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Life cycle assessment of biogas production from organic waste sources in a Norwegian context

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Preface

The report have been written as a study assignment in the Industrial Ecology master programme at the Norwegian University of Science and Technology (NTNU) and is considered a contributing part to the BIOTENMARE research project.

First and foremost I want to give a well-deserved special thanks to Carine Grossrieder for her invaluable tutoring in SimaPro 8 and her willingness to follow up and support, even though outside her job description. I also want to give special thanks to Raymond Jørgensen at Frevar and Tore Fløan at ECOPRO for good cooperation and their sincere wish to contribute to both qualitative information and general overview, particular for Norwegian conditions. I also want to thank them for receiving me and my colleges with open arms and for giving us guided tours of their respective biogas plants. I also want to thank Hellen Hamilton for nice conversations and her positive attitude towards aiding me when I needed help. A special thanks goes also to Fredrik E. Solberg for good discussions about model development, in particular for the MFA modelling, and as a friend I always can ask for help and support. At last I want to thank Helge Brattebø for good discussions with resulting fruitful supervision and that he has let me contribute in the BIOTENMARE project by allowing me to do research in a scientific field I love and have particular interest for.

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Abstract

The waste management sector have attained increasing focus towards minimising environmental impact, resource recovery and efficiency. EU now bans disposal of organic wastes, in order to reduce groundwater pollution and greenhouse gas emission. Alternative organic waste treatment methods have therefore been implemented and studied. Biogas production from anaerobic digestion is one such treatment method, which have shown promising results for this purpose.

Biogas production produces two high utility products, biogas and bioresidual, that can reduce the consumption of fossil fuel and mineral fertilizer, respectively. As such, the impact reduction from biogas production can have implication for several sectors. These include waste management, agriculture and fossil fuel consuming sectors, such as the transportation sector.

In this study the purpose is to assess the environmental impact of the value chain for organic waste treatment, implementing substrates from several sectors. The study includes collection of organic waste for application of recovered energy and nutrient recycling, focusing also on the system expansion possibilities. A MFA based LCA model was developed in SimaPro 8, to assess environmental effect of biogas production in Norway in relation to the most common treatment method of today. Energy and nutrient recovery rates are included in this assessment.

The results from this study confirms previous studies by reviling that biogas production and end product utilisation contribute to low or net negative global warming potential (GWP) and fossil depletion potential (FDP). These results are in relation to the product it substitutes.

In relation to the compared treatment alternative, incineration of organic waste and manure applied as fertilizer, further strengthens assumption that biogas production is a beneficial solution, with respect to environmental impacts. Results also point to increased energy recovery rates and possible increased nutrient recovery.

For other impact categories is the situation different. The result found in this study show increased impacts for both human toxicity potential (HTP) and terrestrial acidification potential (TAP) relative to substituted products. The same is true for biogas production compared to the common treatment method for HTP but with various results for TAP.

The conclusion is that biogas production is beneficial in terms of GWP and FDP. The main stressors causing GWP impacts for biogas production are fossil CO_2 form transport of organic waste, biogenic CH_4 from storage of bioresidual or biogas post treatment, as well as N_2O form application of bioresidual. FDP occur due to crude oil extraction for fossil fuels in transportation. TAP is caused exclusively (97 -99%) by NH_3 emission. HTP main source is the level of heavy metal (HM) in bioresidual when applied for agricultural purposes.

Sammendrag

Avfall sektoren har fått større fokus de senere årene som en mulig kilde til å redusere miljøpåvirkninger, øke resurs gjenvinning samt effektivitet. Eu har nå forbudt deponering av organisk materiale for å redusere grunnvannsforurensning og klimagassutslipp. Alternative behandlings metoder har derfor blitt utredet og bygd for å dekke kravet om behandling av organisk avfall. Biogass produksjon fra anaerob forråtning er en av de behandlings metodene som har fått økt fokus de senere årene, og gjennom mange studier har vist seg å være en av de beste behandling metodene.

Biogass produksjon gir to sluttprodukter, biogass og biorest, som kan benyttes til å redusere forbruket av fossil energi samt kunstgjødsel. På grunn av dette kan biogass produksjon redusere miljøpåvirkninger fra flere sektorer som avfalls behandling, jordbruk samt fossilt intensive sektorer som transport.

Denne studien tar for seg ulike miljøpåvirkninger gjennom hele livssyklusen relatert til verdikjeden for organisk avfallsbehandling fra en rekke sektorer. Studien tar for seg organisk avfallsbehandling fra innsamling av organisk avfall substrat samt mulige bruks områder for biogass samt biorest. Modellen utviklet i SimaPro 8, for å gjennomføre denne LCA studien, er basert på prinsipper fra MFA metodikk.

Resultatene fra denne studien underbygger tidligere studier på området, som tilsier at biogass produksjon er en god strategi for å redusere miljøpåvirkninger fra blant annet avfallssektoren. Biogassproduksjon fører til reduserte utslipp for både klimagassutslipp (GWP) samt forbruk av fossile kilder (FDP). Resultatene fra denne studien peker mot økt gjenvinningsgrad av energi og muligheter for økt næringsgjenvinning i forhold til referanse situasjonen. Referansesituasjonen i Norge er forbrenning av organisk avfall, da med annen type avfall, hvor gjødselen fra jordbruket blir benyttet direkte til spredning uten annen behandling.

For andre miljøkategorier peker resultatene en annen retning. Menneskelig toksisitetspotensiale (HTP) har i denne studien vist til store økninger i alle biogass casene, både i forhold til substituerte produkter men i tillegg sammenlignet med alternativ behandling. Samme tendensen gjelder for forsurings potensial for landområder (TAP), men hvor forskjellen mellom referanse case og biogass produksjon er mindre og hvor biogass i enkelte sammen henger gir lavere påvirkning.

Hovedkonklusjonen er at biomasseproduksjon er fordelaktig for GWP og FDP. For GWP er fossil CO_2 fra transport samt unngåtte utslipp den viktigste stressoren sammen med CH_4 fra lagring av biorest samt etter behandling av biogass. N_2O fra bruk av biorest som gjødsel er dent tredje største faktoren her. FDP er et resultat av uthenting av råolje til produksjon av drivstoff. HTP kommer hovedsakelig fra tungmetaller i bioresten anvendt som gjødsel, mens TAP kommer i all hovedsak (97 - 99%) fra NH_3 fra spredning av biorest.

Nomenclature

Biogas	A gases produced by bacteria due to decomposition of organic material
Biomethane	Biogas with a content of >97.5% CH_4,
Bioresidual	Remaining organic and inorganic solids after digestion
Biofertilizer	Bioresidual applied as fertiliser
CBG	Compressed biogas
CHP	Combine Heat and Power
CH_4	Methane
CO_2	Carbon dioxide
Bioresidual	The undigested leftover after anaerobic digestion
GHG	Greenhouse gas emission
HM	Heavy metal
HTP	Human toxicity potential
GWP	Global warming potential
FDP	Fossil depletion potential
kWh	kilo Watt hour, unit of energy during one hour, 3.6 MJ / 1 kWh
LBG	Liquid biogas, >98.5% CH_4
LCA	Life cycle assessment
MJ	Mega Joule, unit of energy effect
N_2O	Dinitrous monoxide
NH_3	Ammonia
Nm^3	Normal cubic, one cubic of gas at 0 degrees Celsius
MFA	Material flow analysis
OMW	Organic municipal waste
OIW	Organic industrial waste
Sm^3	Standard cubic, one cubic of gas at 15 degrees Celsius
SwSl	Sewage sludge
TAP	Terrestrial acidification potential
VS	Volatile solids

Keywords: Life cycle assessment, Material flow analysis, biogas, biomethane, organic waste treatment, bioresidual, biofertilizer, organic waste incineration

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1. Goal and scope

In this chapter is this the background, objective and scope of the project described along with the contexts and purpose. A further explanation of the goal and scope will be assessed in chapter 3. This chapter is to give an introduction to the life cycle assessment (LCA) that have been performed in this study, and is an important part of any LCA study.

1.1. Background

Norway has a goal of reducing the environmental impact and have in this context, signed several agreements, such as the 2020 greenhouse gas reduction agreement, for restriction and reductions of pollutants. Agriculture, waste management and usage of fossil fuels are major contributors to the release of GHG's such as CH_4, N_2O and CO_2. A growing focus toward material and energy recycling have therefore been the focus of the last years politics, reaching a major breakthrough with EU's ban on disposal of organic wastes to landfill.

Anaerobic treatment of organic wastes with biogas and biofertilizer production, have been analysed in several long term studies and found to be a highly recommendable treatment solution. Wastes, such as Manure, Organic Municipal Waste (OIW), Organic fats, Organic Industrial Waste (OIW) and Sewage Sludge (SwSl) are categorised organic waste substrates that anaerobic bacteria can digest and decomposed into biogas. Undigested volatile solids and inorganic solids are considered the dry fraction of the bioresidual, which can be applied as biofertilizer. Biofertilizer can thereby recycle nutrients and limit the use of mineral fertilizers (P) or artificial (N). Biogas can be used directly to produce heat and electricity or it can be upgraded to biomethane and LBG. These methane-purified gases have a higher level of utility than biogas due to a higher energy density and lower contamination of corrosive gases. Biomethane is commonly used as fuel or can be mixed with natural gas in a remote grid. A downstream utilization of the end products from anaerobic digestion have the benefits of avoided emissions, thus resulting in avoided environmental impacts. For some impact categories however, can the biogas production value chain result in an increased level of stress, which is of interest to assess.

1.2. Objective

The objective in this study where to perform an LCA of biogas production and compare the treatment alternative with likely treatment options. The focus have been on environmental impacts such as (global warming potential, human toxicity, fossil depletion and terrestrial acidification, etc.) and how given assumptions and critical variables affect the environmental performance for a given treatment option. This study will focus on anaerobic treatment where biogas and bioresidual is utilized as substituting products in terms of common energy carriers and fertilizers, in a Norwegian context.

1.3. Boundaries

The goal of the project is to generate a model that can assess environmental performances of organic wastes in Norway. The study does not include any economic aspects or estimates. It is developed and analysed mainly as an LCA and any MFA is only complementary to identify the inventory required to perform the LCA. No specific case have been analysed, but several biogas plants have been used as inspiration for case development. Data have been collected from both state of the art literature and acquired information form empirical studies, executed in relation to this study, have been applied as sources. The mixed data have been applied both in the literature study and as foundation for the model development in SimaPro 8. This includes specific site data achieved through direct contact and earlier studies.

The analysis of the impacts is only midpoint characterized and any further impact craterisation is not to be considered, this includes endpoint characterization and weighting of impact categories. The system is defined and only the most relevant flows have been assessed in the study, as figured in Appendix 2. Several organic waste substrates have been excluded because it is highly unlikely that any of these are to be applied as feedstock for biogas production in Norway. The system is to consider system expansion which special focus towards assessing life cycle substitution effects by applying biogas products and bioresidual for a wide spectre of commonly used products of today. It is also of high interest to identifying the most common treatment options and assess how these affect the overall life cycle impact. All best available technologies (BAT) for storage and application of bioresidual have been excluded in this report.

1.4. Research question

The development of the research question and functional unit (FU) have been evaluated thought the methodology study. In this partial chapter is these defines an put in context to the study at hand.

To fulfil the goal and objective of this master have a research question been formulated. The purpose of the research question is to pinpoint the objective and goal, contributing to a higher degree of specification of the problem at hand. This study has been written in a Norwegian context and is focused towards critical variables that affect the environmental performance of this system. There is many aspect that have been included in this project such as a literature study, methodology study and model development. The most important is the model development whereas the literature and methodology studies are only complementary to the model development itself. This along with the focus on environmental impacts generated from the treatment alternatives have made it clear that a parameter and variable analysis is the core of the project. The results and discussion are to highlight life cycle impacts of the waste treatment of organic substrates. This is to be performed in the context earlier described and to assess the variables responsible for causing or reducing the overall environmental impact including their process of origin. The conclusion are to summarise the most important findings from this study.

With the focus of interest, context and scope limitations have the research question been defined as:

> "Which stressors are critically influencing the environmental life cycle impact of the biogas production in Norway, in comparison to the alternative organic waste treatment option, and which factors and variables limits or enhance these"

1.5. Functional unit

It was important that the functional unit enables the system to be comparable to major changes and still be comparable with the different results generated by the model developed in SimaPro 8, for each of the five cases. It was also, as described later in the methodology chapter, important that the research question is short and consistent. In this project there are several substrates that are treated and products being made. To be able to fulfil the goal and objective of this project a common constant had to be identified, something that remain the same in all cases for the model. Organic waste, have such a role in the model and have thus been chosen as the functional unit. It is described as follow*: The treatment of one ton dry matter organic waste substrate*.

The estimated energy and nutrient recovery as well as the environmental life cycle impacts will therefore be relative to this defined FU. Organic waste treatment contain large quantum of water, which in this FU is included by assessing the water content of each of the included

substrates, as explained later in this report. Nutrient and energy composition will be assessed in the same manner as the water content.

2. Literature and empirical study

The literature study in this thesis considers previous LCA of organic wastes, technological options for organic waste treatment, in particular for anaerobic treatment and an empirical study of the treatment situation in Norway. The objective of this part of the study is to identify previous LCA studies and assess their findings to identify challenges concerning such studies and guidelines for data acquisition. There is a particular focus towards identifying the situation of anaerobic treatment and the alternative treatment method, in Norway particularly. Identifying important technologies and organic waste substrates, with respect to the parameter packages these represent is to be one of the main topics trough this review. In addition is possible utility purposes for biogas, bioresidual and for the produced heat from municipal incineration of organic wastes important to establish. In the end is a summary of the empirical data collected during field trips to Ecopro in Verdal and Frevar in Fredrikstad, presented.

2.1. LCA of organic waste treatment and challenges

LCA is a tool used to assess environmental impacts caused by the use or production of products, systems or services (Baumann & Tillman 2004a; Benoît & Mazijn 2009). The impact caused by these systems can be estimated with substantial certainty as they require a given set of input products, which can have predefined stressors caused by a given unit of use, the FU. Waste management scenarios pose a much higher degree of uncertainty. This is due to the fact waste is composed of a large variety of products that often are unknown and where the stressor inventory is much less defined. To solve this problem, an LCA of waste management requires data from various sources to satisfy a complete LCI. Data from various sources are often generic and does not give the exact waste composition for the case assessed. Therefore, the level of uncertainty is much higher in such studies and the results can often just be interpreted as close assumptions for the given case, thus not give a 100% correct impact estimate (Clavreul et al. 2012). It is therefore required that LCA studies are clear in which assumptions have been made and in which manner they have been acquired, particularly for the parameters that have a profound impact on the system.

The acquisition of LCI data can be defined in three major methodological approaches: (1) default variables, (2) theoretical technical data and (3) onsite-specific measurements (Clavreul et al. 2012). Default variables are highly generic data that cause a high level of uncertainty for a specific case, but give good generic data. Theoretical technical data describes processes that are defined by physical laws (natural gas law etc.). This gives the practitioner the possibility to calculate exact data, but it requires more work and a high level of detail. Onsite-specific measurements give highly reliable data that is relatively easy to access for a specific case. It is however, much less applicable in a generic context because every case is unique. Site measurements therefore lack the normal distributional that average generic data offers. It is also

important to be clear which method that have been applied in estimating the impact from the given LCI to diminish the uncertainty of methodological approach.

In LCA are there several methods to assess production systems and services of various complexity. This generates a source of uncertainty that have to be further assessed. For waste management this is of particular importance as waste is a commodity that has to be treated or disposed of in any case, and is thereby an inevitable process of any product. This makes it a necessity to compare waste management to a reference scenario so as to gain meaningful results (Clavreul et al. 2012). If not, the analysis is just a part in determining the impact caused by waste handling of a given product. For waste management LCA this is not the goal, but to assess the total impact of different treatment options of a homogenous waste mix. In conclusion it is important that the practitioner is aware of and transparent about the uncertainty level. Therefore an extra effort toward transparency for assumptions, sensitivity analysis and uncertainty propagation should be implemented in such studies (Clavreul et al. 2012).

2.2. Previous LCAs of organic waste

A wide variety of LCA's have been applied to biogas production, anaerobic digestion and other organic waste treatment options the last decade.

The observed tendency, is that the actual impact caused by one or another treatment type highly varies dependent on the approach of study, area of interest and treatment option chosen for the given study. This makes it difficult to highlight specific results that are comparable between studies, but a general tendency towards a net benefit for several impact categories for anaerobic treatment compared to incineration has been witnessed (Poeschl et al. 2012a; Hamelin et al. 2014; Modahl et al. 2014; Khalid et al. 2011; Bernstad & Jansen 2011; Lyng et al. 2011). Organic waste treatment is a subject of several treatment options, that in turn has a high degree of technological variation (Bernstad & Jansen 2011). This is especially true for anaerobic digestion treatment and biogas production systems where the end-product utility is an important part of the overall handling of the organic waste (Poeschl et al. 2012a; Modahl et al. 2014; Lyng et al. 2011). The end-products, biogas and bioresidual, has the possibility to substitute energy and fertilizer, respectively (Modahl et al. 2014). Such an assessment method of the end-products may generate negative net emissions due to the potentially saved impacts caused by the products substituted, such as mineral P and artificial N. For biogas can a multitude of energy carriers as fossil fuel commodities, natural gas in grid, remote heat or electricity in addition to other bio-fuels.

Several state of the art studies has concluded that anaerobic digestion options yield negative net GHG- emissions, but there is an increase in both nutrient enrichment and terrestrial acidification potential (TAP) compared to incineration (Bernstad & Jansen 2011; Lyng et al. 2011; Modahl et al. 2014). Other impact categories such as photochemical oxidant formation (POF),

particulate matter formation (PMF) and fossil depletion (FDP) contribute to net savings compared to the reference scenario, incineration (Hung & Solli 2012).

In the two continuous studies Poeschl et al. (2012a) and Poeschl et al. (2012b) several biogas plants were been assessed in a comprehensive LCA and the results were compared. Their results indicate that the biogas production is only yielding negative impacts for small biogas plants. This study is performed in a German context where the German electricity mix is applied. For example heat requirements were covered by heat from natural gas. The study found positive impacts for GWP in the range of 191.68 kg CO_2 eq for biomethane to natural gas substitution and 204.16 kg CO_2 eq for fuel substitution. Only in the case where biomethane is applied for fuels cell application with 60% heat recovery are the GWP emissions net negative (-110.26). It was fond however that application of bioresidual results in net negative impact, for all cases. The environmental impact is highly dependent on the type of waste being assessed in the feedstock, the study concludes. Their result is that the substrate yielding the most net negative impacts is straw. For cattle manure the result is – 23.22 kg CO_2 eq, while municipal solid waste and slaughter house waste yields – 53.05 and 50.6 kg CO_2 eq respectively.

2.3. Feedstock waste and effects on anaerobic processing

I this chapter, the relationship between organic waste substrates and anaerobic processing is presented. The findings comes from several state of the art studies, to secure complete data, eliminate epistemic[1] uncertainty and system understanding in addition to the stochastic[2] variations often found by surveying various studies.

Organic substrates and effects

The organic waste treated, highly determines the outcome of biogas production. This is mainly due to large variations in properties for each individual organic substrate (Carlsson & Uldal 2009; Poeschl et al. 2012a; Rehl & Müller 2011). Organic compounds consist mainly of the three organic carbons fat, protein and carbohydrates. These three, make out the volatile solid fraction (VS) of the organic substrate, table 1. The fraction that is not VS are considered ash weight (Carlsson & Uldal 2009).

[1] Incomplete knowledge
[2] Natural fluctuations or variations

Table 1: Methane production from fat, proteins and carbohydrates

Organic compound	Biogas Nm3/kg/ VS	Methane Nm3/kg VS	Methane %
Fat	1.37	0.96	70
Protein	0.64	0.51	80
Carbohydrates	0.84	0.42	50

Source: Extracted from Raadal & Morken (2008).

Many factors sustain, inhibits or increases the biogas production. The most notable of these factors are the alkalinity, VS, degrability, water content, nutrient composition, pH and temperature (Khalid et al. 2011; Søheim et al. 2010; Carlsson & Uldal 2009; Poeschl et al. 2012a; Hamelin et al. 2014; Bernstad & Jansen 2011; Raadal & Morken 2008; Jørgensen 2015; Li et al. 2010). This indicates that the substrate mix is important when assessing biogas production and operation.

The processing option affects the biogas outcome of the anaerobic digestion to a certain extent, but overall is it the feedstock that determine the main range of the biogas and bioresidual composition and magnitude (Alvarez & Lidén 2008; Poeschl et al. 2012a; Carlsson & Uldal 2009).

In the report "Substrathåndboken" by Carlsson & Uldal 2009 experiments on biogas yield from a large variety of organic products (homogenously) tested where assessed and listed. Their results are show major variety in in digestion time, also called hydraulic retention time (HTR), methane yield and pH from the various products due to the substrate composition (Raadal & Morken 2008; Schievano et al. 2011; Poeschl et al. 2012a), *table 1*. Many substrates contain organics, lignin or cellulose, that are difficult to digest, which thereby causes implications for the degrading of the VS, thus limiting the biogas production. This in turn, leads to a lower methane yield than normally assumed based on the VS content (Hamelin et al. 2014), table 2.

Table 2: Properties of different waste

Disaggregated substrates	DMC (%)	Volatile Solids of DMC (%)	Degradability(D) of VS	CH_4 %	Remaining solids (DS)[9]	CH_4 yield m3/ ton VS[1]
Cattle manure	8%[1]	80%[1]	62%[5]	65%[1]	50.40%	213.01
Pig manure	8%[1]	80%[1]	62%[5]	65%[1]	50.40%	268.01
Frying fat	90%[1]	100%[1]	100%[1]	68%[1]	0%	757.01
Organic municipal waste (OMW)	33%[1]	85%[1]	64%[5]	63%[1]	46%	461.01
Separated animal fats	4%[1,2]	95%[1]	100%[7]	60%[7]	5%	682.01
Fish processing waste (offal)	42%[1]	98%[1]	65%[6]	71%[1]	36%	930.01
Sorted restaurant food waste	27%[1]	87%[1]	85%[8]	63%[1]	30%	461.01

Slaughterhouse waste (blood)	10%[1]	95%[1]	65%[6]	63%[1]	38%	547.01
Slaughterhouse waste (entrails)	30%[3]	83%[3]	63%[3]	63%[3]	48%	688.03[3]
Slaughterhouse waste (offal)	16%[1]	83%[1]	65%[6]	68%[1]	46%	664.11
Diary processing	20%[1]	82%[5]	57%[5]	59%[1]	53%	277.05
Fruit and vegetable waste	15%[1]	95%[1]	57%[6]	60%[7]	46%	666.01
Sewage sludge	17%[4]	80%[4]	50%[8]	60%[4]	60%	336.04

Source: [1](Carlsson & Uldal 2009), [2](Gebauer & Eikebrokk 2006), [3] (Lyng et al. 2011) s. 22, [4](Wadahl 2014), [5] (Hamelin et al. 2014), [6]Assumption based on (Hamelin et al. 2014), [7]Other assumption, [8] (Sande et al. 2008). [9] Calculated as the remaining solids after digestion: $100\% - (VS\% * DS\%) = DS\%$

Anaerobic processing

Anaerobic digestion of organic material produces CH_4 (50 – 80%), CO_2 (15 – 40%), CO (0 – 0.3%), N_2 (1 – 5%), NH_3 (0 – 1%), O_2 (0 – 0.5%) , H_2 (0 – 0.3%), H_2S (0.05 – 1.5%) by bacterial decomposition of organic material (Morken et al. 2007; Seadi 2002), fig 1. This process consists of the three main stages hydrolysis, fermentation, and methagonese, where fermentation can be described as two separate processes acidogenesis and acetogenesis, figure 1. The efficiency of the digestion besides degrability gradient is closely connected to four main parameters: Temperature, digestion time, pH and NH4+/NH_3 concentration (Morken et al. 2007; Carlsson & Uldal 2009; Jørgensen 2015; Fløan 2015).

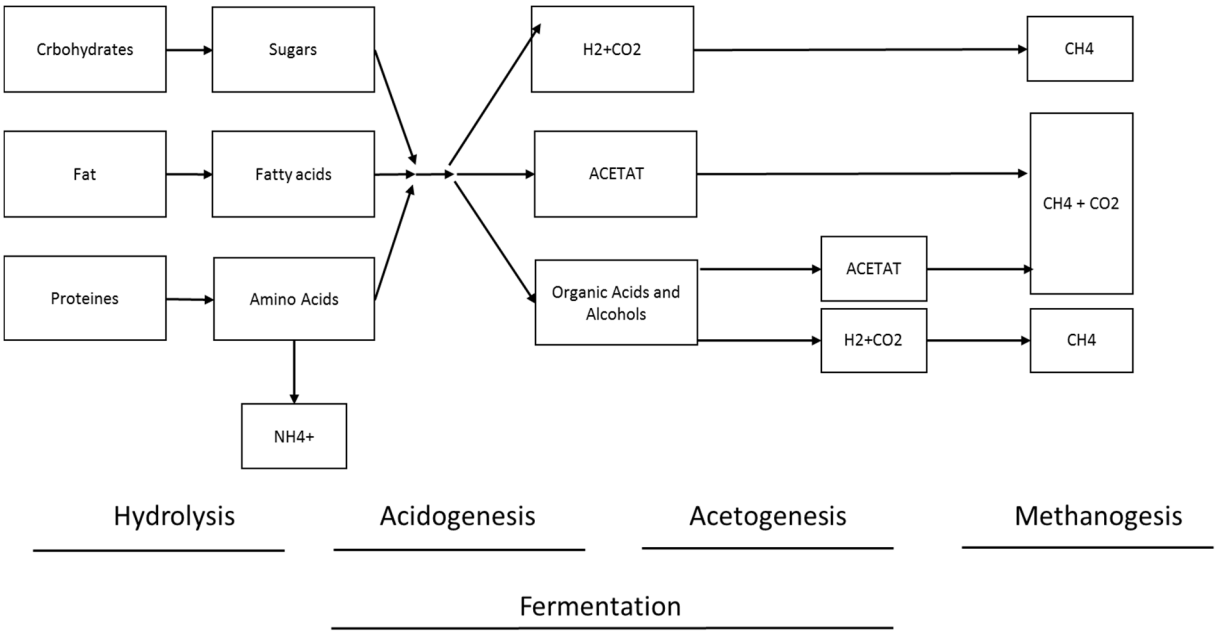

Figure 1: The biochemical stages of anaerobic digestion

Figure derived from figure 2 in Morken et al. (2007)

Temperature

Temperature is one of the most important factors for biogas production, as it determines the rate at which the bioresidual decomposes and the biogas yield (Ariunbaatar et al. 2014; Yingjian et al. 2014; Raadal & Morken 2008; Carlsson & Uldal 2009). There are mainly three types of anaerobic digestion determined by the temperature occurring, both naturally and by human intervention. These are psychrophilic (7-25°C), mesophilic (25-42 °C) and thermophilic (49-60 °C) (Morken et al. 2007). As the temperature increase so does the methane yield, the degrability rate and the total process energy consumption (Morken et al. 2007; Mellbin 2010; Ariunbaatar et al. 2014).

In landfill is the presence of psychrophilic bacteria is the source for the CH_4 and CO_2 emissions emitted. Due to the long landfill time, the organic waste fully degrades and thus the resulting biogas are released directly into the atmosphere. Psychrophilic degradation has a hydraulic retention time (HRT) of 40d (Morken et al. 2007) and is thus little convenient at an industrial scale. Therefore, the major alternative options are either mesophilic (20-25d) or thermophilic (8-15d) (Angelidaki & Ellegaard 2003; Morken et al. 2007). The mesophilic alternative has historically been thought to be easier to start and maintain, in addition to require less energy than the thermophilic option. In later years has experiences shown that the thermophilic process are just as stable as the mesophilic, and thus deemed preferable du to the shorter HRT.

