
-8.3	1.2	0.0	0.1	2.6	4.5	0.1	-0.2	-1.7	-14.9
-59.4	15.8	2.0	3.5	18.9	9.7	3.4	-10.1	-24.5	-78.1
HHV sludge									
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	INC	Ash proc.	Transp. ash	Heat subs.	El.subs	Biofert.
-196.4	652.2	5.9	10.9	79.1	31.9	10.3	-38.4	-154.7	-793.5
-8.2	-4.5	0.0	0.0	0.6	0.1	0.0	-0.1	-0.8	-3.7
-0.4	-0.2	0.0	0.0	0.1	0.0	0.0	0.0	0.0	-0.3
-9.4	1.2	0.0	0.1	2.6	4.5	0.1	-0.2	-2.8	-14.9
-78.9	15.8	2.0	3.5	18.9	9.7	3.4	-13.0	-41.1	-78.1
Power use									

Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	INC	Ash proc.	Transp. ash	Heat subs.	El.subs.	Biofert.
-211.4	652.2	5.9	10.9	36.2	31.9	10.3	-32.9	-132.4	-793.5
-8.3	-4.5	0.0	0.0	0.4	0.1	0.0	-0.1	-0.7	-3.7
-0.4	-0.2	0.0	0.0	0.1	0.0	0.0	0.0	0.0	-0.3
-9.8	1.2	0.0	0.1	1.8	4.5	0.1	-0.2	-2.4	-14.9
-82.5	15.8	2.0	3.5	7.6	9.7	3.4	-11.2	-35.2	-78.1
Transport dist									
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	INC	Ash proc.	Transp. ash	Heat subs.	El.subs.	Biofert.
-177.4	652.2	5.9	10.9	79.1	31.9	1.4	-32.9	-132.4	-793.5
-8.1	-4.5	0.0	0.0	0.6	0.1	0.0	-0.1	-0.7	-3.7
-0.4	-0.2	0.0	0.0	0.1	0.0	0.0	0.0	0.0	-0.3
-9.1	1.2	0.0	0.1	2.6	4.5	0.0	-0.2	-2.4	-14.9
-74.0	15.8	2.0	3.5	18.9	9.7	0.5	-11.2	-35.2	-78.1
N availability									
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	INC	Ash proc.	Transp. ash	Heat subs.	El.subs.	Biofert.
23.3	652.2	5.9	10.9	79.1	31.9	10.3	-32.9	-132.4	-601.7
-7.4	-4.5	0.0	0.0	0.6	0.1	0.0	-0.1	-0.7	-3.0
-0.3	-0.2	0.0	0.0	0.1	0.0	0.0	0.0	0.0	-0.3
-5.0	1.2	0.0	0.1	2.6	4.5	0.1	-0.2	-2.4	-10.9
-50.8	15.8	2.0	3.5	18.9	9.7	3.4	-11.2	-35.2	-57.8

ALT (S3a)											
Baseline											
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El. subs.	Heat subs.	Biofert.
-1162.2	105.1	15.7	53.5	58.1	128.6	343.9	6.2	x	-795.3	-37.7	-1040.4
16.3	0.3	0.1	0.2	0.5	0.4	24.3	0.0	x	-4.2	-0.6	-4.7
41.6	0.0	0.0	0.0	0.0	0.0	25.5	0.0	16.6	-0.1	0.0	-0.4
428.9	0.9	0.2	0.3	6.6	1.3	466.9	0.0	x	-14.3	-13.3	-19.7
-223.1	13.8	4.2	16.7	15.7	19.3	29.8	2.1	x	-211.2	-10.5	-103.0
Substrate DM											
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El. subs.	Heat subs.	Biofert.
-1201.0	64.2	16.6	48.1	58.9	128.6	343.9	6.2	x	-795.3	-37.7	-1034.6
17.3	0.1	0.1	0.2	0.5	0.4	25.5	0.0	x	-4.2	-0.6	-4.7
41.4	0.0	0.0	0.0	0.0	0.0	25.3	0.0	16.5	-0.1	0.0	-0.4
428.5	0.2	0.4	0.2	6.8	1.3	466.6	0.0	x	-14.3	-13.3	-19.5

-234.8	3.0	4.4	14.9	15.9	19.3	29.8	2.1	x	-211.2	-10.5	-102.4
Specific heat											
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El. subs.	Heat subs.	Biofert.
-1156.8	105.1	15.7	53.5	57.7	128.6	343.9	6.2	x	-795.3	-37.7	-1034.6
17.5	0.3	0.1	0.2	0.5	0.4	25.5	0.0	x	-4.2	-0.6	-4.7
41.3	0.0	0.0	0.0	0.0	0.0	25.3	0.0	16.5	-0.1	0.0	-0.4
428.6	0.9	0.2	0.3	6.4	1.3	466.6	0.0	x	-14.3	-13.3	-19.5
-222.6	13.8	4.2	16.7	15.6	19.3	29.8	2.1	x	-211.2	-10.5	-102.4
Temperature											
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El. subs.	Heat subs.	Biofert.
-1115.8	122.8	15.7	53.5	58.1	128.3	361.5	6.2	x	-789.9	-37.4	-1034.6
17.6	0.3	0.1	0.2	0.5	0.4	25.5	0.0	x	-4.1	-0.6	-4.7
41.3	0.0	0.0	0.0	0.0	0.0	25.3	0.0	16.5	-0.1	0.0	-0.4
428.0	0.9	0.2	0.3	6.6	1.3	465.6	0.0	x	-14.2	-13.2	-19.5
-221.0	13.8	4.2	16.7	15.7	19.3	29.9	2.1	x	-209.8	-10.4	-102.4
CHP efficiency											
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El. subs.	Heat subs.	Biofert.
-1093.6	105.1	15.7	53.5	58.1	128.6	343.9	6.2	x	-739.8	-30.4	-1034.6
18.0	0.3	0.1	0.2	0.5	0.4	25.5	0.0	x	-3.9	-0.5	-4.7
41.3	0.0	0.0	0.0	0.0	0.0	25.3	0.0	16.5	-0.1	0.0	-0.4
432.3	0.9	0.2	0.3	6.6	1.3	466.6	0.0	x	-13.3	-10.7	-19.5
-205.7	13.8	4.2	16.7	15.7	19.3	29.8	2.1	x	-196.5	-8.5	-102.4
CH_4 loss from AD											
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El. subs.	Heat subs.	Biofert.
-826.7	105.1	15.7	53.5	317.4	115.8	343.9	6.2	x	-715.7	-33.9	-1034.6
18.0	0.3	0.1	0.2	0.5	0.4	25.5	0.0	x	-3.7	-0.5	-4.7
41.3	0.0	0.0	0.0	0.0	0.0	25.3	0.0	16.5	-0.1	0.0	-0.4
431.4	0.9	0.2	0.3	6.6	1.2	466.6	0.0	x	-12.9	-12.0	-19.5
-202.2	13.8	4.2	16.7	15.7	17.4	29.8	2.1	x	-190.1	-9.5	-102.4
P loss from AD											
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El. subs.	Heat subs.	Biofert.
-1246.3	105.1	15.7	53.5	58.1	128.6	363.9	6.2	x	-795.3	-37.7	-1144.5
18.2	0.3	0.1	0.2	0.5	0.4	26.7	0.0	x	-4.2	-0.6	-5.2
45.7	0.0	0.0	0.0	0.0	0.0	28.0	0.0	18.3	-0.1	0.0	-0.4
426.1	0.9	0.2	0.3	6.6	1.3	466.1	0.0	x	-14.3	-13.3	-21.6

-233.4	13.8	4.2	16.7	15.7	19.3	29.8	2.1	x	-211.2	-10.5	-113.3
Synergy effect											
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El. subs.	Heat subs.	Biofert.
-1229.5	105.1	15.7	53.5	58.1	138.6	343.9	6.2	x	-874.6	-41.5	-1034.6
17.1	0.3	0.1	0.2	0.5	0.5	25.5	0.0	x	-4.6	-0.6	-4.7
41.2	0.0	0.0	0.0	0.0	0.0	25.2	0.0	16.5	-0.1	0.0	-0.4
418.2	0.9	0.2	0.3	6.6	1.4	458.7	0.0	x	-15.7	-14.6	-19.5
-243.4	13.8	4.2	16.7	15.7	20.5	29.9	2.1	x	-232.3	-11.6	-102.4
NH_3 **spread.loss**											
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El. subs.	Heat subs.	Biofert.
-1158.3	105.1	15.7	53.5	58.1	128.6	343.9	6.2	x	-795.3	-37.7	-1036.5
17.1	0.3	0.1	0.2	0.5	0.4	25.1	0.0	x	-4.2	-0.6	-4.7
41.4	0.0	0.0	0.0	0.0	0.0	25.4	0.0	16.5	-0.1	0.0	-0.4
428.8	0.9	0.2	0.3	6.6	1.3	466.7	0.0	x	-14.3	-13.3	-19.6
-222.7	13.8	4.2	16.7	15.7	19.3	29.8	2.1	x	-211.2	-10.5	-102.6
N content											
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El. subs.	Heat subs.	Biofert.
-1225.7	105.1	15.7	53.5	58.1	128.6	363.9	6.2	x	-795.3	-37.7	-1123.9
18.4	0.3	0.1	0.2	0.5	0.4	26.7	0.0	x	-4.2	-0.6	-5.0
45.8	0.0	0.0	0.0	0.0	0.0	28.0	0.0	18.3	-0.1	0.0	-0.4
426.4	0.9	0.2	0.3	6.6	1.3	466.1	0.0	x	-14.3	-13.3	-21.4
-231.9	13.8	4.2	16.7	15.7	19.3	29.8	2.1	x	-211.2	-10.5	-111.9
Sep.eff.											
Total	Manure stor. & appl.	Feedst. & Proc.	Transp.org.waste	Pretreat. & AD	CHP	Posttreat.biores.	Transp.biores.	Nutr. leach.	El. subs.	Heat subs.	Biofert.
-1164.2	105.1	15.7	53.5	58.1	128.6	341.9	6.2	x	-795.3	-37.7	-1040.4
16.3	0.3	0.1	0.2	0.5	0.4	24.3	0.0	x	-4.2	-0.6	-4.7
38.3	0.0	0.0	0.0	0.0	0.0	22.2	0.0	16.6	-0.1	0.0	-0.4
233.5	0.9	0.2	0.3	6.6	1.3	271.4	0.0	x	-14.3	-13.3	-19.7
-223.4	13.8	4.2	16.7	15.7	19.3	29.6	2.1	x	-211.2	-10.5	-103.0