Ammonia and ammonium inhibition

The concentration of NH4+ in the digested material comes from several sources such as poultry manure, slaughter house waste offal and proteins, figure 1 (Carlsson & Uldal 2009; Morken et al. 2007). This affects the methane production mainly in two ways, by inhibiting hydrogen (H) to form bond with carbon (C) by forming NH3 (Morken et al. 2007) and by creating a toxic environment for the bacteria (Carlsson & Uldal 2009; Britto & Kronzucker 2002). Reversely works NH_4+ and NH_3 as a pH buffers as ammonia is converted, in the simplified reaction NH_3+H→NH_4+, in an acidic environment, where responsively NH_4+→NH_3+ H+ in a basic environment (Britto & Kronzucker 2002). This has also been the foundation of several NH_3 inhibiting technologies used for bioresidual and manure storage (Bernstad & Jansen 2011; Amon et al. 2006; Luostarinen et al. 2011). The toxic and inhibiting effect ammonium has on the anaerobic process disturbs the stability of the anaerobic process and is therefore an unwanted compound in the process (Ariunbaatar et al. 2014).

pH

The pH level are important for the process as it generates the conditions, in which the bacteria lives. Biogas production optimum occur at pH of 6.5 to 7.5 (Carlsson & Uldal 2009; Morken et al. 2007). During the hydrolysis decreases the pH, which is essential to maintain around the optimal pH 7. How low the pH reaches in the hydrolysis depends on the alkalinity of the

substrate in the process. The acidification occurs mainly due to increased CO_2 levels in the substrate and the increasing presence of acids from proteins (Carlsson & Uldal 2009), figure 1. To solve this problem can additives with a pH higher than 7.0, such as poultry manure or chemicals such as sodium bicarbonate (NaHCO3) and bicarbonate (HCO3) be used to increase the alkalinity. If the goal is to instantly increase the pH can Lye (sodium hydroxide, NaHO) be added to the process (Hauge 2014). Using Lye at a regular basis makes the process highly unbalanced and it will be difficult to maintain an optimal pH level over any length of time (Jørgensen 2015).

Co-digestion

There are many studies that describes the many benefits and often necessity for co-digestion of different substrates to maintain the biogas production (Carlsson & Uldal 2009; Raadal & Morken 2008; Modahl et al. 2014; Lyng et al. 2011; Luostarinen et al. 2011). However, few has included a quantitative benefit assumption in their assessments because of the high uncertainty it poses.

The main benefits of co-digestion described in Carlsson & Uldal (2009) are pH buffering capacity, good mix of nutrients for the bacteria and a possibility for increased methane yield. Co-digestion reduces or excludes the need for additives to the system such as micro nutrients, pH regulating chemicals (HCL and CaCH3) or buffering chemicals such as $Ca(HCO_3)_2$. In some instants, based on the water content of the feedstock, can virgin water consumption be reduced. The more the bacteria manages themselves without the interference of human activity, the more stable and resilient the process becomes (Jørgensen 2015). Because of this, anaerobic digestion in Norway is in most cases, operated solely by the bacteria alone as described above. There is exceptions to this procedure and a homogenous feedstock has been, in all of these cases, the reason for those exceptions, (Jørgensen 2015). Manure slurry has proven to be a good co-digestion substrate due to its nutrient rich content water witch is essential for the anaerobic bacteria cultures (Labatut et al. 2011).

A quantitative co-digestion benefit of food waste and diary manure have been estimated to be in the range of 0.8 to 5.5 compared to digestion of manure alone (Li et al. 2010). A change in degrability has been the observed cause of this, which also affects the HRT during the acidification stage, figure 1. The optimal mixing ration observed is 6:1 food waste over manure and a acidification HRT of one day (Li et al. 2010).

Achieving a maximum biogas yield is a complex process that depends on the mix and type of input substrate and the nutrient ratio found within the bio-waste. Interestingly has a relationship between high amounts of water and an increase in methane yield been discovered, due to the solubility that CO_2 has in water (Raadal & Morken 2008).

Biogas production, conclusion in literature

There have been observed a very complex biochemical system for biogas production and methane production yields, based on anaerobic bacterial produced biogas (Carlsson & Uldal 2009; Luostarinen et al. 2011; Bernstad & Jansen 2011; Morken et al. 2007). By having identified the most important parameters, it is possible to process organic waste anaerobically in an efficient way (Bernstad & Jansen 2011). There is however, not an easy task to predict the actual biogas production. In several studies, lower theoretical methane yield has been discovered than the actual been observed yield (Carlsson & Uldal 2009; Visvanathan 2014). The main conclusion derived from these findings is that there is no absolute truth when it comes to methane yield theory, but that the theoretical potential is a guideline to assume the total production. To solve this problem, an onsite measurement is necessary to give any accurate information, where the result might prove to be somewhat misleading because of variations over time and from sample to sample. This requires special attention when trying to predict the output of methane based on the input substrate of choice (Carlsson & Uldal 2009) and should therefore be focused upon in the uncertainty assessment (Clavreul et al. 2012).

2.4. End product utility and substitution effects

The main driver for applying anaerobic digestion as a treatment alternative is the recycling potential of energy and nutrients the generated end-products offers (Raadal & Morken 2008; Ariunbaatar et al. 2014; Lyng et al. 2011). There are mainly two by-products produced in the anaerobic digester, biogas and bioresidual. The bioresidual can be used for soil improvement or directly as fertilizer dependent on the heavy metal composition (Tormod Briseid 2010; Lyng et al. 2011; Luostarinen et al. 2011; Landbruks- og matdepatrementet, Klima- og miljødepartementet 2003). The bioresidual substitution effect are affected by several parameters such as nutrient content (N, P, Mg, K, C etc.), water content and the heavy metal content and the origin of the nutrient being substituted. Biogas, which is an energy carrier, can be applied for energy purposes an thusly substitute fossil or other common energy sources.

Bioresidual

Concentration of heavy metals are strictly regulated, both in EU and in Norway, and the greater the contamination the less utility and thus less substitution of conventional artificial or mineral fertilizers are possible. In Norway is it “Gjødselsforskriften” that covers the legalities for use of fertilizer and contamination limitation of bioresidual as a fertilizer (Landbruks- og matdepatrementet, Klima- og miljødepartementet 2003). The regulation distinguishes two forms of bioresidual, one from organic wastes and manure and one from or containing sewage sludge. From the point that the bioresidual is made or partly made of sewage sludge, the whole bioresidual body is treated as bioresidual from sewage sludge and a stricter use protocol is effective §15, 7. There are four levels of contamination in Norway, these are level 0, I, II and

III and are determined by the concentration of several heavy metals (HM) listed in table 3. If any of the HMs exceeds a class, the next class in effect even though the other HMs are within the restrictions levels several classes below.

The first level (0), allows the bioresidual to be used as fertilizer for root vegetables and fruits, § 24. Class 1 and 2 can be applied as fertilizer for surface dwelling crops. For bioresidual that contains sewage sludge there is special application procedure described in § 25. Sewage sludge containing biofertilizer cannot be applied for root vegetables and fruits because, these accumulates as substitutes for nutrients in plants and thus a higher ratio of unwanted bacteria and heavy metal contaminations might occur there. The bioresidual containing sewage sludge does also has to be ploughed into the soil within 18 hours after spreading, § 25, third sentence.

Table 3: Heavy metal concentration limitation for different bioresidual classes

Quality classes	0	I	II	III
mg/kg dry matter				
Cadmium (Cd)	0.4	0.8	2	5
Lead (Pb)	40	60	80	200
Mercury (Hg)	0.2	0.6	3	5
Nickel (Ni)	20	30	50	80
Sink (Zn)	150	400	800	1500
Copper (Cu)	50	150	650	1000
Chromium (Cr)	50	60	100	150

Table replicated from Gjødselsforskriften §10 (Landbruks- og matdepatrementet, Klima- og miljødepartementet 2003).

When applying the bioresidual the presence of nutrients and the plant availability of these plays a crucial role in the substitution benefit (Luostarinen et al. 2011; Tonini et al. 2014). Luostarinen et al. 2011 has identified that the N in the manure alone are only 20 – 30% plant available. Anaerobic digestion with some co-generation increases the N availability to 50 – 85% (Luostarinen et al. 2011), but at the same time reduces the total amount of both nutrients, N and P (Amon et al. 2006; Möller & Müller 2012). For P is it assumed 100% availability, due to no data on the field (Möller & Müller 2012). The loses of N and P occur mainly due to losses by dewatering where the wet body is not applied as fertilizer, (Fløan 2015) in addition to the formation of NH_3, N_2O and N_2 for the N losses (Bernstad & Jansen 2011). Loss of N and P is particularly severe if the bioresidual is separated and only one of the compartments, (wet or dry) is applied as fertilizer (Amon et al. 2006). The reason the loss is so extensive when only one of the bioresidual compartments are applied are due to differences in nutrient composition properties. 70% of N and 10% P found are in the wet body of the bioresidual (Poeschl et al. 2012a) and thus lost if not applied. Two studies, Möller & Müller (2012) and Rehl & Müller (2011), found that the average phosphors loss in the anaerobic digester is approximately 10%

of original P content. Why the phosphorus disappear is poorly documented and understood, but both their experiments resulted accordingly. A fair assumption for this losses could be due to retention in the digesters and storage tanks (Möller & Müller 2012). An aspect that is often forgotten when comparing the fertilization potential by application of biofertilizer is that it contains many more life essential minerals than just N and P. Commercial fertilizer often add N and P in addition to potassium (K) and the other minerals that is essential in smaller amounts is therefore not added such as sulphur (S) magnesium (Mg) etc. A net depletion of these minerals can lead to lower harvest yields, growth rate and quality of the food produced (Möller & Müller 2012; Fløan 2015).

Alternative treatment of manure

Manure that would, in a Norwegian non-anaerobic digestion treatment context, otherwise been applied directly as fertilizer makes it important that the bioresidual is given back to the farmer. Applying the bioresidual will continue to maintain the nutrient circle, in a greater extent than manure alone and secure that the farmers don't suffer economically due to loss of fertilizer (Raadal & Morken 2008; Lyng et al. 2011; KLIF 2013).

Biogas

Biogas can be utilized for several purposes such as heat and electricity production, upgraded for gas grid supplement and fuel purposes. For biogas to be used in CHP no other treatment than cleaning is required (Lyng et al. 2011; Börjesson & Berglund 2006). Due to the relatively low concentration of methane in biogas (45 - 80%) is the gas often burned directly for energy purposes. If the production is greater than the demand for energy is the overproduced gas often torched (Fløan 2015). When the biogas is applied for grid purposes or as fuel is a upgrading in quality to biomethane (>97.5% CH_4) required, which increases energy density and remove potential corrosive gases (Lyng et al. 2011; Luostarinen et al. 2011; Bauer et al. 2013; Hung & Solli 2011; Hung & Solli 2012; Møyland 2012).

Emissions and impact categorization

For organic waste emissions are there several compartments that is of relevance. These compartments (air, soil, ground water, lake, rivers and marine environments) are effected in various ways by release of different stressors (Goedkoop et al. 2009).

Stressors has a tendency to contribute to only a few types of stress and it therefore measured in a given unit that best describes those impacts and is multiplied by a characterization factor for this given unit. This collection of stressors based on the effect is called an impact category, which is the categories in which LCA results are presented measured and interpret as the first step in an LCAI. For biogas production is the most relevant impact categories Global Warming Potential (GWP) given in kg CO_2 eq, Terrestrial Acidification (AP) given in kg SO_2 eq, human

toxicity potential (HTP) given in kg 1.4 DB-eq and fossil depletion (FDP) given in kg oil eq. (Poeschl et al. 2012b; Modahl et al. 2014; Lyng et al. 2011). It has therefore been important to identify the stressor contribution to these impact categories.

Emission to air is caused by mainly the release of uncaptured CH_4, CO_2, NH_3 and N_2O from storage of manure and bioresidual (Luostarinen et al. 2011; Börjesson & Berglund 2006; Andersen et al. 2010; Muha et al. 2014; Amon et al. 2006; Bernstad & Jansen 2011). During digestion itself there is measured low to no emissions of gasses (0 - 1% or 0.5 – 8% of produced biogas) because the treatment occurs in closed containers and pipes (Lyng et al. 2011; Jørgensen 2015; Fløan 2015). This is very case specific and is determined by the type of plant used. In Norwegian biogas plants is it most common to have water traps in the gas pipes so that the gas does not leak out of the system and as a result does these companies operate with no losses during digestion (Jørgensen 2015).

The two main emission stages, storage of manure and bioresidual is composed of as previously explained of two types of emissions, can be divided into C based and N based emissions (Muha et al. 2014). CH_4 and CO_2 is the two major C based gases that is released and both contributes mainly to the GWP category and is measured in terms of kg CO_2 eq (Goedkoop et al. 2009). NH_3, N_2O and N_2 is the main N gasses produced but they contributes to different impact categories. NH_3 has a great impact on TAP which is measured in kg SO_2 eq (Goedkoop et al. 2009). N_2O is a major contributor to GWP with a measured effect of 298 times the GWP of CO_2 (Ecoinvent 2015). For biogenic CH_4 is the characterisation factor 22.3 kg CO_2 eq per kg. In comparison, is the fossil CH_4 25 kg CO_2 eq per kg (Ecoinvent 2015).

N_2 is a non reactive gas that makes up 79% of our atmosphere and has therefore no apparent impact, but contributes to losses of bioresidual mass and N fertilization potential (Möller & Müller 2012; Bernstad & Jansen 2011). Emission such as those mentioned above occur in different stages of the organic waste treatment processing such as post storage or by spreading (Luostarinen et al. 2011; Amon et al. 2006; Modahl et al. 2014), Appendix 12 - 16. CH_4 generation mainly occur due to remaining degradable material in the bioresidual, which is still under anaerobic condition (Amon et al. 2006; Bernstad & Jansen 2011). An aeriation of the bioresidual would therefore prevent unwanted post-produced biogas, Appendix 13. By covering the bioresidual with straw instead of wooden lid an increase of CH_4 would occur, but at the same time reduce the generation of N_2O and NH_3 gases (Luostarinen et al. 2011). This effect occurs because more undigested carbon yields greater CH_4 potential which the straw introduces (Amon et al. 2006). N_2O and NH_3 is mainly produced when the C/N ratio is high and added carbon reduces this value, and thus the potential NH_3 and N_2O formation (Bernstad & Jansen 2011). By digesting slurry the greatest benefit of methane capture and emission is evident, because of the carbon has already been transformed into biogas and thus the remaining C content is relatively low. As expected is the C/N ratio increased during anaerobic digestion and

the NH_3 and N_2O potential is thereby increased, resulting in greater emission rates of these gases if no inhibition measures are implemented (Bernstad & Jansen 2011; Luostarinen et al. 2011). As previously described, can a slight acidification of the bioresidual reduce the formation of NH3 and is among one of the most common inhibition methods applied. By using best available technology (BAT) can the overall reduction in N_2O emission be up to 64% (Luostarinen et al. 2011). For commercial N fertilizer can an emission rate of 2% of the total applied N might occur, but mainly determined by the pH of the soil it enters and poses therefore a high degree of uncertainty (Bernstad & Jansen 2011; Luostarinen et al. 2011). The production of N based gases varies a lot based on pH, C/N ratio, treatment option etc. The division of the N based gases has been found to vary drastically between different treatment technologies, table 4.

Table 4: N based emission partition

Treatment type	NH_3	N_2O	N_2	Sources in paper
AD	96%[1]	0.77%[1]	3.23%[2]	Chung (2007)[3]
Composting	2.40%	1.40%	96.20%	Sonesson (1996)[3]

[1]Found in line 298 and 299 for NH_3 and N_2O respectively. [2] N_2 is found by 100% - NH3% - N_2O. [3] None of these sources where found, and therefore have the study that this is gathered from the main source. Source: Appendix 11 is just a short summary of table 5 in the paper (Bernstad & Jansen 2011), pg 1883.

2.5. Assessment of emissions and literature recommendations

The most common way to assess emissions in waste LCA is by applying transfer coefficient and emission rates found in the literature or by performing onsite measurements (Baumann & Tillman 2004a; Carlsson & Uldal 2009). This is however, yielding unnecessary high uncertainties in a generic context (Clavreul et al. 2012). Muha et al. (2014), another study that assess the emissions from anaerobic treatment, has a rather blunt approach to the calculation of methane emissions caused in the storage tank. The CH_4 emissions is calculated based on the methane yield (MY) per ton dry matter (DM) volatile solids (VS) with a subtraction of the amount already digested in the anaerobic process, determined by the degrability coefficient (Dg) and the average methane yield. Such an approach secures a mass balance correct estimation of emissions and the remaining bioresidual. The reliability of a study that has performed this approach should be higher, they argue, than a study based of predefined emission parameters. This is particularly important for co-digestion facilities because the amount of C and N vary much with the composition of the current feedstock batch in process. The same approach can to some extent be applied for N based emissions but a greater dependence on N loss from other sources is necessary and is thus a source of uncertainty.

Emission to soil is as previously explained caused by increasing levels in heavy metal concentration, but also the remaining levels of chemical oxygen demand (COD) and biological oxygen demand (BOD). High levels of these generates a anaerobic soil that is poor for life and has the potential to continue to produce biogas, which will be released directly to the atmosphere. A high biogas yield reduces the remaining COD and BOD and is therefore preferable in this context as well.

Emission to waterbodies[3] comes in mainly two forms, leaching of nutrients from agricultural spreading and COD and BOD of the leached material. The nutrient N and P is worldwide a common problem due to agriculture and leaching from fields. It has been observed a big difference in leaching based on soil type the bioresidual is spread on (Bernstad & Jansen 2011). Sandy soil has leaching coefficients of 25% to surface waters and 45% to ground water while loamy soil has 0% and 22% of sandy soil and loamy soil respectively for bioresidual.

2.6. Main treatment technologies

When considering waste treatment is there several options, both alternatives and technological options, for each of the technologies. Some of these technologies has been studied closer in this chapter to identify differences in emission, resource and energy requirements.

Pre-treatment

Pasteurization or other types of disinfection are required to disinfect the slurry that goes out of the anaerobic digester. This process is however mainly applied before the anaerobic treatment to ensure that the biogas producing bacteria are not deceased or disturbed in their processing. In addition does the feedstock need to be heated to the optimal temperature before reaching the digester and thus a hygienization in the pre-treatment is the preferable approach to maximize the heat recovery (Ariunbaatar et al. 2014; Jørgensen 2015; Fløan 2015).

Pasteurization can be accomplished at many different temperatures and thus the HRT varies. The main alternatives are 52C/ 10 h, 53.5C/8h, 55C/6h and 70C/1h (Ariunbaatar et al. 2014). Cambi has developed another technology that uses 150C/20min and they claim that the methane yield and degrability increases for the feedstock, this is in particular good for cellulose containing materials such as grass, garden wastes and horse manure (Fløan 2015). In Ariunbaatar et al. (2014)'s study has they indentified some benefits for several pretreatment tetchnologies. They indentified, in their literature study, that 70C/2 h and 150C/1 h pretreatment

[3] Impact such as marine and freshwater eutrophication which mainly affect this compartment has been excluded in this study due to high levels of uncertainty. N and P leaching are however, implemented in the LCA model in SimaPro 8 for later studies and development.

temperatures resulted in mehane increases of 2.69% and 11.9%, respectiviely. This comparison is done for mesophilic continous flow treatmet and is relative to no pretreatment applied.

2.7. Anaerobic digestion

Anaerobic digestion can be applied in several ways that highly affect the outcome of the treatment process (Raadal & Morken 2008; Søheim et al. 2010). There is several technologies of achieving anaerobic digestion, but only three most common digesting methods is mentioned in this chapter.

One stage batch

One stage batch is a process in which a liquefied slurry is fed into a digester, and then the substrate degrades. The digester is then emptied before next batch of organic slurry is to undergo anaerobic digestion. This is known as a one-stage treatment because all processing and degradation occur at the same time in the digester. This treatment option is simple to use, but provide lower biogas yield and is most commonly applied in small-scale farming (Luostarinen et al. 2011).

One stage continuous flow

This method applies the same simple one stage digestion treatment, but instead of emptying and refilling is the substrate continuously filled in, as some of the digested substrate is extracted from the digester. Such a treatment reduces the energy consumption by having a constantly heated main body of slurry in the digester. In addition, does the bacteria culture get a much more diverse nutrient mix and not necessary to start a new bacteria culture each time new substrate is added to the digester (Luostarinen et al. 2011).

CSTR (multi- stage continuous stirred flow)

This is the most common anaerobic treatment technology at current date for big scale treatment plants. The process is similar to the one stage continuous flow, as it treats new substrate as continuously emptied for old (Luostarinen et al. 2011). The difference is that this treatment option digests the substrate in separate stages and that the pasteurization is separated from the digestion. To convert long hydrocarbons into CH_4 is several bacteria species needed. The pre-treatment processes can be size reduction of organic matter, hygienization, pre-separation, sonication, enzyme addition among others[4] (Luostarinen et al. 2011). After the pre-treatment is

[4] For a more detailed description of pre-treatment technologies is Luostarinen et al. 2011 recommended reading, pg. 18.

the substrate filled in the digester where continuously stirred to maximise the contact between the volatile solids and the bacteria to enhance the biogas production even more.

The last stage in a multi stage digester is the post treatment process. possible to recover a significant amount of the biogas emitted from a biogas plant by providing a cover or collection system. 10 – 30% of the biogas potential remain in the bioresidual, which in some cases can be further collected (Luostarinen et al. 2011). If not, will the bioresidual be emitted large amounts of methane to the atmosphere and thus contribute to GHG emissions as earlier described, (Luostarinen et al. 2011; Amon et al. 2006; Khalid et al. 2011; Muha et al. 2014). also possible to do a mechanical separation of the bioresidual where the wet and dry separation fractions are separated[5].

2.8. Incineration

Incineration is the preferred solution for waste treatment in Norway at current date. In many cases is incineration applied for environmental reasons as incineration drastically reduces the need for landfill areas and thus the methane to air generation (Beylot & Villeneuve 2013). Landfill often leads to leaching to groundwater and nearby rivers of organic compounds resulting in chemical oxidant formation, eutrophication etc. also an economical solution as some of the energy can be harvested from the combustion as both electricity and heat, where the latter is the most common appliance in Norway (Jørgensen 2015). By burning the organic material by itself it avoids sources for contamination and can therefore be used as filling in road construction or concreate production if the contaminations meets the quality requirements. However only the bottom ash can be utilized for such a purpose. The incineration process creates two types of ash during combustion, fly ash and bottom ash. The fly ash is treated trough the off gas treatment system and can contain high levels of heavy metals and other cardiogenic compounds. Therefore, is the fly ash not suitable as a fertilizer and should be treated as special waste and landfilled in closed storages. The bottom ash leaves the heavier compounds that does not levitate during combustion and must therefore go through a sorting and later for heavy metal treatment and cleaning. therefore only “pure” ash with quality zero that can be used as a fertilizer. This can be achieved by restricting the inputs to only organic wastes where sewage sludge is considered a source of contamination and should therefore be limited in amount, (Boesch et al. 2014). There is several incineration types that are commercially used today. The two most common plant types is fluid gas bed and incineration moving grate (Jørgensen 2015).

[5] For a more detailed description of the post-treatment technologies is Luostarinen et al. 2011 recommended reading, pg. 20.

Incineration with moving grate secure a good access for air both over and under the solid waste being incinerated. The moving grate moves the waste forward to make space for the new waste, which is continuously moved into the combustion chamber. The heat produced can be used either to produce only remote heat or both heat and electricity, a combined heat and power plant (CHP) (Yingjiang et al. 2014; Hamelin, 2014).

Circulating fluidized bed (CFB) incineration rely on gasification of the waste and direct incineration in a plasma state (Li et al. 2014). using a direct – indirect combined drying where the substrate is pumped into a chamber filled with gas, ash and burning substrates. The water in the input substrate evaporates and the dry matter combusts. This technique is proven to be yielding a high energy efficiency and low emissions (Li et al. 2014) while reliable and safe to operate. The main concussion in the paper Li et al. 2014 is that the CFB can handle high water content materials and still has an combustion efficiency of 98%. This makes it energy efficient and cost effective. The overall efficiency is 74.91% (Li et al. 2014).

2.9. Biogas upgrade technologies

Biogas upgrade is a methane purification process of the biogas that increases the utility of the gas to include fuel and grid usage (Raadal & Morken 2008; Poeschl et al. 2012a; Visvanathan 2014). Biomethane is created when the average amount of CH^4 exceeds 97% and when H_2S, SO2, H_2O *vapour* and other corrosive gases has been removed from the gas,(Raadal & Morken 2008). To achieve this purified state of the biogas are there five main technologies that is common to apply today. These are water scrubbing (WS), Chemical scrubbing, Pressure swing absorption (PSA), membrane capture technique (MC) and cryogenic cleaning (CC), where the most commercial applied technology in Norway is water scrubbing, but with an increase in chemical scrubbers the last few years (Raadal & Morken 2008; Hung & Solli 2012; Bauer et al. 2013)

Water scrubbing is the most commercially used technology is based on the solubility difference for CH_4 and CO_2 in water. The high solubility for water towards CO_2 makes it possible to clean the biogas by leading the gas through a tank or tube filled with streaming water with a temperature of from 8 to 14 ℃ (Bauer et al. 2013). The water has to be flowing to secure the water to be able to absorb as much CO_2 as possible, which is accomplished at highest rate close to equilibrium. The temperature also affects the absorption rate and capacity where lower water temperature increases the solubility (Bauer et al. 2013). The solubility is a measurement of uptake of one compound to another and gives the amount of the absorbed chemical based on the concentration in the solvent. The methane losses in this process is assumed to be around 1.2 – 4.2 % (Luostarinen et al. 2011; Raadal & Morken 2008; Bauer et al. 2013).

Amine scrubbing or chemical scrubbing uses most commonly methyldiethanolamine, dietthnolamine (DEA) or monoethanolamine (MEA) to enhance the absorption capacity and

rate of a water scrubbing system. Because these chemicals introduce a weak basic solution, the solubility of the solvent is greater towards CO_2 than that of clean water and thus more efficient. from the producer side guaranteed that the maximum consumption of MEA shall not exceed 0,000003 kg/ m^3 raw biogas (Bauer et al. 2013). in this project only assumed that MEA is the only additive and the water consumption will be equal to that of water scrubbing (Bauer et al. 2013). The CO_2 removal capacity from the biogas assumed to be 99.8%, which is by far the highest absorption gradient of all the technologies. The biogas loss is thereby the lowest of the upgrading options (0,1%) (Raadal & Morken 2008; Bauer et al. 2013; Luostarinen et al. 2011).

Pressure Swing Absorption is a technology that uses the solubility between a solid compound toward a high-pressurized flow of biogas to react with the CO_2 and H_2S. Because the high pressure increases the solubility, the CO_2 will be stuck in the bed floor material. When the bed floor is fully saturated, the vent will be closed and the pressure in the chamber with the bound up CO_2 will decrease rapidly. This causes the CO_2 be leached fully from the solid compound which is able to do the process over again (Bauer et al. 2013). This technology is the second most used in Norway and it has a CO_2 removal >98% and a methane loss of approximately 2% (Raadal & Morken 2008; Bauer et al. 2013; Luostarinen et al. 2011).

Membrane separation uses a membrane that enables different gasses to escape at different stages in the membrane. This enables a CO_2 leaching >98%. common to divide the process into three steps where there first step is to dry the gas for water and then filter or absorbed out H_2S (Bauer et al. 2013). Even though the membrane is designed to let methane leak trough while restricting the access of others does approximately 3% of the methane escape to the of gasses and will be released to the atmosphere (Bauer et al. 2013; Raadal & Morken 2008).

Cryogenic separation is a developing and promising technology that uses the thermodynamic capabilities and states of different gases to separate the biogas. This is possible because CH_4 condenses at -161℃which is much lower than most gases. However, this technology as with the others has no possibility to clean out N_2 because a chemically very stable molecule and the condensation point is at -196℃which is much lower than the temperature operated in this process. The condensation point of CO_2 is at - 78℃ which means that the cryogenic process should has temperatures lower than this level, but an increase in atmospheric pressure reduces the need for cooling to -50℃. The CO_2 and H_2S cleaning ratio is 100% which makes N_2 to be the only energy-limiting factor in the remaining biomethane. This technology is however energy consuming and is not used for grand scale upgrading plants. Cryogenic cleaning technology is both the most energy intensive 0,45 kWh/ m^3 biogas treated (Hung & Solli 2012) and with the highest methane losses of approximately 5% (Raadal & Morken 2008; Bauer et al. 2013) but at the same time produces the most pure biomethane (99,8%). Another commodity that cryogenic cleaning can produce is industrial CO_2, which is an energy consuming process when not made from cryogenic biogas purification, Ecoinvent 3.0. In the cryogenic purification

technology can 25% of the CO_2 within the biogas be converted into industrial levels (Bauer et al. 2013; Andersson 2009).