Appendix D: MFA (Sensitivity results)

MFA charts for GHG flows generated by Simapro for baseline scenarios for sensitivity analysis.

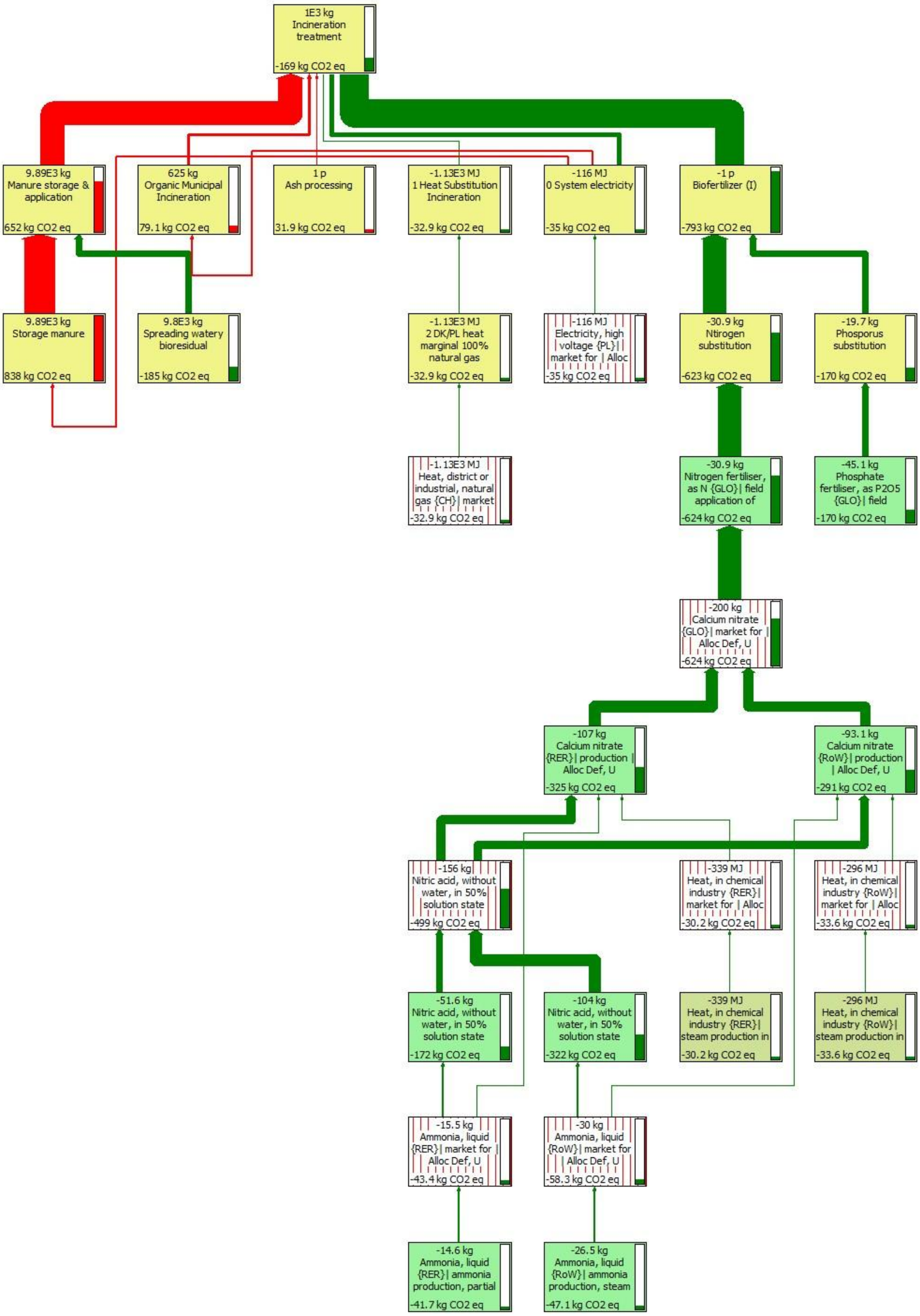