Table 5: Inventory parameters for several biogas upgrading technologies

Technologies biogas upgrade	Electricity kWh / m3 treated	Heat MJ/ m3 treated	Water kg/ m3 treated	monoethanolamine kg/ m3 treated
Water scrubbing, Normal	0.23		0.004	
Chemical scrubber	0.13	1.96	0.003	0.0010584
Pressure swing absorption	0.25			
Membrane technology	0.3			
Cryogenic methods	0.4564			

Source: Data extracted from Bauer et al. (2013).

2.10. Gas storage and application

After the gas have been purified to >97% CH_4 is it necessary to compress the gas (CBG) to 200 or 300 bar or condense the biomethane (LBG) to further increase the increase the energy density so applicable as a biofuel (Hung & Solli 2011; Bauer et al. 2013; Møyland 2012; KLIF 2013). This requires energy in form of electricity to the compressor, which is approximately 0.21 and 0.25 kWh per m^3 raw biogas. In the case where biomethane is injected into a gas grid network is the common pressure 45 to 50 bar, which requires approximately 0.16 kWh per m^3 raw biogas (Bauer et al. 2013).

To liquefy the biogas yields a much higher energy density as one m^3 biomethane is condensed into 1.7 l of space, which is more than the double volume metric energy density of CBG at 250 bar (Bauer et al. 2013). This increases the amount of energy possible to store in a vehicle and reduces the mass of the storage tank in the vehicle and thus the energy consumption per vkm compared to CBG alternatives (Hung & Solli 2011; KLIF 2013). The liquidation occur in a cooling process at - 163℃ at a pressure at 1.5 bar and is stored in cooled tanks at 4 – 5 bar (Bauer et al. 2013; KLIF 2013). The cleaning process for LBG leads to a further 1.8% methane leakage for the biomethane. This is caused by the additional purification of the biomethane to satisfy purity requirement of 50 – 125 ppm CO_2, The concentration of H_2O and H_2O cannot exceed 0.5 ppm and 3.5ppm respectively to avoid plugging and freezing problems (Bauer et al. 2013).

The CBG or LBG can be used to substitute several fossil commodities such as natural gas, gasoline or diesel in common cars, busses, lorry trucks or natural gas network system (Hung & Solli 2011; Raadal & Morken 2008; HOG Energi 2010). For busses is the biomethane consumption estimated to be an average of 0.5367 Nm3 per vkm which is approximately 15.18

MJ per vkm (Hung & Solli 2011). The average diesel consumption is on the other hand is 0.45 l per vkm, which gives an energy consumption of 0.45 l/vkm[6] * 35.8 MJ / l[7] = 14.48 MJ / vkm, which gives that the average energy consumption is higher for biogas busses than for diesel busses (Toutain et al. 2008).

Fuel substitution using biomethane has become increasingly popular in Norway and cities such as Trondheim, Fredrikstad, Oslo and Stavanger has already many busses that utilises biomethane in a mix with natural gas or solely on biomethane (Simonsen 2012). This has created a demand for biomethane and the production has increased and is still increasing today including other sectors such municipal vehicles and municipal waste collection system (Jørgensen 2015; Fløan 2015).

Biogas for energy in a Norwegian context can substitute either one of the two commodities remote heat or electricity. Remote heat is either produced by waste incineration or by natural gas and light fuel oil whereas the electricity consist of 96.7% hydro power and 1% wind and 2.3% from combustion power plants (SSB 2014).

2.11. Bioresidual end treatment

The remaining mass after digestion (bioresidual) consist of remaining non-volatile solids, undigested organic compounds and water (>90) (Carlsson & Uldal 2009; Lyng et al. 2011). The bioresidual has to be stored and applied dependent of the quality level. common in Norway, at current date, to dewater the bioresidual and then only apply the dewatered fraction (30% solids) as biofertilizer whereas the watery fraction is sent to WWT (Jørgensen 2015; Fløan 2015). This limits the amount of N applied to the field as (70%) resides in the watery body of the bioresidual (Möller & Müller 2012). Other treatment methods is to use both the separated bioresidual compartmetns or by not treating it. This has much impact on total release of NH_3, N_2O and CH_4 (Amon et al. 2006; Bernstad & Jansen 2011) as described in chapter 2.5 *Emission to air*.

2.12. Norwegian empirical studies

As mentioned in the introduction has an empirical survey been undertaken. Two highly different plants has been visited, Fervar in Fredrikstad and Ecopro in Verdal and both plants agreed to contribute to this thesis with both data and experiences.

The situation

Biogas production in Norway is yet a relatively small industry, but growing due to the increased attention biogas has achieved this last decade. The science on biogas production and organic waste treatment has mainly been explored in detail the last seven years, with a few exceptions.

[6] Data found in (Hung & Solli 2011)
[7] Data found in (Hofstad 2014)

Plants such as Frevar started with anaerobic waste treatment and biogas production as early as in the 1990's and had therefore limited with scientific knowledge to back the development. This is also the case for other biogas plants in Norway, which might to some extent explain the great variety in technological composition of the relatively few plants in Norway. Another explanation, and maybe the most important, is that organic waste handling itself a paid service. This limits the requirement for profit in sale of the end-products which is an expensive investment (Jørgensen 2015; Fløan 2015). All the big plants in Norway has a cost range around 110 to 200 million NOK and the sale of the end-products must be in demand if such an investment is feasible. Therefore will business strategies affect the development of anaerobic digestion and a step by step development of biogas plants has been one of these strategies.

Organic waste is, in Norway as in the EU, a treaded commodity. This makes the waste in some instants to travel great distances because there is no necessity that handed at the closest plant. This makes the feedstock of the plant to vary beyond the local waste conditions and even the national waste mix. Sewage sludge is maybe the only organic waste that can represented the local conditions, but in some instants is the organic fraction transported to nearby treatment plants. Sewage sludge is often treated separately because of the properties it has, especially the high water content and regulation restrictions for further application of bioresidual (Jørgensen 2015).

3. Methodology

In this chapter follows an introduction to LCA to establish important methodology steps required for the following analysis. Due to the vast diversity of LCA methods, has only the most relevant been included[8].

3.1. LCA

Lifecycle assessment (LCA) is a well-developed quantitative method for identifying environmental impacts in form of environmental stressors caused by a given measurable activity, the functional unit (International Organization for Standardization 2006; Møyland 2012; Baumann & Tillman 2004a; Clavreul et al. 2012; Benoît & Mazijn 2009). This un special for LCA and used to specify the product or service that is assessed and the given quantity of that unit (Benoît & Mazijn 2009). When using stressors is a wider definition of emissions applied because emissions is widely associated with release of gasses and toxins, while stressors also include land use, radiation, ozone depletion etc. (Majeau-Betez & Strømman 2009). As previously mentioned is there are a huge variety of stressors and therefore important to put them in a given impact category, if not they will just be quantification of each individual stressor with a low relevance for the overall impact potential they causes. The procedure method of LCA is defined into the four steps; goal and scope definition, lifecycle inventory, lifecycle impact assessment (LCIA) and interpretation of all of these stages.

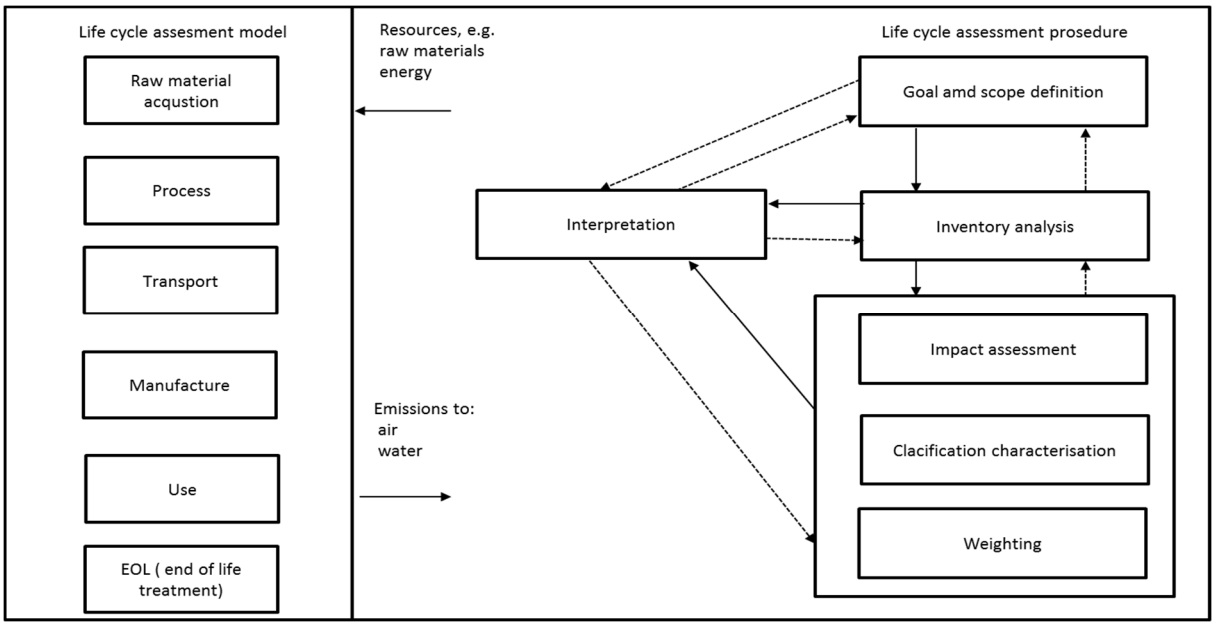

Figure 2: LCA procedure and stages: (Baumann & Tillman 2004a) *pg. 20.*

[8] ISO 14040 2006 and Bauman & Tillman 2004 is recommended reading for a further analysis of the LCA methodology.

Goal and scope define the boundaries of, the purpose and what is to be the interesting subject or object to study. The ISO 14040 describes that the goal and scope definition should clarify what the intended application of the study is, whom the results should be presented for, the purpose and reason of the study and how the results are intended communicated (Baumann & Tillman 2004a). This means that the goal and scope has to include important topics such as the context for the study, the planning of the project and a specification of the functional unit to be used in the LCA model. important to explain why the specific functional un selected and how this will affect the results found in the study.

The functional un in the ISO 14040 defined as "quantified performance of a product system for use as a reference unit" therefore critical to be certain that the functional un of relevance and within the goal and scope of the project. This is why the goal and scope is of great relevance in every LCA survey performed. The fact that any life cycle assessment is of low relevance without any relation to a goal definition including a purpose can be witnessed in the interpretation part of the assessment. If the researcher has loads of data but no knowledge of what they resemble the researcher might end up with not understanding the findings, (Baumann & Tillman 2004a).

3.2. Lifecycle inventory description

When the goal and scope of an LCA project is defined possible to start gathering data about and eventually create an inventory list of the system. A flowchart of the system should be made to reduce the possibility for double counting and all the processes should be explained. The inventory list is the quantified requirements relative to the functional unit for all of the feedstock, processes, transport, manufacturing, use and end of life treatment, figure 2. The data gathered should be referred to and uncertain parameters or values should be tested in a sensitivity analysis and a description of each process should be executed (Clavreul et al. 2012; ISO 14040 2006). The inventory LCI is not for comparison purposes, but as a multiplier for the stressors cause per unit of each inventory unit (ISO 14040 2006). Direct emissions are in the LCI referred to as requirements to fulfil the purpose of the FU, which might be understood as an odd approach but it highlights that the process requires to relive itself of emissions per FU accomplished (Baumann & Tillman 2004a). The result of LCI is quantified into sum of each stressor and leads to the foundation for the next step in LCA, life cycle impact assessment with impact characterisation, damage assessment and weighting.

3.3. Lifecycle impact assessment

LCIAs differentiates from the other methods, such as environmental performance evaluation, environmental impact assessment and risk assessment, because assessed relative to the FU. In the LCIA is the results from the LCI further interpret with respect to the total stress potentially

caused by the FU, and is represented by either midpoint or endpoint characterization, depending of the scope of the analysis. The characterisation can be applied with several scopes of time and a choice of understanding and the method of choice has to be established (Baumann & Tillman 2004a). The scopes are either Individualist (I), Hierarchist (H) and Egalitarian (E) where:

- I perspective: considers a short term (20 yr) impact and an optimism that humans will adapt and that technology can solve all problems (ISO 14044 2006; Goedkoop et al. 2009).
- H perspective: considers a middle term (100 yr) and is based on the most common policy principles (ISO 14044 2006; Goedkoop et al. 2009).
- E perspective: considers a long term (500 yr) and is precautionary with respect to time and that there is impact types that we not yet has discovered or developed (ISO 14044 2006; Goedkoop et al. 2009).

3.4. LCA aspects

The LCA methodology has been divided between *accounting LCA* and *change oriented LCA* methodology. *Accounting LCA* focus on the environmental impact caused by a given functional unit, most usually a product. This is relevant when there is a myriad of products and the goal is to find which impacts are caused by the product of interest. not of interest to change the way the functional un produced or achieved, but simply identifying the emissions and impacts caused by using the average technology for the given products. To allocate the emissions in the system leading to the functional unit, *accounting LCA* is prone to use partitioning. This means that the emissions is divided between the myriad of processes and products required to fulfil the functional unit. The benefit that the accounting LCA requires much less information of each individual process and product and is therefore an easier yet less accurate way to describe the emissions caused by the fulfilment of the functional unit. The obvious disadvantage with this methodology procedure is that there is no flexibility to change the production or use of the product. Therefore has the *change oriented LCA* been developed, and is in essence a much more comprehensive analysis, which divides the system into the subdivisions foreground and background, and the interactions between those. (Baumann & Tillman 2004a).

The *change oriented LCA* is used when analysing the means used to fulfil the functional unit and is often a part of a bigger system. The means is as in real life, possible to achieve by using different technologies and the impacts caused by the product is depending on both the means of achievement and the specific use or substitution possibility of the functional unit. For example, a great difference in the impacts caused by open mining versus tunnel mining, and whether you use the copper mined for outdoor roofing, ship hulls or for electrical wires. This means that the change oriented analysis can be applied for many different processes and life stages such as extraction of raw material, operation of a product, waste treatment etc.

LCA has a bottom up approach which means that it requires data which is at process level (Clavreul et al. 2012). By applying this, often necessary to aggregate several processes into a "black box" processes. The black box process is the sum of all the processes within, without the need of knowing the specific data about all the processes within and thus limits the data requirements.

Parameter and variable estimation

Parameters are coefficients or flows, which does not change when changing other parameters in the system (Baumann & Tillman 2004a). Variables on the other hand are flows and coefficients that change in effect as the parameters are changed. The parameters is established as mentioned in the literature study introduction, based on quantitative information found in the literature study or in case specific cases (Clavreul et al. 2012). A mix between quantitative case specific and literature data is often common as few case specific cases can supply all information required to be true to the completeness of the analysis (ISO 14044 2006).

3.5. Robustness of LCA results

In LCA there is a multitude of processes and parameters that is subject to assumption in particular for waste LCA (Clavreul et al. 2012; Baumann & Tillman 2004a). Therefore is it important to be sure that the data in the inventory is correct for the analysed system, also called a completeness check. also important that epistemic uncertainty (incomplete state of knowledge) are not amplified with inconsistency through the system definition, also called an consistency check. An analysis of stochastic uncertainty (natural variation in parameters) gives a quantifiable uncertainty of the data found and should therefore be based on several sources. In effect, the sources applied for the analysis is also reviewed and uncertainty connected to the reliability of the sources itself is established (Baumann & Tillman 2004a).

Sensitivity analysis is important when dealing with parameters and variables identify which that has a profound effect on the system and which does not (Clavreul et al. 2012). therefore necessary to perform a sensitivity analysis. The sensitivity is measured by applying a change, often one parameter at a time, with dummy variables and compare the results with a constant base case. As mentioned before is sensitivity particularly important in a waste LCA and a wider set of parameters should be tested for sensitivity and interpret in the uncertainty chapter.

Variation analysis or case/scenario analysis is an important part of any LCA to identify how the results respond to variation in product, utility and processing options. This is interesting for many reasons as it gives the overall results of various applications of the LCA model made in relation to the subject at hand. It also furthers the sensitivity analysis in the way that several parameters and thus variables that are changed at the same time, and the results is the combined

effect of these changes (Majeau-Betez & Strømman 2009; Clavreul et al. 2012; Baumann & Tillman 2004a).

3.6. Material flow analysis (MFA)

MFA is a quantifiable methodical tool that is used to trace the flow of elements (SFA), energy (EFA) or materials (MFA) through a system (Squires 1984). defined for a chosen area or system and for a given time. MFA methodology can be seen as four steps, the process, stock or rate of change, flow or flux-rate and system boundary with respect to space and time (Ubstance 2004). A process can be defined as a compartment or occurring transition that is clearly separated from the next process (Ubstance 2004). A flow is a stream of material from one process to another. MFA takes into account the flows in and out, the stock of that material. The stock is either zero or positive but the flow of material from or to the stock can be positive or negative, determining whether the stock increases or decreases. $\Delta Stock = \sum(Xin - Xout)$ and $Stock = Si + \Delta Stock$ where Si is the initial stock and Δstock is the change in stock due to differences in inflow and outflow of the given material (Ubstance 2004). The driver of the system can be many, but it can be narrowed down to feedstock (inflow), stock or output (delayed inflow) driven models. In a stock driven model is the stock the focus of attention and even though the demand is external is it the total stock that determine the need for input. In a waste scenario is it the generation of waste that determine the input of waste and the feedstock driven model is in effect (Clavreul et al. 2012). For systems, that does not has any stocks, but where the demand of the product is the driver of the system is it the outflow that drives the system (Ubstance 2004).

MFA modeling types

The four MFA modeling types are static, stationary, quasi-stationary and dynamic models. All these models is defined with their relationship to time, which for static models is irrelevant because the result will be the same if the results are presented today or in ten years for the same system. The purpose of a static model is that time is excluded from the equation and that $X(t) = X(-t)$ and thus gives a simplified mass balance approach without the confusion often associated with time.

4. LCI

In this chapter, the data collection, inventory estimation and list, calculation setup and method of choice is described. The goal and scope narrows the study to account for exclusively big scale plants, and does therefore not include small family or farm scale plants. The choice of methodology and method of data collection is determined by the given scope. The assumptions made, are based on quantitative data limited to this scale. Inventory analysis or Life cycle inventory analysis (LCI) is the building block of any LCA and is thus the most important part of the procedure (Baumann & Tillman 2004a; ISO 14044 2006). In this study, there are three main inventory lists: Energy parameters[9], material flow transfer coefficients [10], and emission intensities.

4.1. Choice of methodology

In this study the change-oriented methodology has been chosen, exactly because the purpose is to identify attributional changes of different organic waste treatment choices. It is therefore within the system boundary to analyse a part (organic waste substrates) of a bigger system and focus on the relative difference between several treatment, technological and feedstock options. The hierarchist assessment method has been applied, because the purpose is to assess the changes of impacts generated based on basic policies. No further assessment than midpoint levels are included in the LCIA step to avoid unnecessary uncertainty due to endpoint characterisations. Substitution potential for the end-products has proven to be of high relevance for organic waste treatment studies and system expansion has thus been focused upon in this study. The system expansion methodology has been chosen to determine the attributional environmental effects of produced products in each case and as a comparison foundation for each case evaluated. All flows of the system are calculated by applying the static MFA approach as time is of no relevance in this study. Only HM is assumed at point of measurement, due to no relevant HM data found in the literature. The content of HM levels are determined by the classification levels for bioresidual presented in table 3.

[9] Listed in Appendix 21, parameter 256 - 277

[10] Determined by the technology (Appendix 21 parameter 66 – 77) and treatment method applied. All of these are listed in Appendix 21 parameter 3 – 5, 12 – 14, 18 – 21, 224 – 227 and 292 – 335.

4.2. Data gathering procedure and critical assumptions

In this chapter is the procedure of data collection evaluated. It is to highlight sources of uncertainty and how critical data where acquired.

Data collection

The data collected for this study has been aiming towards specific technical data. The main targeted data is of the technical data classification. This enables the collected information to be utilized as transfer coefficients for the MFA, as recommended by Clavreul et al. (2012); Muha et al. (2014). These includes organic waste substrate mass, energy and nutrient flows. However, some supplementary generic data such as N loss for several substrate types have been necessary. These are included in the parameter sheet Appendix 21 parameters 300 – 302. The NH_3, N_2O and N_2 ratio for the lost nitrogen earlier presented in table 4 are to be found in Appendix 21 parameter 298 and 299. In addition have assumptions based on previous case specific studies been made. In particular is the relative change in CH_4, NH_3 and N_2O emitted calculated in such a manner, based on the findings in Amon et al. (2006), Appendix12 – 14. The calculation is further described in chapter 4.7. The estimated parameters for the relative change in CH_4, NH_3 and N_2O has been implemented in the SimaPro 8 model parameter list, Appendix 16 parameter 303 - 332. This particular assumption is subject to a high level of uncertainty as previously described in *chapter 2.1* and is therefore further assessed in the uncertainty chapter, *chapter 6.3.*

The definition of the value chain for treatment alternatives for organic waste treatment are to be found in Appendix 2. This data are based the two Norwegian state of the art studies (Lyng et al. 2011; Modahl et al. 2014). In addition have empirical data acquired at the two biogas plant visits been applied as supplementary data.

All parameters applied in this LCA study have been listed in Appendix 21. This list also contain option for carious technological applications. The variable calculation is presented in Appendix 22. Important variables and stressor calculations are assessed later in this chapter, chapter 4.5 to 4.8.

Important papers

To acquire technical data, some studies have proven to be more relevant than other. Only sources that have been referred to in several relevant studies or studies that have new data and which have referred to some of the older state of the art, are included in this study. These sources are in particular Amon et al. (2006); Morken et al. (2007); Alvarez & Lidén (2008); Raadal & Morken (2008); Carlsson & Uldal (2009); Lyng et al. 2011; Rehl & Müller (2011); Bernstad & Jansen (2011); Bauer et al. (2013); Modahl et al. (2014); Muha et al. (2014).

For a Norwegian context, have the two studies Lyng et al. 2011 and Modahl et al. 2014 been given a particular position as comparison studies for the results presented in this study in Chapter 5. Several less famous studies have been applied as a mean to validate and contribute to unknown data. This one of several approaches applied, which purpose is to identify stochastic uncertainty and to gain comparison or supplementing data for epistemic uncertainty assessment. Such an approach should, according to Clavreul et al. 2012 and Baumann & Tillman 2004, revile bot the previously mentioned stochastic and epistemic variations. Care should be applied however, as multiple sources also contributes as a major source of uncertainty by itself as previously described in Chapter 2.1.

For qualitative data of MFA and LCA methodology, the same procedure has been applied. Throughout the methodology chapter, there only a handful of previous studies and methodological books on LCA methodology, which have been applied as sources on methodology. These are Baumann & Tillman 2004b; Ubstance 2004; ISO 14040 (2006); ISO 14044 (2006); Benoît & Mazijn 2009; Goedkoop et al. 2009; Clavreul et al. 2012). In addition, methods described in previous state of the art LCA and MFA studies have been applied.

4.3. Model development

To performed an LCA on organic waste management particularly biogas production, it was necessary to perform an value chain assessment. In Appendix 6 the resulting value chain have been illustrated. This figure have aimed upon presenting important flows, processes and system expansion for likely product substitutions.

A simplification of the system was deemed necessary to achieve presentable results. The resulting system, presented in figure 3, have been applied to quickly illustrate the main processes. This simplified version contains the may processes and flows, that is presented in Appendix 2, but at a sub level. This means that the modelling and data requirement is determined at parameter level while the results are presented differently. In the SimaPro 8, several of the processes are predefined by Ecoinvent 3.0. Some examples are Waste Water Treatment (process 5), Water treatment (process 7), transport (T1, T2, T3 and T4), energy carriers (electricity NO, natural gas heat, diesel etc.), dewatering (process 10) land fill (process 4) and the substitution of bus by diesel substitution. For other processes have a separate inventory had to be developed as for Biogas upgrade (previously presented in table 5) and the post treatment bioresidual. The latter process, have been based on information from (Amon et al. 2006).

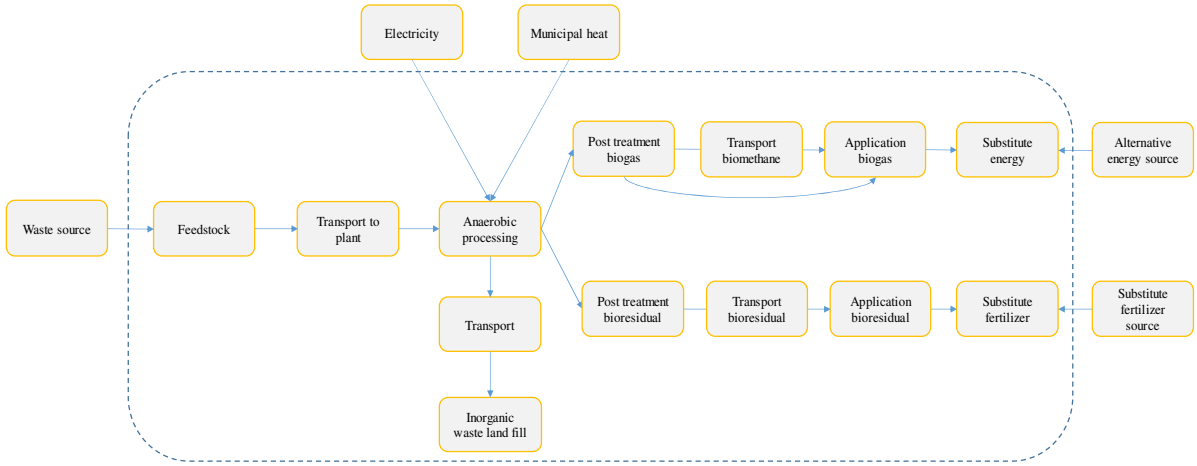

Figure 3 Simplified value chain of anaerobic treatment

One such set of substrates is the initial feedstock presented in table 6. The consequences of this aggregation make it impossible to extract LCA data specific for each individual substrate, without performing homogenous digestion. However, the interesting part for this thesis is to identify underlying parameters and variables. By performing contribution analysis should this be possible, without the mentioned substrate contribution.

Table 6: Organic waste analysed in the Norwegian base cases

Sectors	Sector share	Substrate within each sector	Substrate share in sector	DMC (%)	Sourcess
Manure	50.0%	Cattle manure	70%	8%	Carlsson & Uldal 2009.
		Pig manure	30%	8%	Carlsson & Uldal 2009.
		Sum Manure	100%		
Fat	0.0%	Frying fat	100%	90%	Carlsson & Uldal 2009.
		sum			
Organic municipal Wase (OMW)	13.1%	Sorted houshold food waste	100%	33%	Carlsson & Uldal 2009.
		sum OMW	100%		
Organic Industrial Waste (OIW)	32.1%	Separated animal fats	0%	4%	Carlsson & Uldal 2009.
		Fish processing waste (offal)	49%	42%	Carlsson & Uldal 2009, Genauer & Eikebrokk 2005
		Sorted resturant food waste	15%	27%	Carlsson & Uldal 2009.
		Slaugtherhouse waste (blood)	0%	10%	Carlsson & Uldal 2009.
		Slaugtherhouse waste (blood)	0%	30%	Assumed for this study
		Slaugtherhouse waste (offal)	24%	16%	Carlsson & Uldal 2009.
		Diary processing	12%	20%	Carlsson & Uldal 2009.
		Fruit and vegetable waste	0%	15%	Carlsson & Uldal 2009.
		Sum IOW	100%		
Sewage sludge (SwSl)	4.8%	Sewagesludge	100%	10%	Wadahl. S 2015
Substrate X	0.0%	Sum multipe inputs	100%	50%	

The sector share, presented above, have been developed and adjusted from the average waste mix derived from Morken et al. 2008.