Figure 26. GHG flows for S0a. Red are net impacts and green are net savings

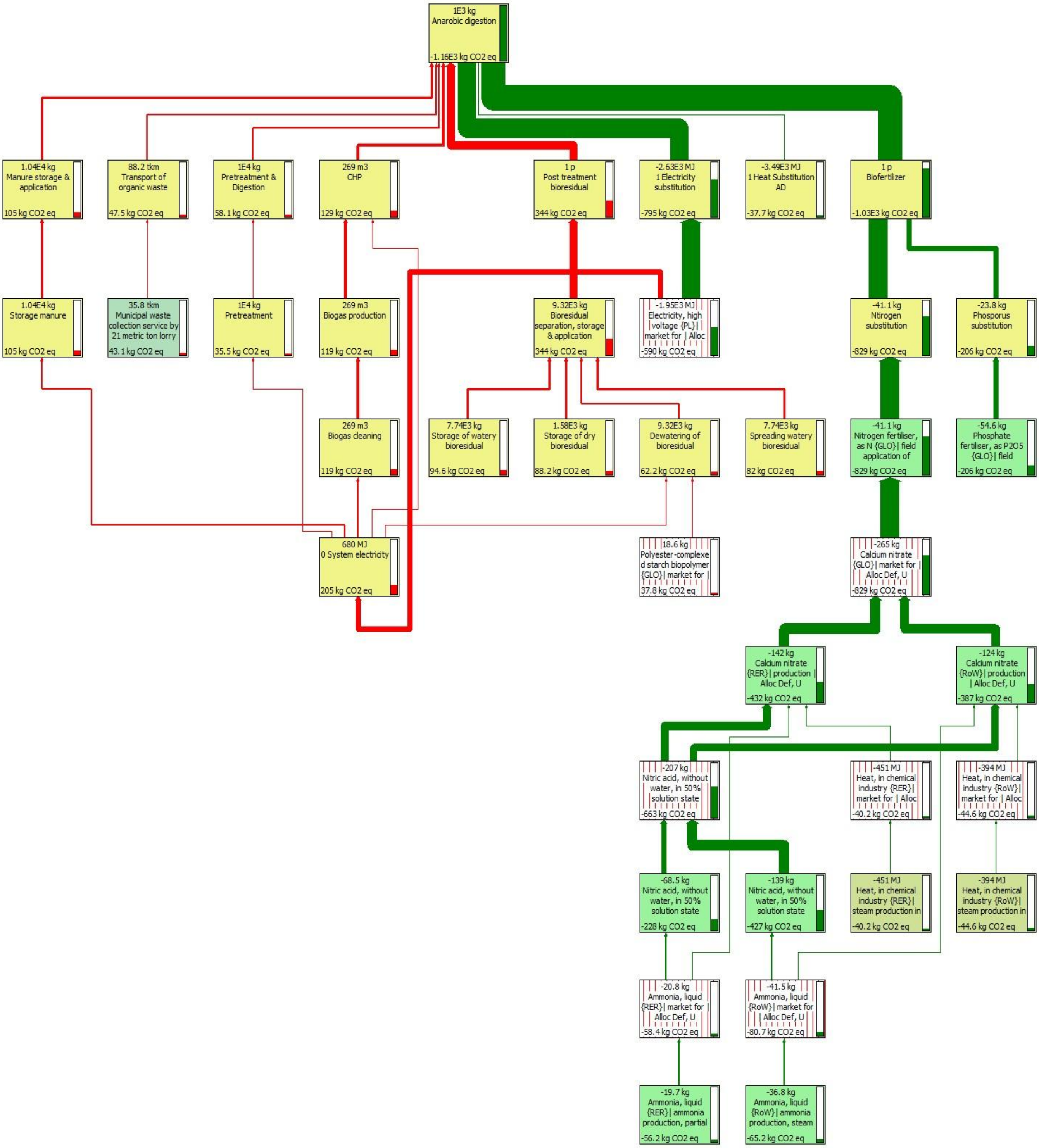

Figure 27. GHG flows for S3a. Red are net impacts and green are net savings

Appendix E: MFA (Modified scenarios)