The SimaPro 8 model consist of two main flows, mass and energy, illustrated in Appendix 6. The mass is adjusted by a series of parameters such as total mass inn, required water to or from pre-treatment, separation of organic and inorganic waste to plant, as well as the production of biogas and bioresidual, including gaseous losses. In the LCA is a separation of these, particularly the gaseous emissions presented.

To maintain a mass balance consistent MFA is, as mentioned in the LCA chapter, not a prerequisite in common LCA, but has proven to be an important source of data in this report while it also limits the sources of uncertainty (Clavreul et al. 2012). However only the most important flows have been balanced in the SimaPro 8 model. Parts of the organic waste treatment are very well documented and analysed, whereas other parts are not even mentioned in the average analysis. To identify these missing flows has a simplified MFA proven to be the best available tool to acquire these without using assumptions. Only some transfer coefficients have to be determined, which in later study makes it easier to reassess the study and validate or disprove the results. MFA has been a handy tool in identifying contradictory parameters found in the literature such as the volatile solid degradation and remaining volatile solids.

It has also been important to trace the flow of N and P through the system to identify the actual nutrient substitution potential of the bioresidual. Further explanation of the equation development and estimate procedure are presented in Chapter *4.5 to 4.8.*

The energy flow is based on the HHV of each individual substrate and the energy density of methane. To acquire these parameters where it necessary to make use of several assumptions and studies. This highly decreases the certainty for the incineration heat estimate and the total energy contend for the feedstock energy. Assumed energy contents for each substrate is included in the parameter list Appendix 21, parameter 195 – 207.

The resulting model, a change oriented LCA model that is built upon the most basic static MFA modelling principles with respect to the mass balance principle for individual processes and stressors. However, stocks has not been included in this study as under the assumption that the flows are equal on both sides of every process and the time range is set for one lifecycle.

4.4. Feedstock estimation

The feedstock is the main driver for the whole system and determines a wide range of all the parameters in the system. The most important parameters are the water content, N and P composition, methane yield, degrability and heavy metal contamination level. This makes it important to assess the feedstock and determine these qualities and the current combination of these substrates. In this model is a sector-aggregated approach applied. The sectors chosen are Manure, Fat, Organic municipal waste (OMW), Organic industrial waste (OIW) and sewage sludge (SwSl), which is composed of several sub waste categories, table 3. The subcategorization has been applied to secure that the variation in N, P, degrability and theoretical methane potential. Particularly substrates for industrial waste have been included, because of the high level of detail the literature present. The initial mass of the organic waste analysed is determined by:

Equation 1: Total mass of waste analysed in the system determined by the DM (FU)

$$DM = \sum_{j} DM_{j} = FU = 1\ ton$$

$$mOW = \sum_{j} \left(\frac{DM_{i,j} * sOW_{i,j} * sOW_{j}}{DMc_{i,\mathrm{j}}} \right)$$

The dry matter (DM) is the sum of the DM for each waste category(j), which for this study is one ton. The total mass of organic waste (mOW) is the sum of the DM for each sub waste type (i) that is classified under waste category with respect to the share of the sub waste ($sOW_{i,j}$) within the waste category share (sOW_j) and divided by the dry matter content for each sub waste ($DMc_{i,j}$).

4.5. Transport assessment

It is important for an LCA to include the processes that the product, system or service require (Baumann & Tillman 2004a). Transport associated to the accomplishment of the FU are, in some studies, (Hung & Solli 2012), given particular attention because of its overall effect on LCA results.

To be able to distinguish between the transportation requirement in this study, per FU, four processes been developed, respective to the product of transport. These are *Transport of organic waste (T1), Transport of biofuel (T2), Transport of bioresidual (T3) and Transport of ash, Appendix 2.* The inventory for these are presented in table 7. The transport processes are further disaggregated to be able to account for distance and mass for each individual organic waste substrate sector. This gives two advantages when aggregating and analysing the system. (1) The system can be applied various transport types within a given aggregated sector. (2) There is no need to apply allocation methodology to identify the most important organic waste category, in a transport context. Aggregating the processes at a later stage is beneficial, because highly detailed impact assessment can lead to more confusion than any fruitful information.

To understand the transport as a term of both mass and distance is it given the unit *ton kilometre* (tkm), which is a product of mass in (t) multiplied by the distance in km (Baumann & Tillman 2004a). In a generic model that is flexible to account for different utilities and processing technologies, changes the impact of transport according to the total required transport in terms of tkm, table 7.

Table 7: Overall view of transport inventory and how end-product choice affects the GWP performance based on changes in tkm

	Inventory				Dry bioresidual				Wet bioresidual / incinerated manure			
Transport	km	ton	tkm	kg CO2 eq/ km	Biogas	Biomethan	LBG	Incineratio	Biogas	Biomethan	LBG	Incineration
Manure	50	5.76	288.0	0.112	x	x	x		x	x	x	x
Fat	50	0	0.0	0.112	x	x	x	x	x	x	x	x
OMW	19	0.73	13.9	1.25	x	x	x	x	x	x	x	x
OIW	50	1.54	76.9	0.112	x	x	x	x	x	x	x	x
SwSl	0	0.282	0.0	0.112	x	x	x	x	x	x	x	x
Sum feedstock (tkm)	169.0	8.3	378.8		379	379	379	91	379	379	379	379
Sum feedstock kg CO2 eq/ FU					58.21	58.21	58.21	25.95	58.21	58.21	58.21	58.21
Bioresidual dry	50	0.6	28.2	0.112	x	x	x					
Bioresidual wet/separated	50	9.4	467.5	0.112					x	x	x	
Biogas	0	0.35	0.0	0.112	x				x			
Biomethane	10	2.24	22.4	0.112		x				x		
LBG	10	0.16	1.6	0.112			x				x	
Fly ash	250	0.1	12.6	0.112				x				x
Bottom ash	1	0.5	0.5	0.112				x				x
Sum post digested (tkm)					28	51	30	13	468	490	469	13
Sum post digested kg CO2 eq/ FU					3.15	5.66	3.33	1.46	52.36	54.87	52.54	1.46
Sum total tkm					407	429	409	104	846	869	848	392
Sum post digested kg CO2 eq/ FU					61.36	63.87	61.54	27.41	110.57	113.08	110.75	59.67

Comment: OMW is transported by municipal collection service, 21 ton and all other commodities are transported by freight 31 metric ton Euro5 standard (Ecoinvent 3.0)

According to Jørgensen (2015), biomethane is transported in pressurised gas tanks called a "Flak". One "Flak" weights about 11.5 ton empty and 12.5 tonnes full. This gives an additional weight of 11.5 ton per ton biomethane transported. This has been taken into account in the model. Added mass, based on the knowledge of tkm calculations, have major implications for the overall stress caused by the transport. As presented in table 7, is the impacts far higher than for LBG. However, transport GWP contribution is low compared to transport of organic waste or bioresidual.

4.6. End-product approach and general calculations

Previously in the literature study has the degrability of several substrates been acquired and the degrability estimated, fig 2. This is an important feature in this model because it will determine both the bioresidual and biogas produced in the treatment procedure.

Equation 2: Adjusted methane yield with respect to degrability

$$VS_i * VS_{Degrability,i} * MY_{VS,\,i} = MY_{Degradable,i}$$

The DM, as previously explained, consist of VS and ash weight $(1 - VS)$, where only the VS is possible to digest. The theoretical degradable part (D) of VS is represented as degradable VS ($VS_{Degrability}$). To identify the new methane yield ($MY_{Degradable}$) per DM, it is necessary to be in accordance with the initial methane yield of VS (MY_{VS}). The resulting parameters have been previously presented in table 2.

Equation 3: Basic methane equation

$$\sum_i \left(DM_{AD,i} * MY_{Degradable,i} \right) = CH_{4,total}$$

Equation 4: Methane equation with respect to increased biogas potential due to co-digestion

$$\sum_i \left(DM_{AD,i} * MY_{Degradable,i} \right) * Co-benefit = CH_{4,total}$$

Multiplying the sum of dry matter entering the AD (DM_{AD}) per substrate (i) with the sum of the adjusted methane yield is the adjusted theoretical methane yield found. In the MFA model has a co-benefit option been implemented and the formula is therefore changed, as shown in *Equation 3*. If an increase in degrability is assumed, the value of Co-benefit[11]is >1 while a decrease should be represented by <1, given in numeric present. Such an assumption will affect both the produced mass of biogas and the remaining mass of bioresidual, *Equation 7 and 8*.

Equation 5: Methane content in biogas when mixing organic substrates

$$S_{type,i,\%} * CH_{4,i,\%} = CH_{4,Bg,\%}$$

The biogas composition, according to table 2, varies for most substrates. The average often used in the literature is 65% (Morken et al. 2007). Assuming the average would not give the flexible and detailed picture this study are trying to accomplish. Therefore, it is important that the measured methane content of biogas for each substrate ($CH_{4,i\%}$) is relative to the substrate mix ($S_{typr,1\%}$) assessed. Such an approach generates thereby an average methane content ($CH_{4,Bg,\%}$).

Equation 6: Biogas calculated on basis of methane produced

$$\frac{CH_{4,total}}{CH_{4,\%Bg}} = Bg_{total}$$

The total biogas produced (Bg_{total}) can be found by dividing the total methane produced by the average share of methane found within the biogas. The average CH_4 is based on the share of each waste substrate, in DM, and the average composition of treated waste.

Equation 7: Total carbon dioxide produced in the AD

$$Bg_{total} * \left(1 - CH_{4,\%Bg}\right) = CO_{2,total}$$

It is important to find the total CO_2 produced as well. This can be accomplished by assuming that the remaining non methane part og Bg is CO_2. The mass of CO_2, as described later, is much higher than for CH_4, thusly important on a mass balance context, *Equation* 7.

[11] Found in Appendix 21, parameter 13.

Equation 8: The mass of the biogas

$$(CO_{2,total} * m_{CO_2} + CH_{4,total} * m_{CH_4}) = m_{biogas}$$

To identify the mass of biogas is it necessary to treat the independent gases separately. By applying the ideal gas law with respect to mass (m) per volume (V), where P is the pressure of the gas (Pascal), M is the Molar mass (g/mol), R is the universal gas constant and the temperature (K) in kelvin. Assumed that the gas is in Nm^3 the temperature can then be defined as Kelvin (K)= 273,15 K (0℃). This gives that the mass per Nm3 is define by:

$$\frac{m}{V} = \frac{(P*M)}{(R*T)}$$

The resulting mass per NM_3 is thereby given as:

$$m_{CH_4} = 714.3\,\text{g/ Nm}^3 = \frac{(1*16)}{(8.314*273.15)} \text{ and } m_{CO_2} = 1964.2\,\text{g/ Nm}^3 = \frac{(1*44)}{(8.314*273.15)}$$

Equation 9: Mass balance equation for bioresidual

$$m_{organic\ waste,wet} - m_{biogas} = Dg_{total}\ ^{(12)}$$

$$m_{organic\ waste,DM} - m_{biogas} = Dg_{DM}$$

When the mass of the biogas has been established can one find the output of the bioresidual as the remaining mass of the organic waste ($m_{organic\ waste,\ wet}$) that was fed into the digester. It is also assumed, that the only produced gases are CO_2 and CH_4. To find the remaining DM (Dg_{DM}), the same equation has been applied. However, here it is the DM organic waste in mass ($m_{organic\ waste,DM}$) that enters the equation.

Nutrient compositions

The composition of nutrients (N and P) in the organic waste varies a lot dependent on the substrate and the literature describing them. also hard to come by reliable data on this field and as such is the uncertainty relatively high. When considering co-digestion is it important to estimate the total nutrient mass with respect to both N and P found within the bioresidual that ca be applied as biofertilizer. To estimate the total N and P in the feedstock waste was the following equation constructed:

[12] Dg is the definition of digested material or bioresidual.

$$DM_i = FU * sOW_{i,j} * sOW_j$$

$$\sum_i \left(DM_i * N_i \right) = N_{in,total}$$

$$\sum_i \left(DM_i * P_i \right) = P_{in,total}$$

DM for each sub waste is determined by the quantity of the FU, in this study one ton, and the respective share of the sub waste relative to the waste category it falls under. P_i and N_i is the mass of P and N per ton DM for substrate (i), respectively and DM is the total dry matter of substrate (i) accounted for in the given case, Appendix 21 Nitrogen parameters nr. 162 – 175.

4.7. Emission approach and general calculations

The bioresidual is as described in the literature the solid leftover after the anaerobic digestion or composting. The emissions caused by this commodity is substantial and has been subject for several studies to identify these parameters. The results for these studies varies substantially, as described in the emission chapter, because there is a multitude of factors determining the emission rate. This increases the uncertainty of the study substantially and therefore important that a comprehensive sensitivity analysis are performed, (Clavreul et al. 2012).

Emissions to soil are found by measuring the content of heavy metals in the bioresidual after digestion. Most biogas plants has no measurement of the heavy metal input from feedstock, and are under the assumption that no heavy metal is lost in the process. The law also specifies that the contamination of the biofertilizer that has to be taken into account. However, an assumption that most of the heavy metals comes from sewage sludge is common. Vast differences in treatment option and addition of micronutrients or a high degree of pig manure can alter this picture. The heavy metal content is therefore, in this study, given by the categories standardized for Gjødslesforskriften §10 right after digestion table 4.

Emission to waterbodies is limited to N and P runoff by leaching. The rate for N-biofertilizer has been found in the literature to be 0% and 22% of applied nitrogen, for surface water (river) and groundwater respectively (Bernstad & Jansen 2011).

P runoff from bioresidual has been hard to come by so an average leaching for the Norwegian total P to agriculture has been calculated based on (Hamilton et al. 2015). They assume that of 25 000 ton organic fertilizer, 3000 ton plant and seeds, 84 000 ton mineral fertilizer goes 12 000 ton to run off. This estimate consist of both mineral fertilizer, the majority, and organic fertilizer. The ratio found here is 10.7% P loss after application to run off. This is a highly uncertain estimate in this context, but has proven to be the most reliable source in a Norwegian

context. It is therefore excluded in the current study, but can be applied as comparison data for epistemic uncertainty in further studies.

Emission to air **has** in this model been divided into the two mentioned sub compartments, C based and N based. To preserve the feedstock driven modelling approach, are these emission based on mass balance principle. The carbon-based emissions (CH_4 and CO_2) are assumed to be the part of the biogas potential that has not been degraded in the anaerobic processing, (Luostarinen et al. 2011; Amon et al. 2006; Bernstad & Jansen 2011). Thus has the C based emissions been calculated by:

Equation 10: Emissions caused by storage of digested organic wastes

$$CH_{4,storage} = \left(\sum_i (\mathrm{DM}_{AD,i} * \mathrm{MY}_{VS,i}) - \sum_i (DM_{AD,\mathrm{i}} * \mathrm{MY}_{Degraded,i}) \right) * \mathrm{SL} * \mathrm{S}_{\mathrm{Technology}}$$

$$Bg_{,storage} = CH_{4,storage} * CH_{4,\%Bg}$$

$$CO_{2,storage} = Bg_{,storage} * \left(1 - CH_{4,\%\mathrm{Bg}}\right)$$

$$Bioresidual_{in} - (CH_{4,storage} * \mathrm{m}_{CH_4} + CO_{2,storage} * \mathrm{m}_{CO_2}) = Bioresidual_{out}$$

$$i = sub\text{-}waste \text{ in } mass$$

$Bg_{,\,storage}$ here, are defined as the post produced biogas, during storage of bioresidual, MY_{VS} is the Methane yield per ton VS of each substrate (i), $MY_{degradable}$ is the actual CH^4 potential in digester while DM_{Pt} is the total dry matter mass entering the digester. (SL) represents CH_4 produced from the total remaining biogas potential in the bioresidual and S is the methane inhibiting factor the storage type, covered or tight sealed, offers. $CH_{4,\%,Bg}$ is the methane share found in the biogas produced, and a uniform share between Bg_{AD} and $Bg_{Storage}$ is assumed.

Table 8: Emissions per type of bioresidual type relative to digested manure. Data based on Amon et al. (2006)

Storage type	NH_3	N_2O	N_2[3]	CH_4
Digested	100%	100%	100%	100%
Undigested(1)	99%	77%	100%	301%
Separated(2)	178%	119%	100%	58%

[1]Based on the relative differences between Digested manure Undigested manure (Amon et al. 2006).
[2]Based on the relative differences between Separated manure Undigested manure (Amon et al. 2006).
[3]Assumed to bee 100% for all treatment methods due to no data found on the field.

To account for possible emissions occurring by storage of manure before digestion a similar approach has been used where a fraction (10%) of the digestible material are digested before transport to the plant.

The nitrogen-based emissions include N_2O, NH_3 and N_2. There are other emissions as well, but those has not been included in this study. Mainly due to the increased complexity and lack of

data availability at current stage. Nitrogen based emissions is a part in the loss of N in the anaerobic treatment and has therefore implication far beyond the direct effects. As the literature describes is there many approaches for calculating N-emissions and even more factors affecting it. In this thesis however, has a new method that is particular designed to the feedstock driven model based on both literature and empirical data been developed, equation 2.

Equation 11: Nitrogen loss due to gaseous losses

$$mN_{xx,j,i,z} = \left(\frac{mN_{in,z} * Nl_{\%,z} * N_{xx,\%,\mathrm{dig,j}}}{N_{Nxx,\%,\mathrm{j}}} \right) * \mathrm{L}_{Nxx,j,\frac{dig}{i}}$$

$$mN_{Nxx,j,i,z} = N_{Nxx,\%,\mathrm{j}} * mN_{xx,j,i,z}$$

$$mNl_{i,z} = \sum_{j} mN_{Nxx,j,i,z}$$

$$mN_{out,i} = \sum_{z} mN_{in,z} - \sum_{z} mNl_{i,z}$$

$$z = manure\ digested, manure\ undigested, organic\ waste$$

$$j = NH_3, N_2O, N_2$$

$$i = digested\ (dig), separated, (undigested)$$

To determine the mass of gasses is essential, both in an MFA and for the LCI step. The N loss due to release of N containing gasses is no exception. The mass (m) of nitrogen compound (Nxx) for either NH_3, N_2O, N_2 (j) due to treatment type (i) of manure or organic waste (z) is found by determining several parameters that affect the generation of these in the different treatment types. The model is feedstock driven and thus the mass of mN_{in} that enters the system, determines the production of NH_3, N_2O, N_2. The ratio between these N containing gasses is based on the loss ratio (%) of nitrogen (Nl) for a given waste type (z), the ratio of NH_3, N_2O, N_2 produced in digested bioresidual ($N_{xx,\%,dig,j}$) devided on the mass of N within each of the gases (j). A further adjustment for the loss rate differences for each of the gas (L_{nxx}) with respect to the different storage types, digested, separated or untreated (i), where digested is the option of reference(dig*i^-). The untreated option is only optional for manure applied directly as fertilizer (MDF) Appendix 2. To find the actual N lost in terms of mass due to generation of these gases (mN_{Nxx}) for each of the treatment options (i) and waste types (z) is it necessary apply $N_{Nxx,\%,j.}$ To find the total N lost in terms of mass is it necessary to sum the mass of N that is converted into NH_3, N_2O, N_2. By applying the new total N lost, in terms of mass, ($mNl_{i,z}$) can the transfer coefficient for N gases from the bioresidual be identified, both for storage and application.

As such is the mass balance principle kept, with respect to variations in treatment of the bioresidual. However, the sum of N loss have in the SimaPro 8 model have been, at current stage, applied as one in the variable calculation step. This makes it impossible to determine how

much gas are emitted during either storage or application in terms of MFA modelling. MFA have in this thesis only been applied for its ability to generate variation consistent outputs trough the system. Therefore have no particular attempt towards complete system flow analysis been attempted. The LCA however have been modelled to account for this variation..

The $N_{xx\%}$ split is based on the assumption that, of the total N-emissions are 96% NH_3, 2% N_2O and 2% N_2 (Bernstad & Jansen 2011). A further adjustment for the technology option of storage has been applied ($L_{Nxx,dig/i}$), as with the carbon-based emissions, table 7. Because each compound has different mass, an adjustment for mass of each molecule had to be made based on the amount of N lost, table 9. *t*

Biogas emission from anaerobic digester (AD) and upgrading technologies (UpT).

To estimate the loss of methane and biogas mass in the digester, an assumed value of 0% has been applied, based on qualitative empirical data from Frevar and Ecopro. In the literature however, a range from 0% to 8% (Bernstad & Jansen 2011) been suggested. Biogas emission from biogas upgrade technology depends on the technological upgrading choice, and as such varies a lot.

In Norway, waster scrubbing have been assumed the most common technology applied. The resulting emissions is proportional to 98% CO_2 and 2% CH_4 of input biogas gas. These are listed in Appendix 21, parameter 66 – 77. In an eventual case of biogas utility for heat, has an assumed methane loss of 2% been implemented, based on the water scrubbing technology assumption, Appendix 21, parameter 66. The CO_2 content of the leakage methane is at the same ratio as the biogas CO_2 content and is found by: $Lr_{CO2} = (CH4*Lr_{CH4}/BG_{CH4})*BG_{CO2}$. For LBG is the situation a bit different, because it already requires to be upgraded to biomethane and thus the found 1.8%[13] biogas loss comes in addition to the biogas upgrade loss.

Pasteurisation Calculations

Pasteurisation calculated with a level of recycling that is determined by the energy difference between the temperature inn and the temperature out after the pasteurization. This is a common technology that makes it possible to pre-treat the organic wastes with a lower energy input that would otherwise be the case. The specific heat capacity (SHC) for water is 1.1704 kWh / ton*°C (Coultry et al. 2013; Hamelin et al. 2014). The amount of mass pre-treated is the difference in temperature (ΔT_{Tot}) between the theoretical heating difference in to pasteurization (ΔT_p) and the theoretical difference between pasteurization temperature (T_p) and the digestion temperature (T_{AD}), (ΔT_{AD}). The energy loss is an assumed value in the range of 5 – 10%

[13] Found in Appendix 16, parameter 20

dependent on the applied system, and determined both for the heating process (α) and the energy recapturing process (β). This is executed on the basis of the mass (m).

$$E_p = SHC * \Delta T_p * m * (1+\alpha)$$
$$E_{rc} = SHC * \Delta T_{AD} * m * (1+\beta)$$
$$E_{p,rc} = E_p - E_{rc} = SHC * \Delta T_{p\gamma} * m * (1+\alpha+\beta)$$

In the SimaPro 8.01 model is the energy loss defined as Lr_HP= α+β, and there is thus no distinguishing between the loss in recapture and heating loss. This calculator is developed into a pretreatment heating table where the variables is the feedstock input temperature, pasteurization temperature and the input temperature after heat recovery (γ), Appendix 18.

4.8. Sensitivity analysis development

As previously mentioned a change in parameters might has a small or great effect on the total outcome of the system. Therefore is it important to identify which of the variables that is affected by this change and their importance for the overall performance of the system, both for environmental and energy recovery purposes. To connect the relative change in variables compared to variation in parameters is important to establish the sensitivity described in Baumann & Tillman (2004). By using knowledge found in the literature is there serval variables that should be tested for variations, such as NH_3 and CH_4 losses, mass variation based on feedstock variations and their effect on total ton kilometre and changes in substitution potential do to a change in feedstock in addition to end treatment utility for both energy, impacts and fertilization potential. The results for the executed sensitivities is described in the results and summarised in figure 5. For he substrate mix have the changes applied been based on information from the average feedstock of several plants in Norway, table 10.

Table 9: Organic waste substrate composition for the sensitivity analysis of organic waste mix

Case	Tønsberg	Bergen	Fredrikstad	Norwegian mix
Manure ratio[1]	20.3%	0.0%	0.0%	50.0%
Fat ratio[1]	0.0%	3.0%	0.0%	0.0%
OMW ratio[1]	79.71%	9.0%	21.8%	13.1%
OIW ratio[1]	0.0%	0.0%	50.8%	32.1%
Sludge ratio[1]	0.0%	88.0%	27.5%	4.8%
Product X[2]	0.0%	0.0%	0.0%	0.0%
Source	(Hegg 2015)	(Sande et al. 2008)	(Jørgensen 2015)	Morken et al. 2008[3]

[1] The sub organic waste composition of all the categories have been assumed the same as for the Norwegian average, Case 1 found in table 6, and the waste mix is therefore not representative for the assumed biogas plants in table 10. [2] Product X is a product in which own assumptions for individual cases can be developed and thus not included in this report. [3] The Norwegian mix is applied as described in table 6, Appendix XX.

Technology sensitivity

To test the variation of technological options in this thesis can be of relevance to determine the variations in technological output potential. Norway is still in the developing stage and therefore is it relevant to apply a sensitivity to technology. The technological options in the model is limited to upgrade technologies of biogas, biomethane and LBG substitution options in addition to different treatments and utility rates of bioresidual. Those options are handpicked for technological variation as they represent the greatest variation in the Norwegian biogas industry.

4.9. Case sensitivity development

In all of the cases below are a Norwegian feedstock applied as well as a Norwegian electricity mix. All of the cases are fictional, some are closely related to actual plants and treatment methods applied in Norway, but not representative for the actual plants at given stage. A further agreement with those firms must be in order before an actual case specific analysis is attempted, which is outside the boundary for the given thesis. A further assessment of such nature would be part of further development and model usage.

Case 0 – Incineration

Organic waste today, undergo the same treatment as other wastes, municipal incineration. The organic fraction makes up approximately 6% of the total incinerated mass (Ecoinvent 2015). Manure undergo seldom any form for treatment but is stored at the farm and applied during spring. The manure does therefore not require any transport other than the spreading. An average transport distance have been assumed for the organic waste. These are transported approximately 50 km for each waste type, except the OMW (19 km). The plant has a standard efficiency of 80% (Hung & Solli 2011) heat and no electricity production, which substitutes electricity for heat as is the most common heat source in Norway. The fly ash is transported to Langøya for hazardous waste landfill underground (250 km) and bottom ash to local landfill 0.5 km.

Case 1 – General case for Norway, Bus substitution

The general case for Norway have been based on a high level of assumptions. As it was discovered in the empirical studies is there no uniform biogas plant structure. However, based on the number of applied technologies and information form Tore Fløan (2015) and Raymon Jørgensen (2015) have a definite production line been developed.

Biological is in almost all cases pre-treated to a water content of 90%, Appendix 17 parameter 3, and pre pasteurised at 70℃. Most of the heat, with a small loss of 5%, is recovered form the pasteurization process and used to pre-warm the organic waste substrates form 14 to 44 - 52 ℃ before heating of organic waste requires any added heat. It is possible to recover some of the heat from after the digestion, but this has been excluded in this study. The biogas produced is in most cases upgraded by water scrubbing, but PSA was more applied before, and chemical scrubbing is about to take the marked. So water scrubbing is implemented in this case. For the biogas production is there accounted for no increase or decrease compared to digestible potential of each individual product. The energy for heat requirements is highly dependent on the plant, but the Norwegian electricity mix have been assumed here. The transport of organic waste can in many cases be far, but in this case have an average of 50km for each organic waste category been assumed. Only OMW have been given a shorter transport value. This have been applied as most of the big biogas plants in Norway is being built in relative close proximity to nearby cities. It has also been assumed that the bioresidual requires the same distance as the organic waste. For the biofuel is the situation different as it often is built close to the plant. The HM content in bioresidual is in most cases satisfying class 1 requirements, figure 4.

Case 2 – The regional biogas CHP with an aim to maximize biogas production

In this case is the main objective to treat biogas most efficiently. To reach this goal have an extensive effort towards degrability amplification been executed. Based on information from Ariunbaatar et al. (2014), can high temperature pasteurization achieve an 11.9% increase in methane production. The digestion afterwards is thermophilic to ensure that an increase in methane production is true. Due to the increased methane production, an overproduction is evident and thus approximately 5% of the produced gas are torched. The biogas in this case is, for time being, just converted into 12% electricity and 86%, achieving a total energy output (biogas) to energy efficiency of 96% (Hung & Solli 2011). The plant is located 15 km form town, which in this case result in a total OMW transport of 30 km. The farmland around the plant (5km) have no live stock. The closest farm, which delivers all the manure, is 150 km away and only the sewage sludge from that region is applied in this case. Instead of using electricity, utilizes the biogas plant wood pellets as heat source.