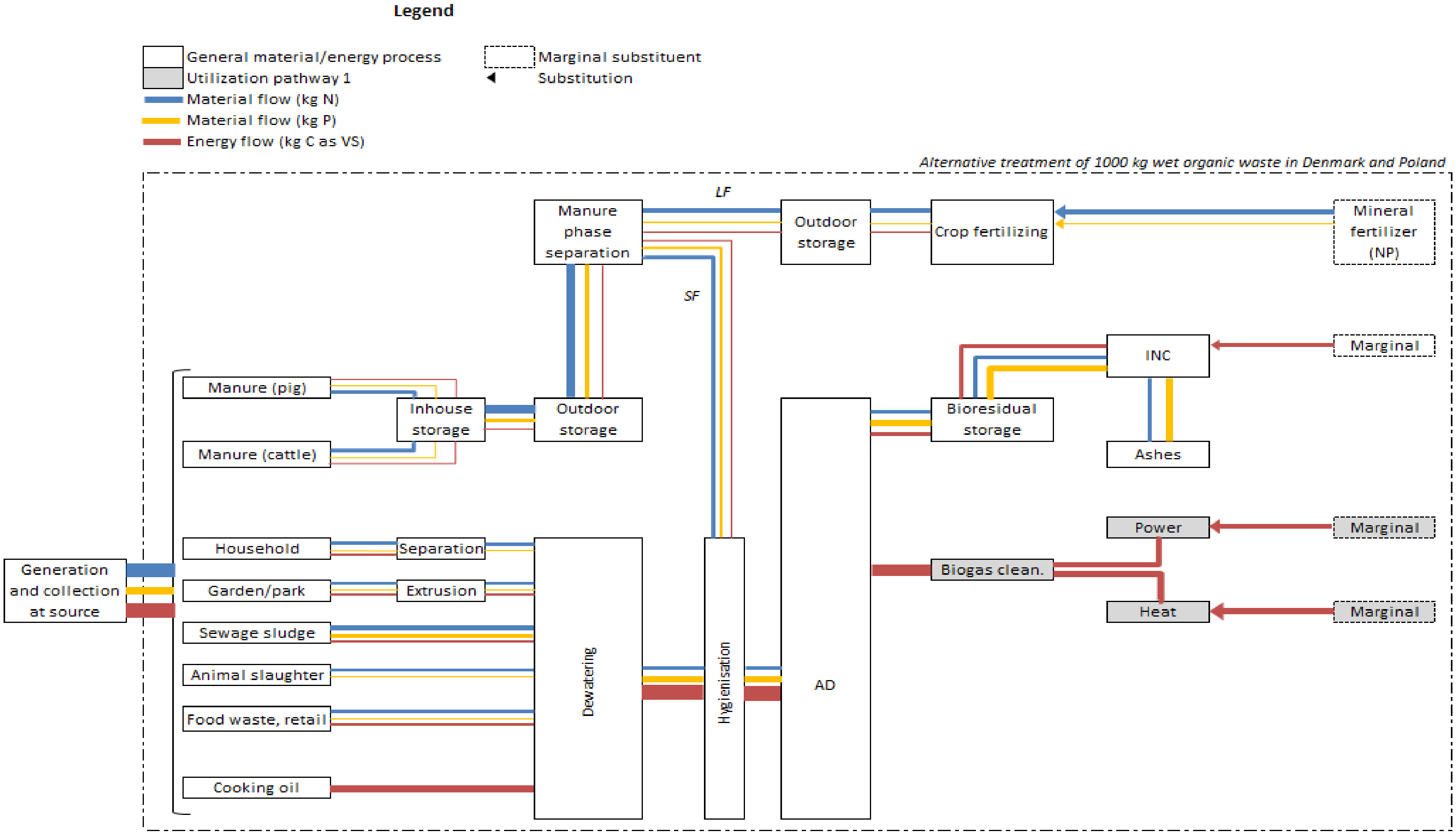

Figure 28. Conceptual illustration of MFA flow chart. The sankey is approximatly normalized according to the N, P, and VS data, here per 1000 t ww. The nutrients are not to scale to energy flows