Case 3 – A plausible biogas treatment with LBG for buses

By aiming towards reducing fossil fuel consumption and thus both the impact categories GWP and FDP have biogas for buss been implemented in this case. However to increase the fuel capacity for the bus is it necessary to further upgrade the biomethane to LBG. Another reason for the applied LBG treatment is that the fuel station is 150 km from the plant and to increase the lorry capacity was LBG thought a good solution. The bioresidual is to be applied 10 km from the biogas plant and thus has it been deemed unnecessary to dewater the bioresidual. also proposed that no treatment of the bioresidual would maximise the nutrient recovery rate and its

utility is maximised because the measured HM levels here is within class 0. The most efficient and common pasteurization temperature 70℃ is before thermophilic treatment where the local municipal plant supplies the biogas plant with necessary heat. In this case was it assumed that a 2.69% increase in biogas production was caused by the choice of pasteurization technology.

Case 4 – General case for Norway, Natural gas grid

By substituting natural gas in grid is it predicted that the impact for GWP should decrease as a result of avoided fossil CO_2 emission. The bioresidual in this case have proven to contain class tree level of HM contaminations and is thus composted and applied for urban recreational purposes. The assumption that NH_3 and N_2O, table 4, is represented by another share than for bioresidual have been implemented for the compost. The feedstock is pasteurized at 80 degrees as proposed as a reference temperature in the study Ariunbaatar et al. (2014), but no change in methane yield due to co-digestion is apparent, figure 4. The pasteurisation heat is produced form electricity. No (BAT) inhibition technologies have been applied in this case. The biogas undergo upgrade to biomethane, by PSA technology, and distributed to the local gas grid network. The only place in Norway with an extensive gas grid network is the Stavanger region and Case 4 can therefore be seen as an assumed setting for this region.

Table 10: Parameter changes according to each case

No.	Input parameters	Case 1	Case 2	Case 3	Case 4	Case 0
1	DM_AD	1	1	1	1	1
2	Incineration	0	x	x	x	1
13	Codigestion_Benefit	1	1.119	1.0269	1	
33	MDF	0	x	x	x	1
34	Digestate_Use_Wet	0	x	1	x	x
35	Digestate_Use_Dry	1	1	x	x	x
36	Digestate_Use_Separated	0	x	x	x	x
37	Digestate_Use_Compost	0	x	x	1	x
47	Bm_Compression200	1	1	1	x	x
48	Bm_Compression300	0	x	x	x	x
49	Bm_Compression45_50	0	x	x	1	x
55	Torch	0	0.05	x	x	x
56	UpT_Cleaning	0	1	x	x	x
57	UpT_WS	1	x	x	x	x
58	UpT_ChS	0	x	1		x
59	UpT_PSA	0	x	x	1	x
60	UpT_Membrane	0	x	x		x
61	UpT_Cyrogenic	0	x	x		x
62	HM_0	0		x		x
63	HM_1	1	1	x	x	x
64	HM_2	0	x	x	x	x
65	HM_3	0	x	x	1	x
92	DMC_SwSl	0.17	0.4	0.17	0.17	0.17
213	AFSH	44	120	32	32	
214	PH	70	155	70	80	
278	km_Manure	50	150	50	100	x
279	km_Fat	50	x	x	x	x
280	km_OMW	19	30	50		x
281	km_OIW	50	150	50	100	x
282	km_SwSl	0	150	0	0	x
287	km_Fly_Ash	0	x	x	x	250
288	km_Bottom_Ash	0	x	x	x	1
298	NH3_dig	0.96	0.96	0.96	0.024	
299	N2O_dig	0.02	0.02	0.02	0.014	

All the parameters above are represented in Appendix 21. The No. list represents the parameter number in the Appendix 21 list. All explanations are found in this appendix.

5. Results

The LCA results for the presented cases developed is chapter 4.9 is presented. Total transport required in addition to energy and nutrient efficiency have been included to assess other important aspects of each case. Focus towards reviling environmental impact caused by various biogas combinations, seen in relation to the common organic waste treatment, is the purpose of these results. Stochastic variations have been fond trough the literature study, and the most common of these have been tested. Epistemic uncertainty is less quantifiable, but an effort towards interpreting this will be performed in the *Uncertainty chapter*, later in this report. However, some of the quantifiable sensitivities are based on an assumed epistemic uncertainty in particular the effects N have on the overall biogas value chain. Four impact categories have been chosen to assess various aspects of the organic waste management options. GWP and TAP, the wo that is most frequently assessed and HTP and FD, which is much less included in such studies. Particularly studies in an Norwegian context lack HTP. These results will be discussed and interpret in *Chapter 5, Discussion.* All results are assessed per FU as described in the introduction to this study, *the treatment of one tone dry matter organic waste substrate*.

None of the results are applicable for any specific Norwegian biogas plants. However, Case two and three have been largely inspired by two biogas plants, Ecopro and Frevar, respectively. However, their contribution is to illustrate the variation Norwegian biogas production are subject to. s

5.1. LCA results

The LCA results have highlighted the most important variables, processes and parameters. The relation between these will be discussed in the *Discussion* chapter. The main case results are presented in figure 5 below, while the LCA sensitivity results are presented in figure 4.

Of the four cases, Case 1 contributed to the largest net CO_2 savings (-306 kg CO_2 eq) for the GWP category. The reference case is the only case that contributes to increased GWP impacts (622.9 kg CO_2). Case 2, have proven to be the biogas case that yields the least negative impacts for this category (-10.7 kg CO_2 eq). The four biogas cases contribute to net GWP savings, as illustrated in figure 5.

Case 4 is the case that yields the least impacts of TAP, where the categorized result is estimated to be 1.1 kg SO_2 eq. The organic waste treatment method applied in Case 3, yields the largest net impacts of a total increase of 22.2 kg SO_2 eq. The reference case yields a total impact of 16.6 kg SO_2 eq, which is the third least impact.

Case 1 yield an potential impact of 171.6 kg 1.4 DB eq, which is the largest HTP of all cases, both biogas and incineration. Of the anaerobic treatment value chain cases, Case 4 causes the

least potential HTP impacts, 20.2 kg 1.4 DB eq, compared to the assumed biogas treatment methods. The reference case, yields the least HTP impacts, only contributing to a total of 6.8 kg 1-4 DB eq. Seen in relation to the biogas options is these impacts one third, in comparison to the least HTP intensive biogas case.

Case 1 and 4 contribute to almost equal net negative impacts for the FDP category, - 116.4 and -116.7 kg oil eq. This despite the difference in transport distance between those two cases. Case 0, is the only case that contributes to positive net FDP impacts by increasing the net FDP to 26.1 oil eq per FU.

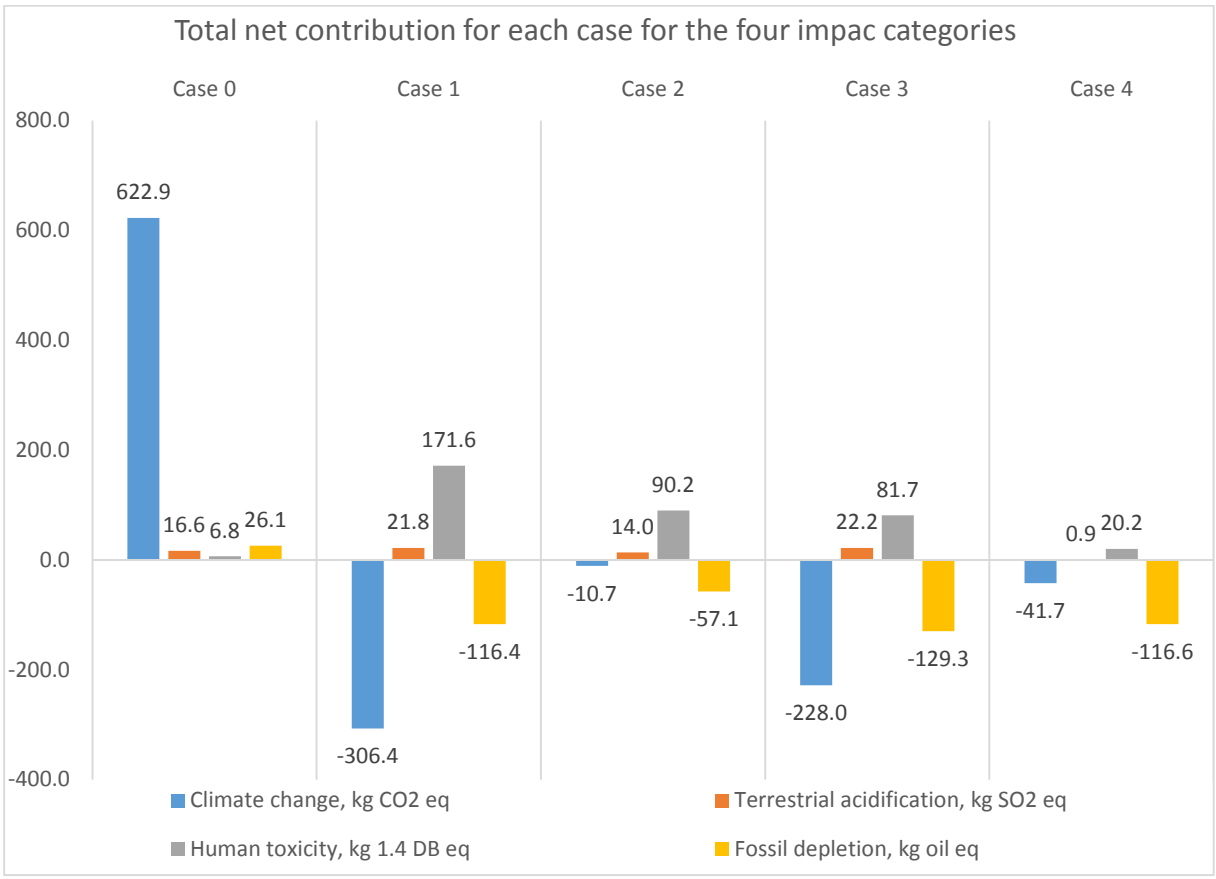

Figure 4: Net impacts for each impact category for each organic waste treatment case. Climate change is measured in kg CO_2-equivalents(eq), Terrestrial acidification is measured in kg SO_2-eq, Human toxicity is measured in kg 1.4 DB eq and fossil depletion is measured in kg oil eq.

GWP

The stressor that causes the largest share of GWP impacts varies for each of the value chain options investigated by each of the cases. These are evidently fossil CO_2, biogenic CH_4 and N_2O. In all cases, are the combined effect of these stressors responsible for more than 89.3%

of the total GWP. Of these, CH_4 contributes to the largest share of impacts for Case 0 and 3, while fossil CO_2 causes the largest impact for Case 1, 2 and 4 as represented in table 11.

Table 11: Stressor contribution GWP, positive emissions

GWP	Case 0	Case 1	Case 2	Case 3	Case 4
Biogenic CH_4	68.9%	22.6%	19.1%	43.4%	32.7%
N_2O	10.4%	16.4%	14.9%	15.9%	18.9%
Fossil CO_2	19.3%	50.4%	56.8%	37.5%	44.7%
Others	1.4%	10.7%	9.2%	3.2%	3.7%
Total for the three main contributors	98.6%	89.3%	90.8%	96.8%	96.3%
Total contribution	100.0%	100.0%	100.0%	100.0%	100.0%

The stressor contribution, however, is of another composition when the negative impacts are assessed. It is evident that fossil CO_2 is the stressor that is substituted the most for all of the cases, represented in table 12.

Table 12: Stressor contribution GWP, negative impacts

GWP stressors saved	Case 0	Case 1	Case 2	Case 3	Case 4
Biogenic CH_4	15.2%	4.3%	8.1%	2.70%	0.0%
N_2O	12.6%	2.2%	4.7%	5.4%	0.0%
Fossil CO_2	68.1%	91.4%	78.4%	89.6%	95.2%
Others	4.1%	2.1%	8.8%	5.0%	4.8%
Total for the three main contributors	95.9%	97.9%	91.2%	95.0%	95.2%
Total contribution	100.0%	100.0%	100.0%	100.0%	100.0%

Biogenic CH_4 and N_2O mainly occur due to storage of either manure or bioresidual. For fossil CO_2, transport is the main source of origin. The process that contributes to the largest impacts is, as with the stressors, highly dependent from case to case. There is however, some patterns that have been detected. For Case 0, *Storage of manure* is the main contributing process for the GWP impact category by 70.8% of the total impact. The incineration process itself contributes only to 22% for this impact category, as seen in figure 7.

TAP

The greatest amount of TAP contribution is evident in Case 3. The main contributor to this impact category (TAP) is the release of NH3 (95%), where 99.7% of the NH3 originates from

the *Post treatment storage and application* process. Only when compost is applied is the TAP close to 0 as seen in Case 4, where it amounts 1.1 kg SO2 eq. For all other cases, the net TAP varies from 14 for Case 2 to 22.2 kg SO2- eq for Case 3. The greatest contributor to negative TAP impacts is the application of biofertilizer, as seen in Case 0, 1 and 3 or by substitution of heat (see figure 7). In Case 4, which does not apply biofertilizer, the major contributor to negative TAP is the process *Transport of bioresidual*. The greatest contributor for Case 1, 2 and 3 can be related to the process *Post treatment bioresidual processing*, *figure 7*.

HTP

Table 13 shows the responsible stressors for this impact category. The main stressors are heavy metals (HM) released to the atmosphere, soil or air, as well as the non HM arsenic (As). For the biogas, cases where biofertilizer is applied cause the presence of zinc (Zn), cadmium (Cd) and lead (Pb) the greatest impacts. Zn is the most important HM of these, and contributes in Case 1 and 2 to more than 50% of the total HTP impact. For Case 0 and 4, arsenic is the most prominent chemical, which also is not a HM. It is also evident that, for incineration, any of these previously mentioned HTP stressors contribute the most, but other chemicals such as selenium (Se).

Table 13: Stressor contributor to caused HTP

HTP	Case 0	Case 1	Case 2	Case 3	Case 4
Zinc (Zn)	1.5%	54.0%	52.0%	35.0%	4.0%
Cadmium (Cd)	4.5%	23.0%	23.0%	20.0%	10.0%
Lead (Pb)	11.4%	11.0%	11.0%	13.0%	27.0%
Arsenic (As)	18.7%	4.0%	4.0%	13.0%	39.0%
Others	82.6%	12.0%	14.0%	32.0%	59.0%
Total for the tre main contributors	17.4%	88.0%	86.0%	68.0%	41.0%
Total contribution	100.0%	100.0%	100.0%	100.0%	100.0%

The processes that contributes the most to caused HTP impacts are for Case 1, 2 and 3 the *Post treatment of bioresidual*. This is true for Case 4 as well, but both the absolute and relative impact is less due to this impacts. Only 32% fo the total impacts, in Case 4, is due to *Post treatment of bioresidual. On another hand is the Natural gas substitution* responsible for 24% of the total impacts. For Case 0 it is the incineration process itself that contributes to the largest impacts, while the processing of ash is responsible for the second largest contribution.

The processes that have most significant effect as a contributor to negative or avoided HTP impacts is *Heat substitution*, - 91.1 kg 1.4 DB eq for Case 2. *Biofertilizer substitution* contributes in the case of Case 0 and 2 to -6,4 and 5.8 kg 1.4 DB eq. However the respective share is much lower for *Biofertilizer substitution* in Case 2 than for Case 1. Case 3 contributes to – 21 kg 1.4 DB eq which is the greatest amount of avoided HTP impacts.

FDP

For all cases are the extraction of crude oil the main cause of FDP. It is responsible for between 47.1% for Case 4 and 54.1% for Case 3. Natural gas extractions is the second most influential stressor for fossil depletion in all the assessed cases in the range of 33 – 37 % of the total impacts. It is Case 1 that have the largest potential to reduce the impacts of FDP by a total of 181.1 kg oil eq where 171.8 kg oil eq of these occur due to substitution of fuel by use of biomethane. For the other cases is the situation the same as for GWP that the substitution of fossil fuels is the major contributor to possible avoided FDP. These two impact categories is also closely linked as fossil CO_2 occur from burning fossil fuels which contributes deplete a fossil reservoir to be made.

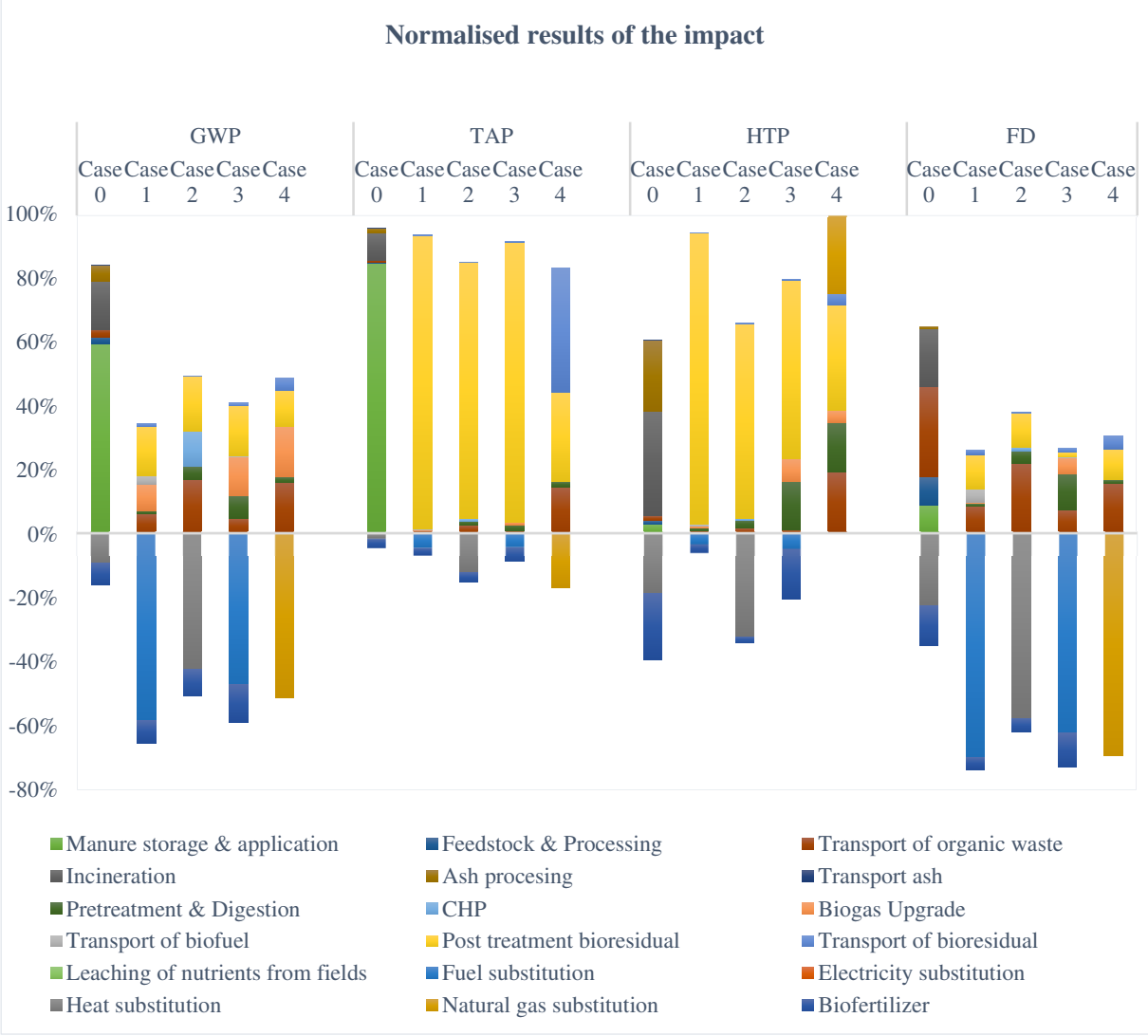

Figure 5: Normalised results for all cases in comparison to all impact categories and process of origin

5.2. MFA results

In this partial chapter is the results from the MFA presented. The results of interest are the energy flow of the system (EFA) and the overall substance flow of the nutrients N and P and these are included in this chapter.

Table 14: Total Transport required for each case

Process	**Transport energy**	Case 0	Case 1	Case 2	Case 3	Case 4	
T1	Organic waste substrate (Feedstock)	374	678	1570	678	274	MJ / FU
T2	Biofuel	0	22.6	0	26.9	0	MJ / FU
T3	Bioresidual	0	121	9.51	101	164	MJ / FU
T4	Ash	14.8	x	x	x	x	MJ / FU
	Energy Transport	388.8	821.6	1579.51	805.9	438	MJ / FU
	Diesel Transport	10.9	22.9	44.1	22.5	12.2	l / FU
	Diesel Transport	9.2	19.5	37.5	19.1	10.4	kg / FU

Case 2 requires the most transport and also is the case that causes the largest transport energy demand and thus requires most diesel. As previously describes is transport a product of mass and distance where the only variable for organic waste is distance. All cases is subject to the same amount of mass transported, except for Case 0 where manure is applied directly as biofertilizer and thus requires no transport. evident that the total transport of the organic wastes to treatment facility vastly exceeds the other transports stages. This is due to the reduced transport mass in Case 1 for dewatered bioresidual. For the other AD cases is drastic reductions in transport distance to application destination the cause of reduced total transport for bioresidual.

Table 15: Energy efficiency for each case with respect to output energy

Efficiency rates	Case 0	Case 1	Case 2	Case 3	Case 4	
Feedstock energy	16800	16800	16800	16800	16800	MJ / FU
Process energy	416.7	1982.3	1929.1	3323.6	2232.2	MJ / FU
Transport energy	388.8	821.6	1579.5	805.9	438.0	MJ / FU
Available energy out	4600.0	8750.00	9470	8,880	8,750	MJ / FU
Feedstock to Biogas efficiency	27.4%	52.1%	56.4%	52.9%	52.1%	
Process to biogas efficiency	1103.8%	441.4%	490.9%	267.2%	392.0%	
Transport to biogas efficiency	1183.1%	1065.0%	599.6%	1101.9%	1997.7%	
System energy requirement to biogas	571.0%	312.1%	269.9%	215.0%	327.7%	
Total efficiency rate	22.6%	35.4%	35.5%	28.3%	36.2%	

Based on the previous assessments of the process energy requirements and the energy required by transport in the various transport stages, is it possible to derive the total efficiency of the value chain.

The most efficient system in term of energy is Case 4 which a total efficiency of 36.2%. The least efficient case is the reference case with only 22% energy recovery. The least efficient biogas case is Case 3 recovers about 28.3% of the total energy found in the feedstock. Case 3 also have the largest share of process energy, 3323.6 MJ / FU. Case 0 requires the least energy for both processing and transport as much of the mass assessed in terms manure, is treated directly as fertilizer and does not require any other processing that spreading. However, by applying manure directly at field is much of the potential energy "lost" and therefore have Case 0 the lowest energy output, which is the reason for the low efficiency.

5.3. Parameter sensitivity

The parameter sensitivity is of particular importance as described by Clavreul et al. (2012). The most variating and unsertain parameters has been tested with respect to the total impact due to the fullfillment of the FU through Case 1. This case has been applied sa sensitivity because the assumed average production method in Norway and to be able to answer the reseach question of the study.

Stoahstic uncertanities is the main targeted data in this sesnitivity analysis. However, some assumptions towards inkonplete knowlede have been assumed for to include epistemic uncertaities.

Development of the sensitivity analysis

Leakage of methane during digestion is one shuch parameter, described by Modahl et al. (2014). The average assumptiopn in the industry is 0, while most reseach studies include an variation from 0% to 1% or 1 – 8%. This is maby due to both stohastic and epistemic uncertaities. Recyceling of proces water from anaerobic digestion is an common aspect, viewed as an posibilit for Norwegian biogas plants. And 50% recyceling have therefore been tested. A 12 – 30% variation in post digestion of remaing VS has been found in Luostarinen et al. (2011). Compared to findings from (Amon et al. 2006) was an 12% post digestion indentified. However, a sensitivity towars this stohasic uncertanity have been executed. N_2O production from bioresidual is a parameter which yielst high stohastcic uncertanity. Bernstad & Jansen (2011) have refferred to sreveral assumed values for this paramter. However, while the 2% of total N loss where selected have the sensitivity been tested for 0.7%. The latter parameter was also one of the assumed values in Bernstad & Jansen (2011). It have also been interesting to investigete the impact of premature digstion of manre at storage, before digestion. An assumed 10% of the total degradable biogass yield were tesed, comared to the normally assumption of

0%. The last sensitivity test where for N, where a 20% change where implemented, both as an increase and decrease.

Sensitivit results

The loss of 8% CH_4 during storage is the parameter change that has casues the higest increase in GWP (111.3%) leading to an total ipmact of 34.7 kg CO_2-eq. Taking into account that a 1 % loss of the same parameter causes the third largest emission output (-263.8%) and at the same time being the smallest change applied in the paramerter sensitivity, is it fair to assume that this parameter is the most sensitive to change and the one that affect the GWP the most. For TAP is a 20% increase in N contnten that induced the largest impact cahnge in TAP with an total impact increase of 23.1%. This value is assumed on basis of the high variation in N content found in the literature for various wase commodities. HTP vaires relatively little for all of the tested parameters and is therefore assumed to not be very sensitive to changes in these parameters. However, as previously found in the case result variese the HTP maily by changes in assumed HM contents, in his thesis based on the clasification requirements. The greatest increase in FDP (11.0%) is caused by the largest applied leakage of methane (8%) while the biggest decrease in FDP (-6.9%) is caused due to an increase in methane production (5%), figure 5. Total results are presenter in Appendix 3.

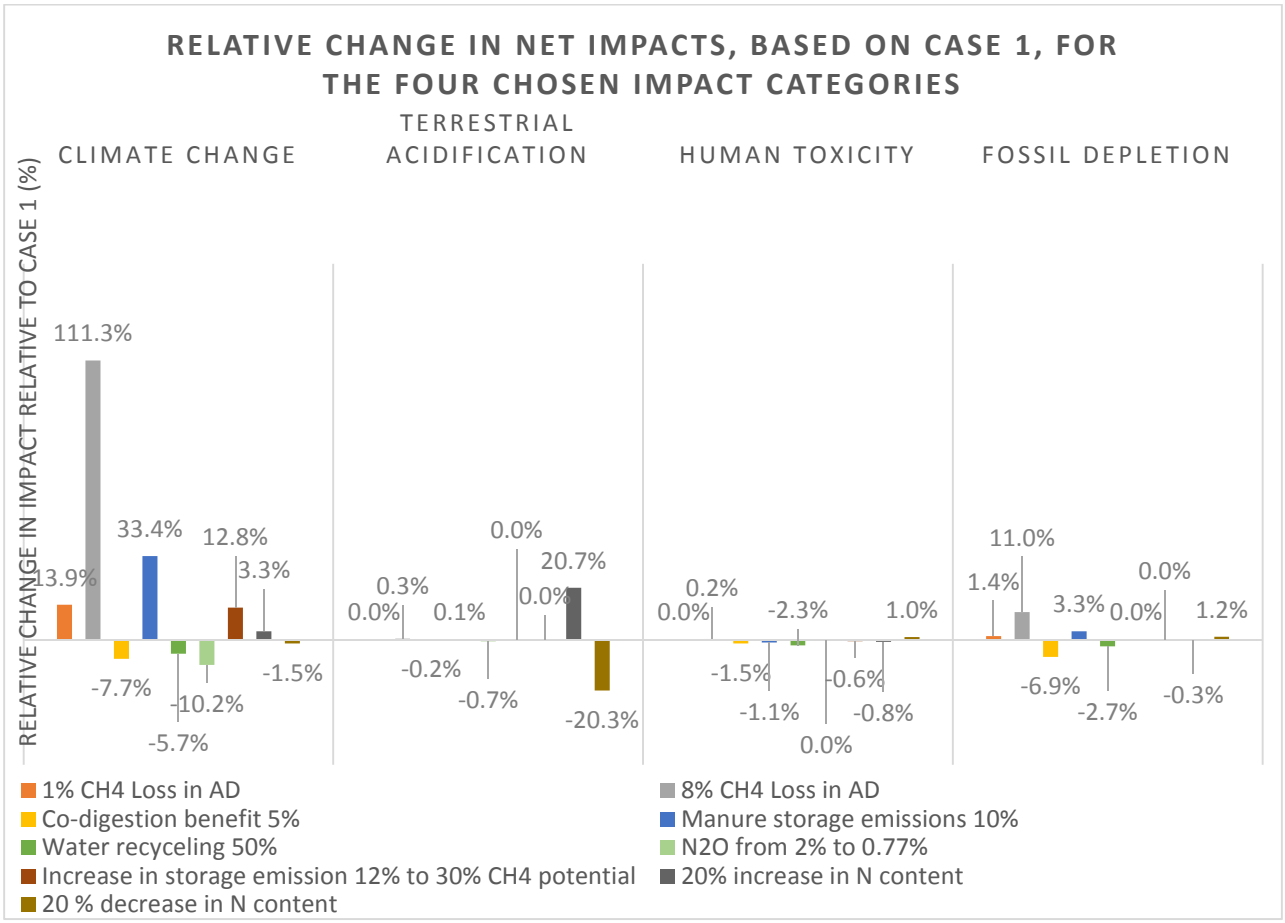

Figure 6: Relative parameter sensitivity results for Case 1

6. Discussion

The evaluation of critical variables is the focus of this discussion. Earlier in the report, both a literature study and the model development have been the focus to identify important parameters and processes. The development of the model has been shaped by the information found in the literature study and data from the literature study have been applied in the model. By implicating data found into the model and develop scenarios likely for Norway, has made it possible to estimate the quantitative environmental impacts of biogas production and alternative treatment methods. The scenarios has been tested as forecasting scenarios, due to the particular scope of the study.

Two main treatment possibilities for organic wastes, biogas production and incineration, are assessed in this study. The five developed cases has been examined closely to give a fair background of understanding of how the SimaPro 8 model works, and to answer the research question of the study. These cases are composed of different technological alternatives, where four of the cases assess biogas production.

During the literature study and LCI where several critical variables and stressors identified. These have been given particular focus in this discussion with respect to the results in this thesis. The overall environmental impact of anaerobic digestion and the reference case where presented in previous chapter. Assessment of these results are to be further discussed an interpret in this chapter.

The purpose of this chapter is to assess and discuss the variables and stressors that affect organic waste treatment in Norway, and as such answer the research question given in the introduction of this study.

"Which stressors are critically influencing the environmental life cycle impact of the biogas production in Norway, in comparison to the alternative organic waste treatment option, and which factors and variables limits or enhance these"

6.1. Critical variables and process relationship

The most critical variables found in the LCA results of this study, highly depends on the characterisation category that are assessed, and the stressors that affect them. During this chapter these stressors are seen in relation to underlying variables such as processes of origin and the affecting flow of mass and energy. The most important flows are presented in Appendix 20.

GWP

It is evident that biogenic CH_4 and fossil CO_2 contributes to the majority of the emissions. N_2O contributes to about with 14.9 – 18.9% for the biogas cases and 10.4% for the incineration case. It is interestingly to identify where these emissions occur and as presented in chapter 5.1.

Fossil CO_2 can occur mainly form two processes, transport and energy production. For the energy production it is evident, in figure 5, that *Pretreatment and digestion* contributes to relatively small impacts, but for Case 3, where municipal waste incineration have been used as heat source, this picture changes. This variation is also evident in Appendix 5, *Pasteurization heat energy sensitivity,* where several heat sources have been tested and analysed. This shows that a change of heat source have major impacts on the overall GWP for biogas production. It is also evident, from the process energy assessment in Appendix 19, the heat required varies from case to case. The mass to pretreatment is constant at 10 ton wet mass, thus the choice of pasteurization temperature is the important factor. This gives that one set of important parameters for CO_2 stressor release, is the pasteurisation temperature and the choice of heat source.

Transport the major variable that directly affect the release of fossil CO_2. As determined before, the transport is given in the unit *tkm* and thus a result of the two parameters mass and distance. The treated mass is the same for all cases, except for Case 0. The result yields large variations in total transport and transport contribution to GWP. The changes in this impact category are due to the variations of the distances of transport in each of these cases, table 11. The results for increased distance have the larges share, as the distance varies relatively much more form case to cast than mass of the end products, figure 13. This is evident as case 2, which have the longest transport for several of the defined transports, also requires most energy. This is also causing an increased requirement for diesel, which in tor releases more CO_2. Transport is a major contributor to the impacts of biogas production, where the feedstock for the assessed cases is the main contributor. In another setting, where wet bioresidual have to transport equal or a greater distance than feedstock this transport process be the greatest GWP contributor, as earlier described in table 7, *Chapter 4.5.*

For CO_2 is the substitution critical for the total stress caused in the system. This is evident as more than 91.2% of all avoided impacts is due to potential reduction of this stressor. The origin of fossil CO_2 is the cause of these results. The biogas, causes in most of these cases reduced the consumption of fossil fuels, which is the main origin for fossil CO_2. This is not the case for either incineration or hydropower, respectively substituted by Case 2 and Case 0. For hydropower is construction the main cause of emission, which requires much concrete. For municipal incineration, large parts of the waste consist of fossil products such as plastic. Which product being substituted is essential, as presented in Appendix 8.

Biogenic CH_4 emission occur mainly during premature or post digestion of organics and during *Biogas Upgrade as described in chapter 4.7.* The results in table 5 supports these findings. This might however, be a misinterpretation as the results are in fact based on those assumptions. However, it supports that the model in SimaPro 8 works in accordance with previous findings.

In Case 0, the main process causing GWP impacts are *Manure directly applied as* fertilizer. The emissions caused by this process are indeed almost exclusively CH_4. The ratio of CH_4 produced depends on the type of storage utilized and the storage time. It is however, identified that the manure is stored over long period of time, thus releasing most of the potential CH_4. This data have been adjusted in the SimaPro 8 model to fit the findings in Amon et al. (2006) of approximately 4 kg CH_4 per ton stored manure. During storage major reduction potential have been proposed by Luostarinen et al. (2011) and Amon et al. (2006). The same is true for storage of bioresidual. The end treatment can cause an added or reduced effect based on the assumptions made from Amon et al. (2006), Appendix 330 – 332. These changes are significant, particularly for untreated manure and to a relatively lesser extent the separation of bioresidual, further explained in Appendix 6. An comparison of emission data between the model in SimaPro 8 and Amon et al. (2006) where done. To fit the data, as previously mentioned, was a 12% of remaining required, Appendix 21, parameter 12. This correlates to data found in Luostarinen et al. (2011) which operated with 10 – 30% digestion of remaining VS.

Biogas Upgrade also have a significant effect on the biogas production, both by direct emissions and by regulating the saving potential when considering system expansion. This is evident in all of the cases analysed in chapter 5, figure 7. None of the cases assessed, yields the same *Upgrade* GWP impact, which can be explained by the variation in technology, table 11. A sensitivity of *Biogas Upgrade* has been carried out in Appendix 8 and the result shows increased GWP impacts for all technologies except for *Chemical scrubbing*, relative to water scrubbing.

N_2O, the third most influential GWP stressor as found in the results, table 13. These emissions occur in relation to NH3 and N_2 generation described by Bernstad & Jansen (2011) and Amon et al. (2006). The N_2O calculation according to equation 11, and therefore vary due to the N content and the bioresidual treatment. There also several technologies that can be implemented to supress the generation of N_2O, but these have not been implemented in the current study. However, an increase in N of 20% responded the system to 3.3% increased GWP, figure .8. These impacts are not linear as an eventual decrease in N of 20% resulted in – 1.5% impacts. It is however, the treatment method of the bioresidual that mainly determines the N_2O potential, as shown in Appendix 14. In Case 4 is the total impact associated with post treatment of bioresidual the lowest. This is not only due to the reduced production of N_2O due to composting,

but it plays a significant due to its characterisation factor determined in Ecoinvent 3[14]. A further sensitivity of *Post treatment bioresidual* have been performed and the results is shown in Appendix 6.

TAP

The main contributor to this impact category is almost solely caused by NH3 formation form the bioresidual as discovered in the results. There is however, large relative variations in TAP form case to case, which has to be assessed.

NH3 occur in the same process as N_2O and is also determined by equation 11. The magnitude of its formation is much larger than for N_2O, figure 4 in the literature study. The applied treatment of the bioresidual is the main factor determining the NH3 formation potential and thus the TAP.. By decreasing the C/N ration is it possible to eliminate the NH3 formation. A mean to do this is to increase the amount of organic carbon, which can be achieved by composting. By mixing soil into the bioresidual is it in this study assumed that is gain the same characterisation as composts. This is the most effective mean to reduce the formation of NH3 and thus reduce the TAP. This have been separately assessed in Appendix 6 and is in accordance in accordance with major studies on the field, Bernstad & Jansen 2011 and Amon et al. 2006. Further study on the field is recommended however, to ensure that this in fact is the relationship.

Another factor that is of major importance for the NH3 formation is the feedstock mix and particular the N content. An increase of 20% in N causes a reflective 20.7% increase in TAP, figure 8. A separate organic waste mix sensitivity have been executed and the results are shown in Appendix 10. The results in the separate study (Appendix 10) suggest that a high level of sewage sludge, case Bergen, causes low TAP levels. This can be explained with the relatively low N content in sewage sludge, Appendix 18 parameter 174.

HTP

This impact category mainly caused by the presence of HM in the bioresidual for the biogas cases. This is a result of measured output, which is the main way of assessing the HM in Norway. Such an approach is not mass balance estimated and is in this context not representative for changes in organic waste to treatment. This is visible in figure 8 where almost no change is evident for the HM. The small amount of change is due to variations of mass of substrate, not the initial content of HM in the feedstock substrate. For Incineration is the situation another as it is determined by the combustion of meat, which where the only premade process, and have as such been applied. This highly affect the results, but would however represent the Norwegian case. To limit the HTP of anaerobic treatment is the most effective mean to apply the bioresidual for urban recreational purposes rather than agricultural purposes

[14] Earlier described as 297 times as intensive compared to CO_2

as have been applied for Case 4, figure 5. This will lead to the loss of potential GWP, TAP, HTP and FD savings because mineral P and artificial N is not substituted, figure 6.

FDP

This impact category is mainly affected by the extraction of crude oil for fossil fuel. Therefore is transport the main contributor to this impact category. Another energy source than electricity form electricity (NO) where to be applied, would the FDP drastically increase as assessed in the separate study of alternative heat source sensitivity in Appendix 5.

The part of the value chain that have the largest effect on the extraction is the system expansion of the system, particularly the cases where fossil fuels are substituted. Figure 5. However, the biogas cases yield large savings potential for the FDP category, while the reference scenario causes small contributions. This can be understood by the fact that in addition to transport requirements, requires the incineration some natural gas in its processing.

A change in transport distance causes a change in FDP, which is almost parallel to the change in GWP as earlier described. However, big changes in distance is necessary to cause any major increase in impact, but in a Norwegian context is this often the case than assessing various plants. These results are presented in the separate study executed in relation to transport sensitivity in Appendix 10. This means that the more fossil fuel that can be substituted, the more FDP savings are feasible, which is supported by the fact that changes in biogas production, as where some of the focus in the sensitivity analysis, causes a noticeable effect in FDP, figure 6.

6.2. Main findings and agreement with literature

To assess the validity of this study have it been compared to two very similar studies which assess biogas production in Norway. These are namely Lyng et al. (2011) and Modahl et al. (2014) in addition to some extent Hung & Solli 2012 for their comprehensive analysis for transport.

The main findings are that biogas production form organic waste substrates yields benefits in form of net negative impacts for both GWP and FDP, for all the assessed biogas cases. This is correlating to similar findings in the two similar studies (Lyng et al. 2011 and Modahl et al. 2014).

The first study have some instances where similar cases as assessed in this study gives very different results. A reason for this might be variations in transport distances and in particular the substrate mix. In both Lyng et al. 2011 and Modahl et al. 2014 have they applied mono digestion of the two manure types, pig and cattle, in addtion to OMW. This should, based on erlier findings in this stydy affect the overall effect of the system. In addition have, two of the cases in this study accounted for co-digestion benefits, Case 2 and 3. This should also affect the results to yield higher enevironmental benefit for this study, compared to the findings both of

the compared studies Lyng et al. (2011), which is the case and as such it is assumed that the correlation valid. The range in which the GWP impacts have been found to be is also the same in this study as for Lyng et al. (2011).

Compared to Modahl et al. (2014) is the tendancies even more convinsing, as they have also assessed TAP where the results is almost the same. They have major TAP contributions due *Post treatmetn bioresidual* wich is the exact same resuts as found in this study. However the range in which GWP results are presented deviates a lot. Despite that they, in their model have assumed a co-disgestion factor of zero, have they results that give a larger share of net impacts. This might be due to their assumption, based on Bernstad et al. (2011), that the datalevel for N_2O emissions is to low to be assessed. This is as previously mentioned, applied in this model based on data form the same source. In this study have the aim been to estimate emissions on basis of the N contained within the feedstock, and therefore have N_2O emissions been setimated od date found within this source, table 4.

6.3. Uncertainty

In this chapter is the uncertainty of the model and thesis being assessed to determine the validity of the results and the general conclusion derived later in this thesis also a foundation to determine, which parameters that should be studied and further developed.

Stochastic uncertainty – Natural variation

There are several critical sources for uncertainty applied into this model. As Carlsson & Udahl (2009) described is the methane yield a theory with no exact answer and as seen in the results (*figure 5 and 6*) have the methane production great impacts on the environmental performance of the system due to direct, indirect and the substitution benefit in the value chain. Co-digestion adds an additional layer to this uncertainty as the preferred treatment method in Norway, the direct biogas benefits associated is highly uncertain, and no literature or empirical data gives any consistent answer.

Epistemic uncertainty – Knowledge uncertainty

As described by Clavreul et al. (2012) can several sources of data cause room for uncertainty. The data in this thesis is based on several data sources, as earlier described in the *data-gathering* paragraph in *chapter 4.1.1.* Because this yields a vast degree of quantitative uncertainty for the model and thus the result for each of the cases have the sensitivity been based on a change in one parameter, both quantitatively determined by a source and qualitatively assumed.

The HM estimation is one such set of parameters that poses some level of epistemic uncertainty. The HM is measured from the bioresidual, because the utility requirements is affected only by the output concentration of HM. This yields some variation in total mass of

the HM due to fluctuations in total bioresidual. Which means that an increase in bioresidual DM gives a total increase in HM and vice versa. The sensitivity show this relationship as an increase or decrease in biogas yields slight variations in HTP, but these changes is very small. A <1,5% relative change for all sensitivities was detected which assumes that the assumption is not very critical on the total system. The uncertainty concerning to this is therefore to be interpret as low, particularly because the Norwegian average HM concentration has been found to fluctuate between class 1 and 2.

The transport for all of the cases in this analysis is fictional and is therefore highly uncertain for a Norwegian context. The data for these parameters are assumed on basis of conversation with Jørgensen (2015). This is no factual exact box, but have proven to be the most reliable source of information. The purpose of the study have however, been to assess empirical data, to gain the Norwegian context. Two sets of sensitivity have been applied for the transport as a result. The first assesses only GWP and in relation to distance and mass for various cases of end-product utility, presented in table 7. The other is a full-scale LCA sensitivity with respect to distance, presented in Appendix 7. Both sensitivity analysis show that transport have a significant effect on the overall environmental performance. Thusly, is the uncertainty for the results are significant. However, the variation in distance in Norway is significant. This, increases the validity of the results as one assumed length of transport is as valid as another, within a curtain range.

Calculative errors might have severe consequences for the validity of the results. There are many calculations in the model made in SimaPro 8. These have been occasionally tested by mass balance and secures that there is no unbalanced flows in the system. The sensitivity analysis compares the relative changes in a given parameter variation, figure 6, and gives an indicator to which parameters should be given particular attention, also for miscalculations.

The N based emissions that is described in *chapter 4.1.7* have been based on the study performed by Amon et al. (2006) and the respective shares of NH_3, N_2O and N_2 found in Bernstad & Jansen (2011). The parameters extracted from Amon et al. 2006 is based on the relationship between manure and various post treatment possibilities where anaerobic digestion is a part of this. However, it have in this study been assumed that the separation of manure and the following increase in NH3 and N_2O and the reduction in CH_4 produced, is the same as if separation where applied to the bioresidual. This is something they in their report does not guarantee, and that the applier should be careful when making such an assumption. The study performed by Amon et al. (2006) have proven to be the only source where the relationship between gaseous emissions and manure have been sufficiently describes. Therefore have the data been applied, despite the obvious uncertainty associated with this application. For NH3, N_2O and N_2 emission partitioning, presented in table data have Bernstad & Jansen (2011) been

the key source. They found however that there is a great variability in this partitioning and have compared many of the results.

Bus substitution have proven to be a very uncertain parameter when applying a change in persons per km accounted for. In SimaPro 8, by using Ecoinvent 3.0 is the only bus process alternatives given in tkm, which have proven to be not applicable a context where fuel is substituted. It have also been important to account for impacts caused by the production and use of bus as well. Based on Simonsen (2012) was it an average 12 person per km for bus trips in Norway. This compared to an assumed 10 person per km and 14,7 was the resulting sensitivity yielding severe changes in impact for all impact categories, but in particular GWP, Appendix 9.

6.4. Strengths and weaknesses of the method

The methods applied here have required extensive data gathering. This was early found discussed in Clavreul et al. (2012), particular because technical data have been targeted. The study developed here supports this statement, as the development of the LCI and the following parameter list was time consuming. The benefit of the applied method should, according to Clavreul et al. (2012), be a lower level of uncertainty. However, the level of epistemic uncertainty for several of these data have proven to be high, and therefore also affecting the uncertainty in this study.

The method applied have also reviled much information relevant for further LCA and MFA in particular for biogas production. It have also, due to the technical nature of the data applied, been necessary to develop calculation. These can, after further validation, contribute to better understanding of the various relationships of organic waste treatment and biogas production.

6.5. Strengths and weaknesses of the model

The two models made in this thesis, MFA and LCA, has the possibility to identify material and energy efficiency as well as the environmental impacts caused by the processing. The MFA model is thus the driver behind the LCA, which means that the LCA is based on a mass balanced system to secure consistency, even by changing several parameters. This method has been recommended by Schievano et al. (2011) and Clavreul et al. (2012).

By applying MFA can the system be tested for several technological and utility options, which is one of the main strengths and purposes of the model. Another feature that strengthens the model is the possibility to easily change a parameter should it be necessary for a given case. The whole system would then change accordingly, thus is the utility perspective of the model very high. And it have been aimed at modelling for both Norwegian and international purposes, both specific and general contexts.

A weakness is that the model is feedstock driven that secures material flows based on the input of substrate, but which makes it necessary to know the exact composition of the waste undergoing treatment. In some instants have assumptions been necessary to fill gaps in the literature. This increases the level of uncertainty. This weakens the environmental impact results found in this study, with respect to validity. However, the goal with this study was to identify variables affect the environmental impact tog organic waste treatment, which also have been the focus of the results and discussion in this study. Strengthening the validity of data should make the model produce much more reliable environmental results.

6.6. Implications of this work

This Study aim to highlight environmental impacts due to variations in organic waste composition, alternative and likely treatment options and possible substitution benefits of recycled energy and nutrients. This study has been developed in the context of the BIOTENMARE project at NTNU and is projected to be a contribution to the accumulation of knowledge this project represents.

The LCA model have, on basis of a MFA model, been developed to assess the implications of likely treatment methods for organic wastes and aim as such to highlight the most important variables and the following relationship to processes with various parameter compositions. In the report have several cases and scenarios been investigated and presented, figure 4 – 5 and Appendix 3 – 9, and aim to give the reader a general understanding of the leas environmental intensive treatment methods.

It is important to keep in mind that the LCI have been performed on basis of a Norwegian generic context, and the report should not be used to represent any specific plant or generic contexts in other parts of the world.

The model developed for this study, have been developed with the aim to be applied for specific as well as other generic contexts where the applier can use their set of parameters. Such an application of the model is already in effect as two other master students are currently using the developed model to assess the environmental impacts of their respective cases. It is also projected to be further developed at NTNU, both for educational purposes and for yet another study.

The results from this study, give a strong indication that anaerobic treatment is an effective mean to reduce the dependency of fossil fuel and at the same time vastly reduce the environmental impact of waste treatment, figure 4. This should be taken into consideration when future plans for organic waste treatment is investigated and projected. However, the aim of this study has been to assess variables that might affect the organic waste treatment. The

results have therefore not been aimed towards assessing the exact but the average life cycle impacts biogas situation in Norway.

Another feature of this report is that it have been developed a calculation tool to assess pasteurisation energy requirements as described in *chapter 4.6* and which is found in *annex file 3.* This tool along with developed emission and methane production calculations can be applied studies of other plants and can therefor contribute to significantly simplify the work in development of other models and assessments of the value chain.

6.7. Challenges

During the development of the model has there been several obstacles that had to be handled. Firstly, to assess the environmental impacts of a substrate, technology and a flexible end product utility system, was a MFA approach chosen as the best method. However, to develop an LCA model based on MFA principles requires a high level of detail for the data gathered as mentioned in *chapter 4.1*. Thus have theoretical technical specification data been sought. This is data that requires much work and adaptation to fit to the system description as explained in chapter 4.1.6 and 4.1.7.

Secondly, have the collection of inventory data been a time consuming activity, that due to high source variability causes increased levels of uncertainty. It have also been difficult to acquire data for HHV for each of the chose substrates, Appendix 21 parameter 194 – 207, where several have been made as assumption of other similar substrates. The general difficulty in the collection of N and P data have led to a high degree of source variety, Appendix 21 parameter 162 – 189.

Thirdly, to assume a Norwegian average have proven to be one of the most challenging tasks as there is no such thing as a general anaerobic treatment method. To handle the treatment variety have several comprehensive technology and utility sensitivity analyses been implemented, Appendix 3 – 10, and have been explained in the uncertainty chapter.

6.8. Further work

To secure that the model is up to date and can produce reliable information in the future is a further development and continuous maintenance important. The model has been developed in half a year by only one student and concerning the uncertainty should a thoroughly assessment of the model be performed before further use in a business context.

Exact MFA modelling in another data program, is recommended to assess various aspects of the biogas production. Such an assessment should be part in validating the simplified MFA made in SimaPro 8. In relation to this, should further study of NH3, N_2O and N_2 and their

respective formations for various treatment methods of bioresidual. The data found in (Amon et al. 2006) and (Bernstad & Jansen 2011) have proven sufficient to assess stressor and variable effects, for biogas production in Norway. The resulting LCA is however yielding high uncertainty due to this. The level of agreement with literature, particularly (Modahl et al. 2014), counter weights much of these uncertainties, but still further study would possible further confirm or disprove these results.

As described in *Implication of this work* the model is to be applied for two parallel ongoing studies. Both assesses various aspects of biogas production, both for a Norwegian specific case and in a more international setting. Experiences from these students should be taken into consideration for further development of the model.

7. Conclusion

"Which stressors are critically influencing the environmental life cycle impact of the biogas production in Norway, in comparison to the alternative organic waste treatment option, and which factors and variables limits or enhance these"

The main critical stressors and the relating variables in an LCA perspective are:

- The formation of fossil CO_2 from transport
- N_2O, NH_3 and CH_4 from manure or bioresidual and the respective treatment method.
- Biogenic CH_4 released during biogas upgrade and thus the applied upgrade technology
- Zinc, cadmium and lead contained within the bioresidual and is application purpose
- Extraction of crude oil for fossil fuel to transport or other energy requirements

Applying a good substrate mix and applying pasteurization, as a pre-treatment, is a small co-digestion benefits a fair assumption. Co- digestion can therefore result in a higher biogas yield in form of degrability in addition to a stable and self-sustained process. By securing zero leakage form digester and apply chemical scrubber as the upgrading technology, would reduce direct GWP and increase the energy output in form of biomethane applicable for grid or fuel purposes. Composting biofertilizer would achieve the greatest decrease in impact for all impact categories. However, it is further necessary to study the consequences of adding soil to the bioresidual and determine the NH_3, N_2O and N_2 relationship more closely.

The biogas production in Norway at current stage, with co-digestion, can reduce the total impact for GWP and FDP substantially. By substituting fossil fuels of today or substitute remote heating municipal incineration, biogas production proves to be a good solution. Application of biogas and biofertilizer production to reduce GWP impacts and FDP would also prove an efficient mean, without being seen in reference to other organic waste treatment systems.

To apply the basic MFA methodology have proven a good tool to assess various value chain aspects of organic waste substrates and should be further developed and assessed in relation with LCA studies to generate more reliable results. The GWP and TAP results from this study have been in accordance with previous studies even though the applied LCA assessment method have differentiated.

The developed SimaPro 8 model have proved to be a good tool to assess environmental impacts and have worked well by applying MFA principles in the assessment. A complete MFA procedure have proven difficult however when applying only this program. An alternative separate tool is therefore recommended for such studies.

The results from this study is deemed reliable as an assessment of various variables and their effect on the overall Norwegian biogas production and organic waste treatment.

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Appendix 1 – Master thesis contract

NTNU

Norwegian University
of Science and Technology

Department of Energy
and Process Engineering

EPT-M-2014-79

MASTER THESIS

for

Student Simon Aleksander Saxegård

Spring 2015

Life cycle assessment of biogas production from organic waste sources in a Norwegian context

Livsløpsanalyse av biogassproduksjon fra organiske avfallsressurser i en norsk kontekst

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 carry out a life cycle assessment (LCA) 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 Norway.

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.
2) 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 in Norway, compared to alternative waste treatment methods.
3) Further development of the LCA model made in the Master Project to be suitable for such an analysis.
4) 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 in Norway. Document own assumptions and sources for your input variables and choices,
5) Calculate the potential life cycle environmental impacts of the system, and perform a sensitivity analysis of your system.
6) 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, and that they are presented in tabular and/or graphic form in a clear manner, and that they are analyzed 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.

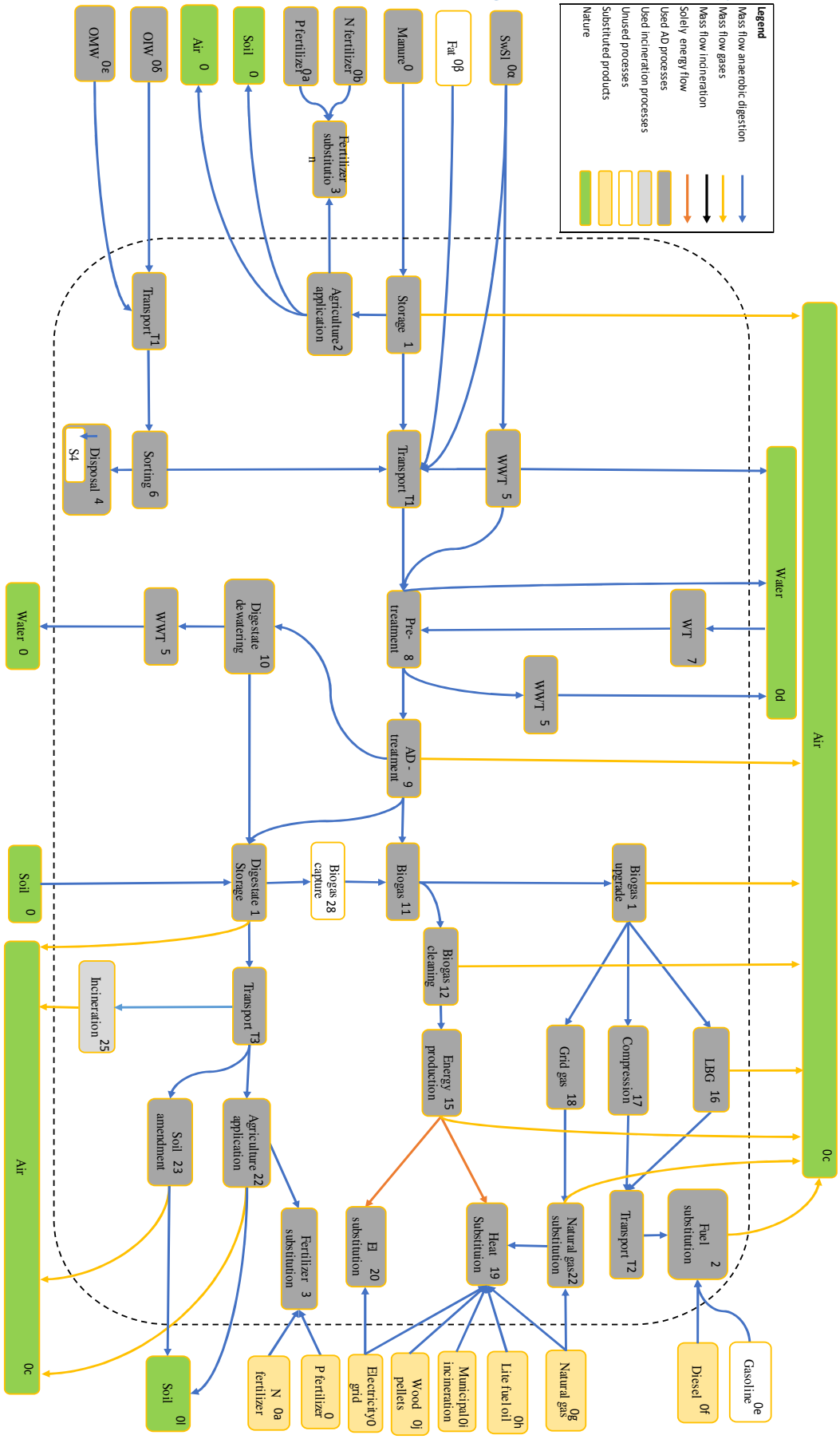
☐ Work to be done in lab (Water power lab, Fluids engineering lab, Thermal engineering lab)
☐ Field work

Department of Energy and Process Engineering, 14. January 2014

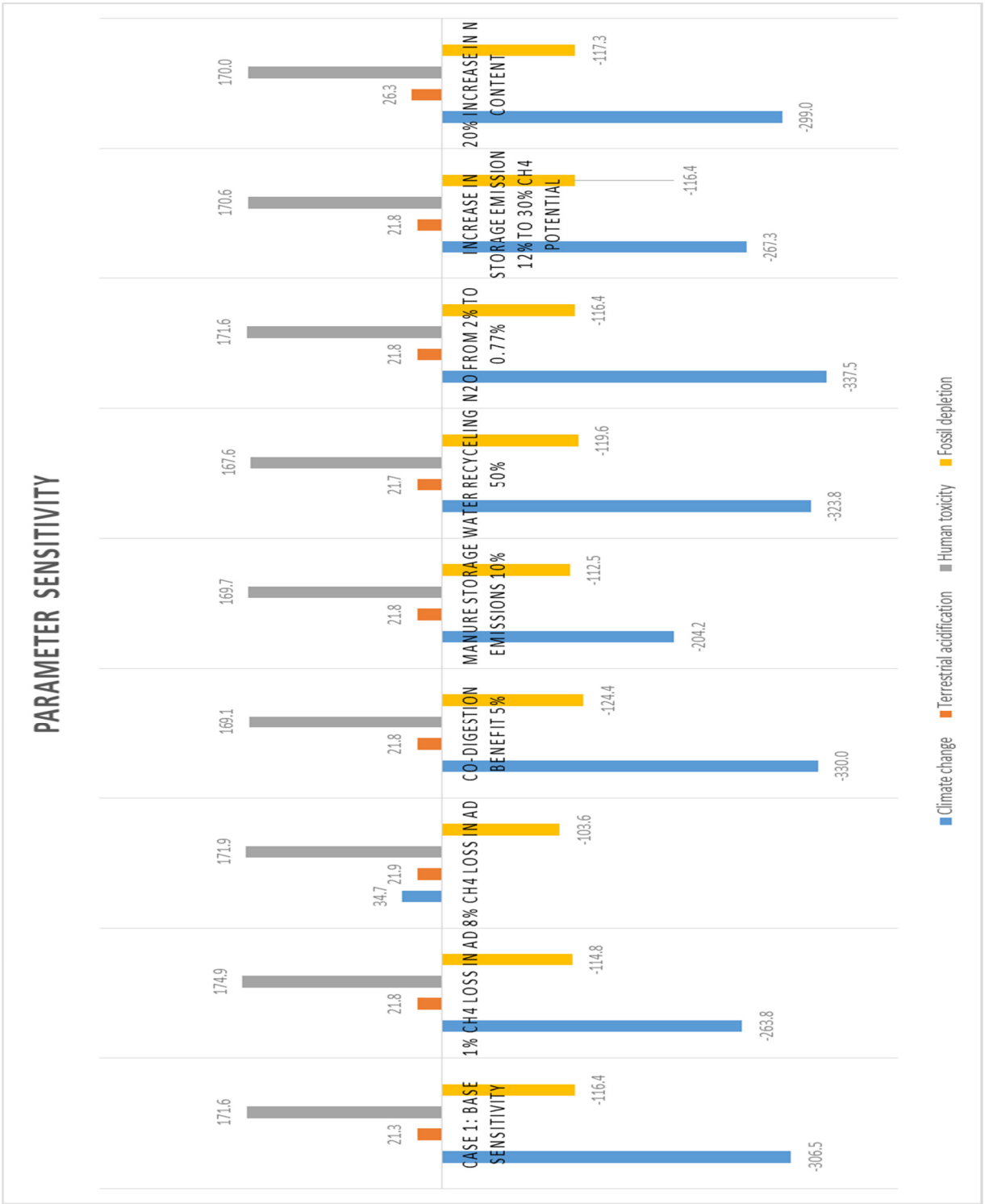
Olav Bolland
Department Head

Helge Brattebø
Academic Supervisor

Appendix 2 – Value chain for anaerobic digestion in Norway

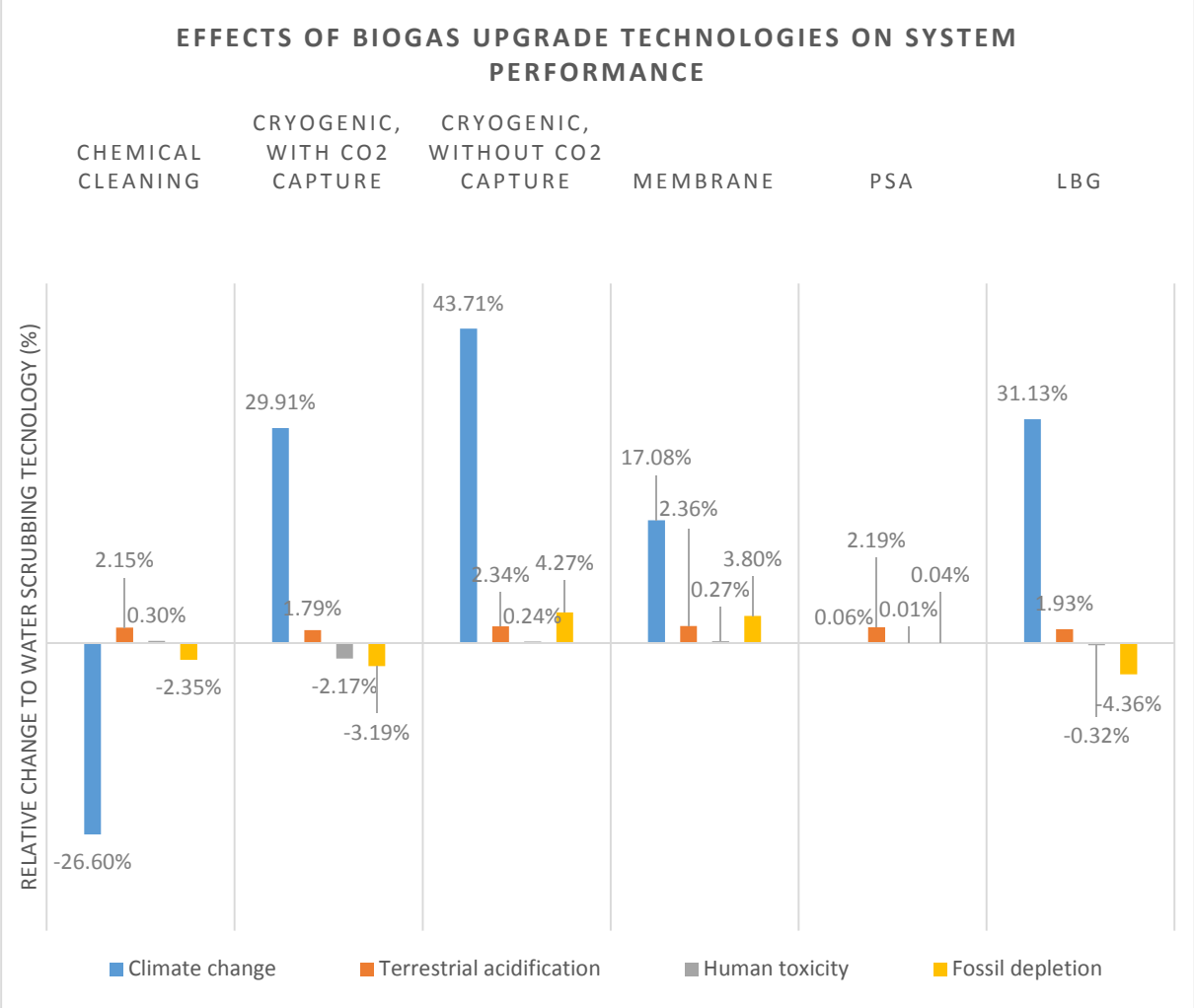


Appendix 3 – Total impact results for sensitivity



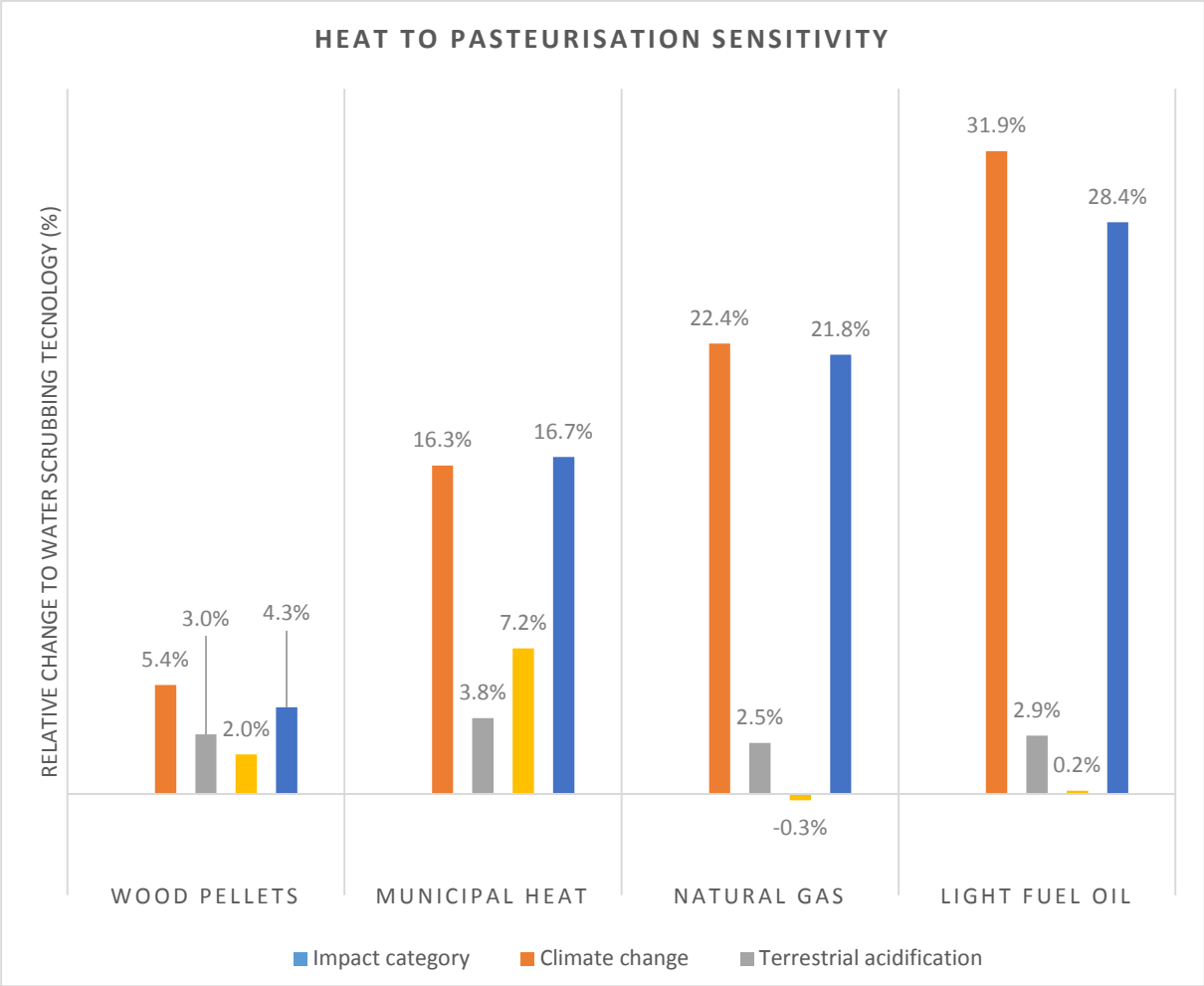
Climatechange is in kg CO_2-eq, Terrestial acidification kg SO2-eq, Human toxicity kg 1,4-DB eq and Fosil depletion kg oil-eq

Appendix 4 – Sensitivity of different biogas upgrade alternatives



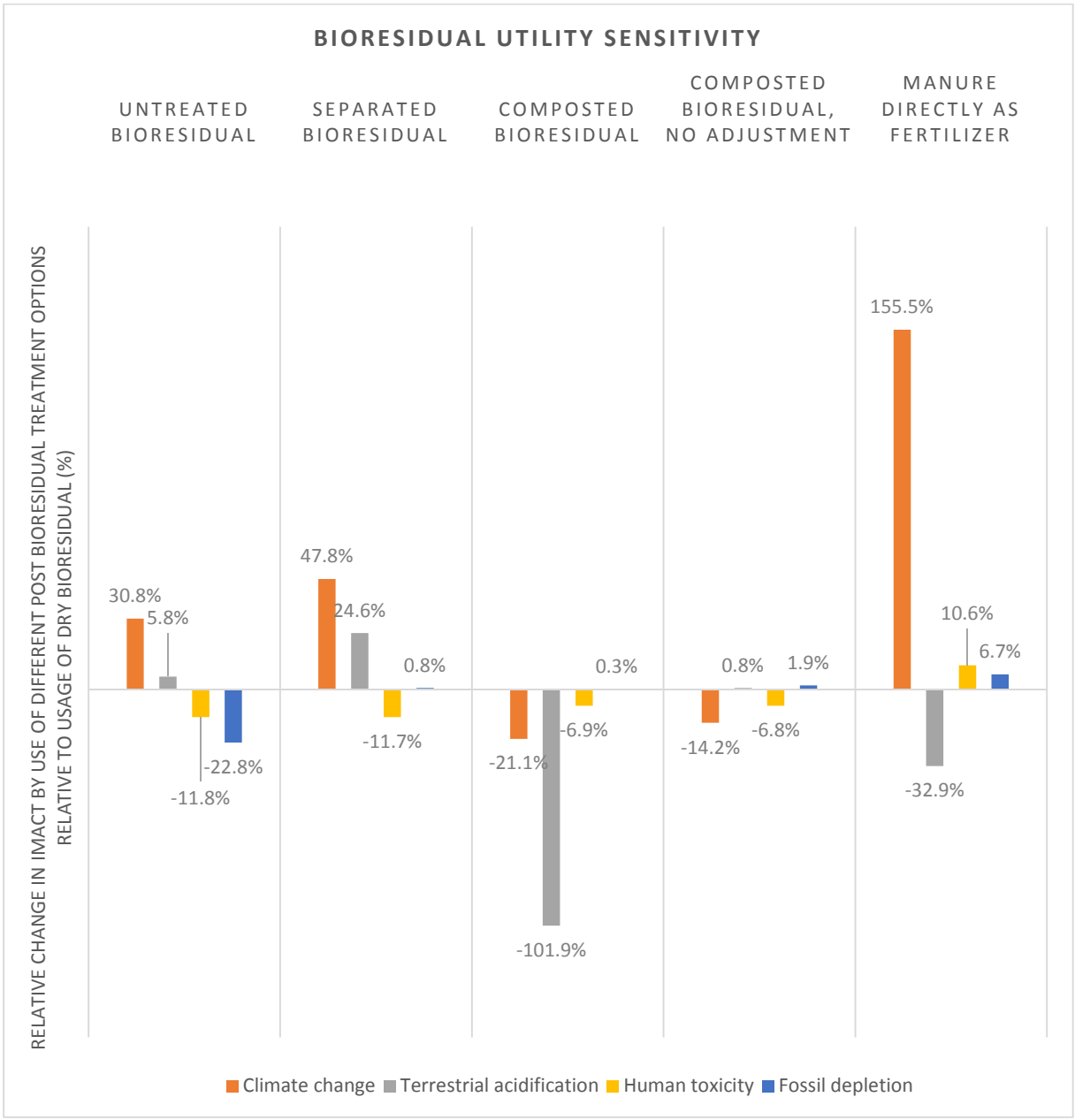
The relative change per impact category due to a change in biogas upgrade technology with the following parameter composition. Chemical cleaning results in the least GWP related impacts and is also the upgrade technology that have the least CH_4 leakage emissions, table 9. Cryogenic cleaning without industrial CO^2 production from captured yields the largest impact increase for al impact categories. The impacts is severely reduced by utilising the CO_2 (maximum 25% of the separated CO_2 for the biogas) for GWP, but is still the cleaning method for biomethane that yields the highest impacts for relative to all of the cleaning technologies. PSA is the technology that generates the least relative change in impacts in comparison with the reference technology, water scrubbing.

Appendix 5 – Pasteurisation heat energy sensitivity



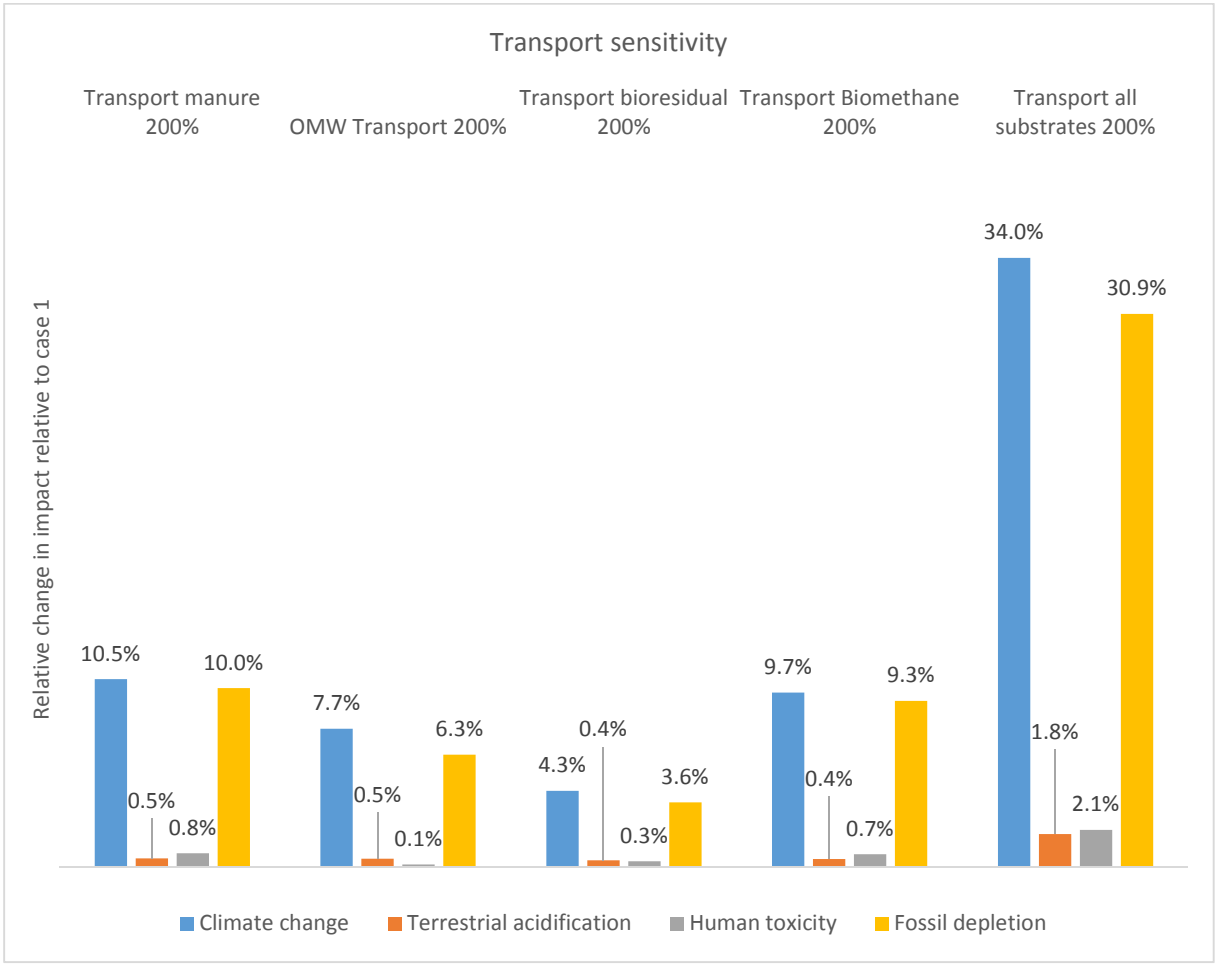
Various sources for heating energy to pasteurisation in comparison to the reference heat source, Norwegian electricity mix, medium voltage. Norwegian electricity mix yields the least impact for all impact categories compared to all alternative heat sources, except for HTP by applying natural gas. Biomass burned from wood pellets, yields the least increase in GWP and FDP relative to Norwegian electricity mix.

Appendix 6 – Sensitivity of different post treatment alternatives for bioresidual



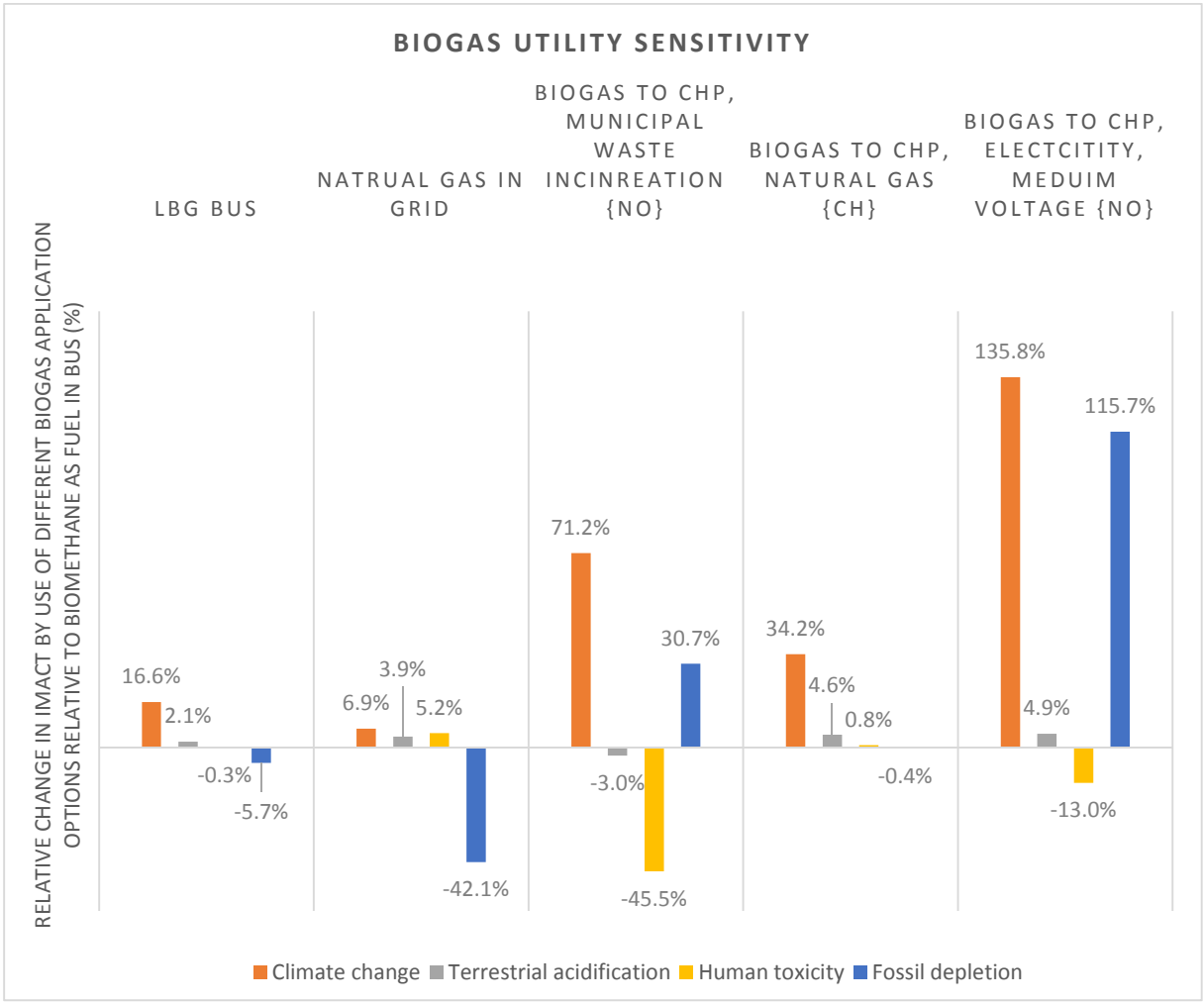
Various post treatment methods for bioresidual and the associated change in impact relative to the Norwegian common treatment alternative, applied dry bioresidual. Composting by addition of soil to bioresidual and dewatering of excess water relative to water content of compost (30%), where water is treated as waste water, yields the largest reduction in both GWP and TAP category.

Appendix 7 – Sensitivity of transport based on a 200% increase in distance for a single substrates and for the whole system, separately.



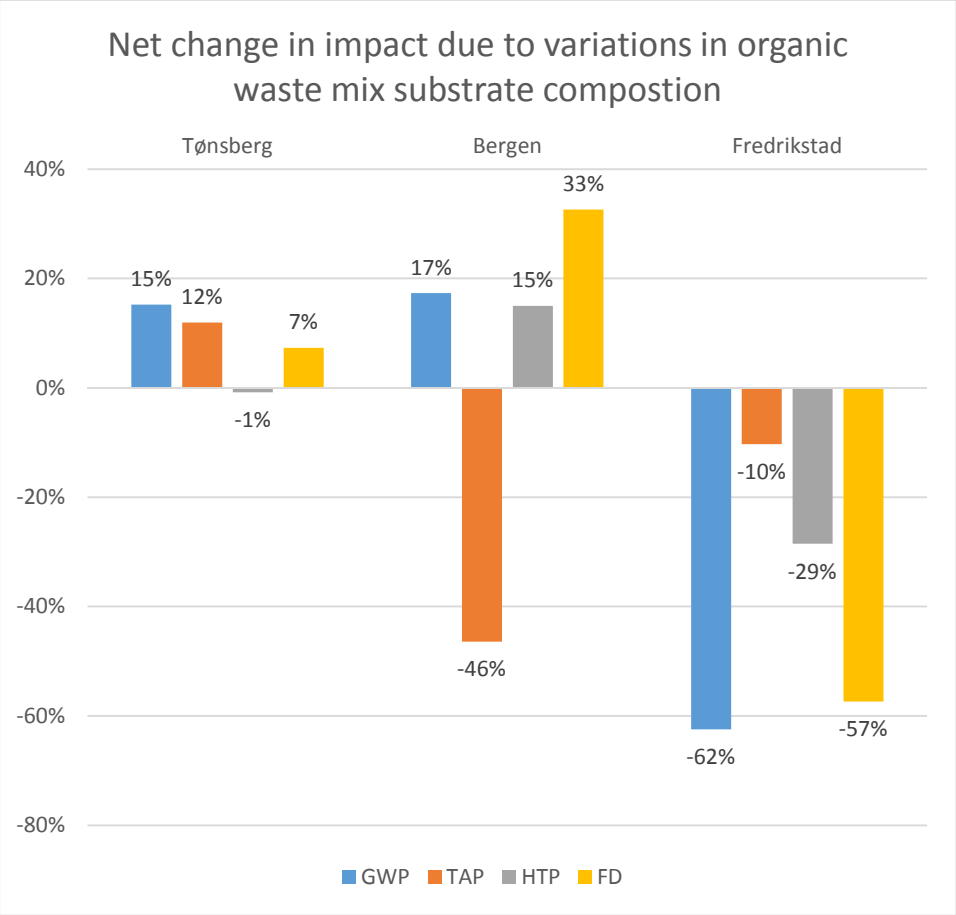
The reference transport distance is as shown in table 6 and the sensitivity has been applied for each of the most prominent transport routes. The change, for only one commodity, that yields the highest increase in impact is manure, closely followed by transport of biomethane. If all transport distances for all commodities in the model where to be increased with 200% is a 34% increase in the GWP impact category evident. Still for 200% increase of all substrates would the GWP and FDP, which is most affected by the change in distance, result in -202 kg CO_2 eq and -80 kg oil eq respectively.

Appendix 8 – Sensitivity of biogas utility options



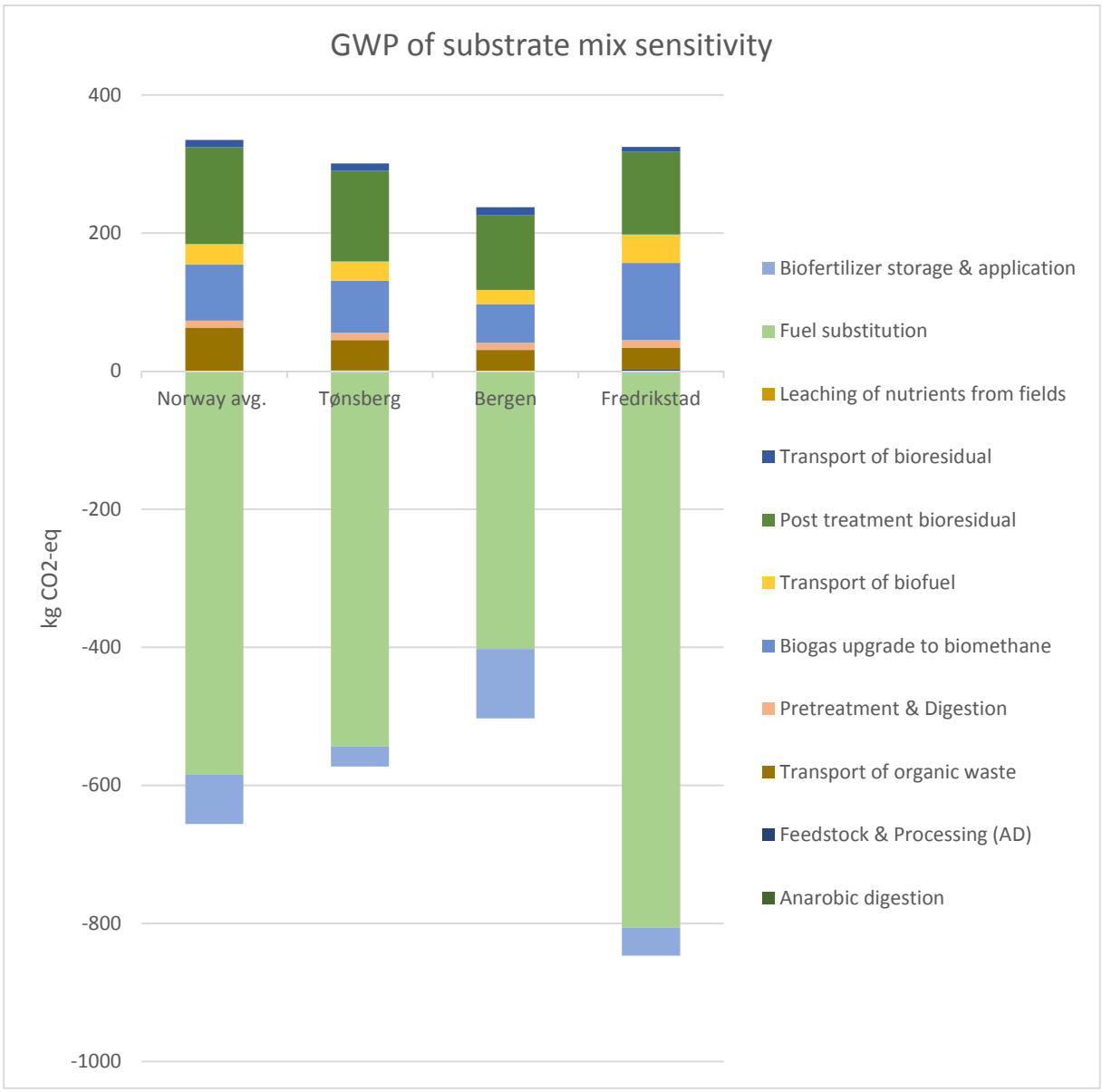
The usage of biogas to substitute natural gas in grid is the second most preferable usage if reduction of GWP is the target and the best application if FDP reduction is the target. For reduction in HTP is CHP the preferred usage biogas if municipal incineration or Norwegian electricity mix is applied.

Appendix 9 – Substrate sensitivity

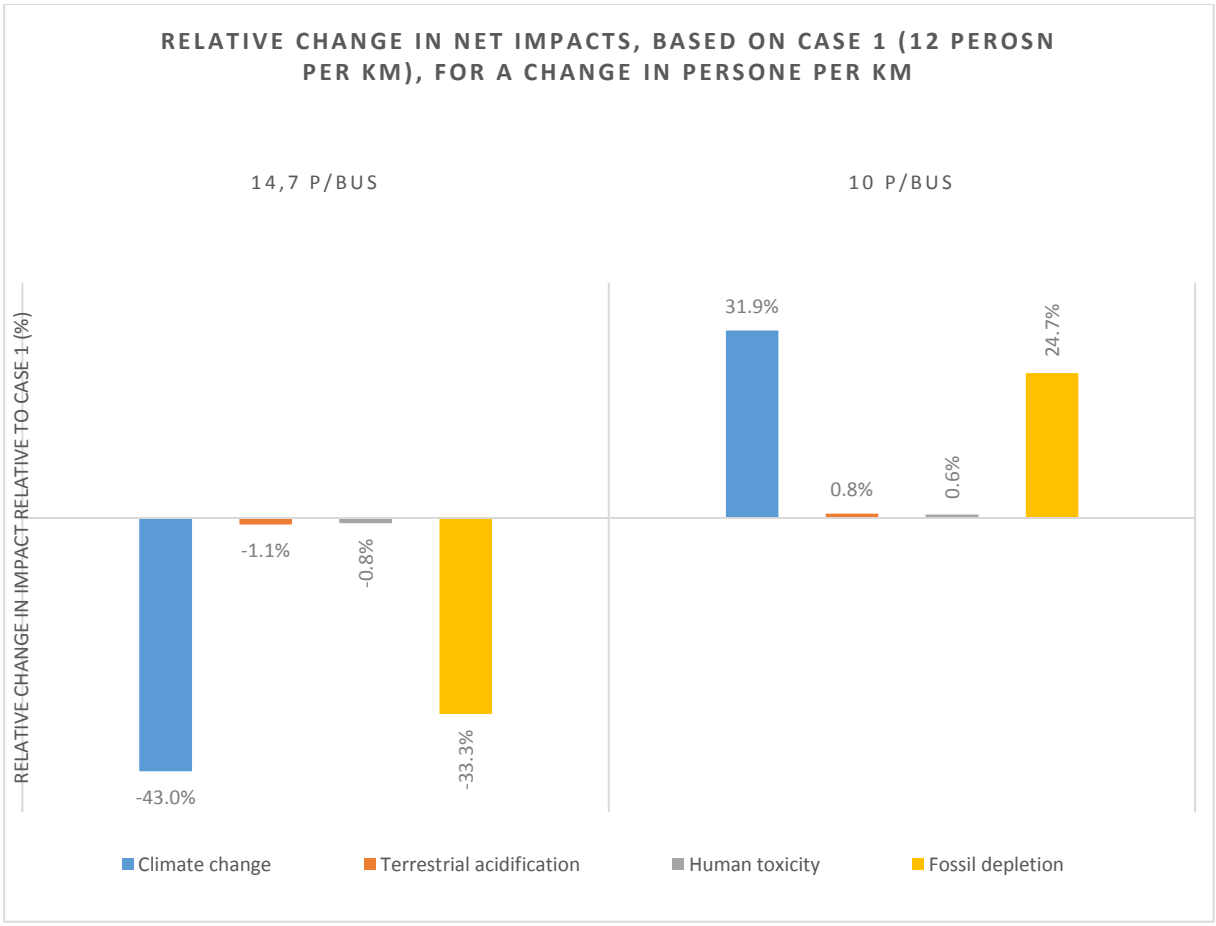


Fossil CO_2, biogenic CH_4 and N_2O all contributes to large shares of GWP. CO_2 is the gas that is also avoided the most by applying biogas for energy purposes, particular to substitute fuel or natural gas. Biogenic CH_4 is the most sensitive variable for the whole system with respect to GWP. An increased loss in upgrade or by digestion means less biogas leaves the system, and thus is the direct impact increased due to its characterisation factor and less methane available for application. Increased formation of N_2O can drastically alter the GWP impact and at the same time reduce the N content found in the bioresidual. NH_3 is mainly responsible for TAP and causes losses in N contained within the bioresidual. The treatment of the bioresidual is the factor that affect NH_3 formation the most and thus the TAP. Compost have proven to be the treatment that yields least kg NH_3, but a further assessment of the relationship between NH_3 formation and soil additive for bioresidual is highly recommended. The level of HM in bioresidual can restrict the application to only be applied for recreational purposes. By applying bioresidual as biofertilizer can substantial parts of N and P be recovered, but determined by the level of HM is a high level of HTP identified. The extraction of crude oil for transport purposes in particular the transport of Organic waste have been identified as the essential FDP stressor. By reducing transport distance or increasing biogas production, where the biogas is to substitute fossil fuels, can this category be limited the most.

Appendix 10 – Substrate sensitivity GWP results



Appendix 11 – Bus sensitivity due to a change in persons per km



Appendix 12 – Nitrogen loss during storage and application of bioresidual and manure

Applied	Nitrogen loss due to gaseous escape	N-loss ratio	Main source	Sub source
	Digested, organic waste	67%	Bernstad & Jansen 2011, Table 5	Chung (2007)
	Digested, organic waste	30%	Bernstad & Jansen 2011, Table 5	Measurement
X[1]	Digested, manure	21.8%	Amon et al 2006 table 2	Measurement
X[2]	Digested, organic waste	17.0%	Bernstad & Jansen 2011, Table 5	Sonesson (1996)
X[3]	Undigested, manure	17.9%	Amon et al 2006, table 2	Measurement

[1]Found in line 302 in appendix 17, [2]Found in line 300 in appendix 17, [3]Found in line 301 in appendix 17

Appendix 13 – NH_3, N_2O and N_2 gas formation table REMOVE!!!

Treatment type	NH_3	N_2O	N_2	Sources in paper
AD	96%[1]	0.77%[1]	3.23%[2]	Chung (2007)[3]
Composting	2.40%	1.40%	96.20%	Sonesson (1996)[3]

[1]Found in line 298 and 299 for NH_3 and N_2O respectively. [2] N_2 is found by 100% - NH3% - N_2O. [3] None of these sources where found, and therefore have the study that this is gathered from the main source. Source: Appendix 11 is just a short summary of table 5 in the paper (Bernstad & Jansen 2011), pg 1883.

Appendix 14 – Net total NH_3 emissions during storage and after field application

NH_3 cattle manure emissions	Untreated[1]	Separated[1]	Digested[1]	Straw cover[2]	Aerated[2]
Storage	41.0	39.3	9.9	50.9	208.6
Storage solid fraction		287.8			
Field application	185.8	75.8	220.0	269.5	214.0
Total emissions	226.8	402.9	229.9	320.4	422.6
% storage	18%	81%	4%	16%	49%
% application	82%	19%	96%	84%	51%

[1]Applied in the model [2]Excluded and not mentioned in the model or paper, out of scope at current level. Source: (Amon et al. 2006)

Appendix 15 - Net total CH_4 emissions during storage and after field application

CH_4 cattle manure emissions	Untreated	Separated	Digested	Straw cover	Aerated
Storage	4054.7	1833.0	1342.6	4904.1	1731.7
Storage solid fraction		510.6			
Field application	1.3	19.7	2.0	22.2	7.6
Total emissions	4047.0	2363.3	1344.6	4926.3	1739.3
% storage	1.0	1.0	1.0	1.0	1.0
% application	0.0	0	0	0.0	0.0

[1]Applied in the model [2]Excluded and not mentioned in the model or paper, out of scope at current level. Source: (Amon et al. 2006)

Appendix 16 - Net total N_2O emissions during storage and after field application

N_2O cattle manure emissions	Untreated[1]	Separated[1]	Digested[1]	Straw cover[2]	Aerated[2]
Storage	20.2	9.0	28.5	42.2	49.3
Storage solid fraction		13.2			
Field application	3.8	6.4	2.7	10.3	4.9
Total emissions	24.0	28.6	31.2	52.5	54.2
% storage	84%	78%	91%	80%	91%
% application	16%	22%	9%	20%	9%

[1]Applied in the model [2] Excluded and not mentioned in the model or paper, out of scope at current level. Source: (Amon et al. 2006)

Appendix 17 –Process data for electricity requirement

Process data	Crusher[1]	Sorting[1]	Pasteurization	AD	Total	Unit	Use (h)	Use (d)	Use year	Unit
Pump 1	7.5	45	25	30	107.5	kWh	8	5	2080	h
Pump 2	7.5				7.5	kWh	8	5	2080	h
Total	15	45	25	30	115	kWh	8	5	2080	h
Use per year	31200	93600	52000	62400	239200	kWh				
Ton processed	30000	30000	30000	30000	30000	Ton				
Use per ton wet	1.0	3.1	1.7	2.1	8.0	kWh / ton				
Energy per process		4.16[1]	1.73	2.08		kWh / ton wet				

[1]Crushing is aggregated into sorting. Assumed average process data for the biogas plant at FREVAR, (Jørgensen 2015).

Appendix 18 – Pasteurisation calculator

Pasteurisation	Parameter values	Sources	Temperature in, °C	Sources	Temperature out, °C	Sources	Units
Specific heat capacity kWh/ ton/℃	1.1704						kWh/ton/°C
Specific heat capacity kJ/ kg/℃	4.18	Coultry et al. 2013					kJ/kg/°C
Heating of Feedstock (℃)	56	Frevar 2015	14[1]	Hamelin et al. 2014, Bauer et al 2013, Frevar 2015	70[2]	Morken et al 2008	°C
Heating actual (℃)	26		44[3]	Ecopro (120 C à 155C)	70[2]		
Input mass	1.00[10]						ton
Loss rate Heating Pretreatment	5%[4]	Hamelin et al. 2014					%
Sum Pretreatment kWh/ton	68.82[7]						kWh / ton
Actually energy to heating	31.95[9]						
Measured values							
Recapture Celsius (℃)	31.5		70[2]	Morken et al 2008	38.5[6]	Hamelin et al. 2014, Bauer et al 2013	°C
Output mass (ton)	1.00						ton
Loss rate Heating	0%[5]	Hamelin et al. 2014					%
Sum energy recapture kWh/ ton	36.868[8]						kWh/ ton °C

Calculator description:

[1]Natural actual average feedstock temperature, [2] Pasteurisation temperature for the given system, [3] The measured input temperature from witch the actual heating does start after heat capture, [4] Recovery loss, [5]Heat loss from reuse of heat. [6] Temperature to digester (36 – 38 ℃ mesophilic and 48 – 52 ℃ thermophilic) adjust ([3]) accordingly to find the approximate ℃ in ([6]), [7] theoretical energy requirement before heat recapture, [8] recaptured heat in term of energy (kWh), [9] actual heat requirement after heat recovery (kWh), [10] the total amount pasteurized in terms of wet weight. NOTE: This calculator does not account for the lower heating value of organic substrates, as it for this study only account for 10% of the total pasteurised mass. Green fields is where values is changeable.

Appendix 19 – The process energy requirements for each case.

Process	**Process energy**	Incineration Case 01	AD Case 1	AD Case 2	AD Case 3	AD Case 4	Units
P1	Storage manure	0	0	0	0	0	kWh / FU
P2a	Agriculture application	4.6	0	0	0	0	kWh / FU
P3	Fertilizer substitution	0	0	0	0	0	kWh / FU
P4	Disposal	0	0	0	0	0	kWh / FU
P5	Electricity WWT	0.17	3	3	0	1	kWh / FU
P6	Sorting	0	2.7	2.7	2.7	2.7	kWh / FU
P7	Water treatment	0	1.1	1.2	1.1	1.1	kWh / FU
P8	Heat pretreatment Original	0	2460	5970	2460	2900	MJ/ FU
P8	Heat pretreatment actual	0	1140	1320	1670	1580	MJ/ FU
P8	Electricity pretreatment	0	20	17.3	17.3	17.3	kWh / FU
P9	Electricity AD	0	20.8	20.8	20.8	20.8	kWh / FU
P10	Electricity Dewatering	1.0	18.6	18.6	0	8.1	kWh / FU
P11	Biogas	0	0	0	0	0	kWh / FU
P12	Biogas cleaning	0	0	0	0	0	kWh / FU
P13	Electricity biogas upgrade (UpT)	0	80.3	0	46.2	87.2	kWh / FU
P13	Heat biogas upgrade (UpT)	0	0	0	193	0	MJ / FU
P14	Bioresidual storage	0	0	0	0	0	kWh / FU
P15	Electricity CHP	0	0	105.3	0	0	kWh / FU
P16	Electricity Compression	0	49.5	0	0	37.7	kWh / FU
P17	Electricity LBG	0	0.0	0	310	0.0	kWh / FU
P18	Grid gas	0	0	0	0	0	kWh / FU
P19	Heat substitution	0	0	0	0	0	kWh / FU
P20	Electricity substitution	0	0	0	0	0	kWh / FU
P21	Electricity tanking	0	37.7	0	0	0	kWh / FU
P2b or P23	Electrisity spreading	0	0	0	7.7	5	kWh / FU
P25	Electericity incineration	110	0	0	0	0	kWh / FU
	Total electericity	115.8	234.0	169.2	405.7	181.2	kWh / FU
	Total heat	0	1140.0	1320.0	1863.0	1580.0	MJ / FU
	Total energy consumption	416.7	1982.3	1929.1	3323.6	2232.2	MJ / FU

Appendix 20 - Main mass flows (simplified)

Inventory results, MFA (simplified)

	Anaerobic treatment	Mass flow	Case 1	Case 2	Case 3	Case 4	
1	Feedstock & Processing						Unit
	WWT, SwSl	X0α,5	0.282	0.12	0.282	0.282	ton / FU
	Storage manure	X0γ,8	5.76	5.76	5.76	5.76	ton / FU
	Premature biogas	X1,0c	0	0	0	0	kg / FU
	Manure directly applied as fertilizer (MDF)	X1,2	0	0	0	0	ton / FU
	Manure fertilizer and artifical fertilizer	X,2a,3	0	0	0	0	ton / FU
	Biofertilizer (manure)	X2a,0l	0	0	0	0	ton / FU
	Gases from land application	X2a,0c	0	0	0	0	ton / FU
	Artifichal N fertilizer	X3,0a	0	0	0	0	kg / FU
	Mineral P fertilizer as P_2O_5	X3,0b	0	0	0	0	kg / FU
	Sorting, OIW	X0δ,6	0.573	0.573	0.573	0.573	ton / FU
	Sorting, OMW	Xε,8	1.33	1.33	1.33	1.33	ton / FU
	Disposal inorganics	X6,4	0.509	0.509	0.509	0.509	ton / FU
	Landfill waste stock	X4,S4	0.509	0.509	0.509	0.509	ton / FU
2	Pretreatment & Digestion		1				
	Pretreatment, SwSl	X5,8	0.282	0.12	0.282	0.282	ton/ FU
	Pretreatment, Fat	X0β,8	0	0	0	0	ton/ FU
	Pretreatment, Manure	X1,8	5.76	5.76	5.76	5.76	ton/ FU
	Pretreatment, OMW,OIW	X6,8	1.397	1.397	1.397	1.397	ton/ FU
	Water, produced	0d,7	2.56	2.72	2.56	2.56	ton/ FU
	Water to pretreatment	X7,8	2.56	2.72	2.56	2.56	ton/ FU
	Waste water from pretreatment	X8,5	0	0	0	0	ton/ FU
	AD	X8,9	10	10	10	10	ton/ FU
	Biogas lost	X9,0c	0	0	0	0	kg / FU
3	Post-treatment Bg		1				
	Bg raw	X9,11	402	422	406	402	kg / FU
	Bg raw	X11,12	0	422	0	0	kg / FU
	CO2 and CH4 losses / Torch	X12,0c	0	9.51	0	0	kg / FU
	CHP, biogas	X12,15	0	399	0	0	kg / FU
	BG raw	X11,13	402	0	406	402	kg / FU
	CO2 removal and CH4 lost	X13,0c	231	23	234	231	kg / FU
	LBG	X13,16	0	0	172	0	kg / FU
	Compression	X13,17	171	0	0	171	kg / FU
	Gas to Grid	X13,18	0	0	0	171	kg / FU
	Biomethane to fuel	X17,21	171	0	0	0	kg / FU
	LBG fuel	X16,21	0	0	172	0	kg / FU
4	Post treatment bioresidual						
	Raw bioresidual	X9,10	9.59	9.44	9.44	9.43	ton / FU
	Post treated bioresidual	X10,14	1.84	1.78	9.44	6	ton / FU
	WWT, liquid biowaste	X10,5	7.74	7.66	0	3.43	ton / FU
	WWT, liquid biowaste	X5,0d	7.74	7.66	0	3.43	ton / FU
	Uncontaminated soil	X0l,14	0	0	0	0.7	ton / FU
	Biofertilizer	X14,2b	1.84	1.78	9.44	0	ton / FU
	Bioresidual	X14,23	0	0	0	6	ton / FU
	Gassous lossess from bioresidual storage	X14,0c	17.16	16.1	17.96	17.5	kg / FU
	Biofertilizer	X2b,0l	1.84	1.78	9.44	0	ton / FU
	Bioresidual	X23,0l	0	0	0	6	ton / FU
	Artifichal N fertilizer	X0a,3	7.23	7.59	32.9	0	ton / FU
	Mineral P fertilizer as P	X0b,3	5.49	6.09	7.767	0	kg / FU
	Gases	X23,0c	0	0	0	0	kg / FU

Appendix 21 – Inventory parameter list for case 1 and model functions from LCA model developed in Simapro 8

No.	Input parameters		
1	DM_AD	1	Functional unit, should always contain a value >0
2	Incineration	0	Applied when Incineration case is being applied to extract the correct MFA results found below in the calculated parameters!
3	OWc_AD	0.9	insert numeric % of the optimal water content in the anaerobic digester
4	OWc_I	0.6	insert numeric % optimal water content in incinerated organic waste; Ecoinvent 2.2
5	Recycle_W_AD	0	insert numeric % of water reused in the anaerobic digester from the dried digestate, if dewatered; insert value (numeric %)
6	Manure	0.5	insert numeric % manure of the total DM of organic waste
7	Fat	0	insert numeric % Fat of the total DM of organic waste
8	OMW	0.131	insert numeric % Organic Municipal Waste of the total DM of organic waste
9	OIW	0.321	insert numeric % Organic Industrial Waste of the total DM of organic waste
10	SwSl	0.048	insert numeric % Sewage sludge of the total DM of organic waste
11	ProductX	0	insert numeric % self adjusted product of the total DM of organic waste

12	AD_SL	0.12	numeric % of CH4 produced from the total remaining biogas potential in the bioresidual; Amon et al 2006, derived from digestate emission / by methane yield for cattle and is given in numeric % of this actual loss
13	Codigestion_Benefit	1	Numeric % of the co-digestion methane yield benefit; Lyng et al. 2011, Ariunbaatar et al 2014 (+11.9% at 155C Pt); 1=100= no change [>1= increase, (=1)=no change, <1= negative change]
14	N_Adjustment	1	Numeric % change to find N sensitivity ; 1=100= no change [>1= increase, (=1)=no change, <1= negative change]
15	Optic_sorting	0	Optical sorting technology; Turn on by applying 1, then turn off the other by giving them value 0
16	Inorganic_OMW	0.15	numeric % of indigestible material (waste) that is sorted out from Organic municipal waste and sent to incineration; Jørgensen (2015)
17	Inorganic_OIW	0.25	numeric % indigestible material (waste) that is sorted from Organic industrial waste and sent to incineration; Jørgensen (2015)
18	Lr_ADg	0	numeric % loss of methane (CH4) in the anaerobic digester
19	Lr_Sorting	0	numeric % during sorting OMW and OIW to remove inorganic waste such as plastic, metals, sand etc. Jøregnesen (2015)

20	Lr_LBG	0.018	numeric % loss of methane by converting biomethane to LBG; Bauer et al. (2013)
21	Lr_DW	0.05	numeric % of DM lost to the dewatered reaction after dewatering of bioresidual; Jørgensen (2015)
22	N_DM	0.1	Numeric % of N that is found in the solid fraction of the bioresidual; Poeschl et al 2012a: Helm, 2010
23	P_DM	0.7	Numeric % of Phosphorus that is found in the solid fraction of the bioresidual; Poeschl et al 2012a: Amon et al. 2007
24	NH3_Inhi	0	Implementation of NH3 inhibition in storage tanks for bioresidual; Turn on by applying 1, then turn off the other by giving them value 0; Gjødselsforskriften §10
25	NH3_Red	0.65	numeric % of the NH3 inhibition in storage tanks for bioresidual, Luostarinen et al. (2011)
26	ST	0	Tight storage technology for bioresidual, only CH4 inhibiting, se Lr_ST for reduction value; Turn on by applying 1, then turn off the other by giving them value 0; Luostarinen et al. 2011
27	SC	0	Cover over storage technology for bioresidual, only CH4 inhibiting, se Lr_SC for reduction value; Turn on by applying 1, then turn off the other by giving them value 0; Luostarinen et al. 2011

28	Lr_SC	0.65	Numeric (%) loss reduction Storage cover ; Luostarinen et al. 2011
29	Lr_ST	0	Numeric (%) emission from storage tight ; Luostarinen et al. 2011
130	Storage_Emission_BD	0	numeric % of the methane yield in manure pre digested during post storage of manure; insert value (Numeric %)
31	Sp_Bat	0	Best available technology for NH3 emission reduction during spreading. Turn on by applying 1, then turn off the other by giving them value 0; Luostarinen et al. 2011
32	Spr_NH3	0	numeric % reduced NH3 emission when applying best available technology (BAT); Loustarinen et al. 2011
33	MDF	0	Manure applied directly as fertilizer; Turn on by applying 1, then turn off the other by giving them value 0
34	Digestate_Use_Wet	0	Untreated bioresidual containing both processing water and remaining inorganics and undigested VS; Turn on by applying 1, then turn off the other by giving them value 0
35	Digestate_Use_Dry	1	Dewatered bioresidual containing a share of process water, inorganics and VS. Dewatered water to WWT; Turn on by applying 1, then turn off the other by giving them value 0
36	Digestate_Use_Separated	0	Dewatered bioresidual containing a share of process water, inorganics and VS. Dewatered water is also applied for fertilization purposes; Turn on by

			applying 1, then turn off the other by giving them value 0
37	Digestate_Use_Compost	0	Composted bioresidual by addition of uncontaminated soil and dewatered to fit the optimal water content for compost (see: Compost_Soil); Turn on by applying 1, then turn off the other by giving them value 0
38	Biomethane_Use_Grid	0	Biomethane applied in a gas grid system; Turn on by applying 1, then turn off the other by giving them value 0
39	Biomethane_Use_Bus	1	Biomethane applied as bus fuel, substituting diesel; Turn on by applying 1, then turn off the other by giving them value 0
40	Biomethane_Use_gasoline_car	0	Biomethane applied as car fuel, substituting gasoline; Turn on by applying 1, then turn off the other by giving them value 0
41	Biomethane_Use_diesel_car	0	Biomethane applied as car fuel, substituting diesel; Turn on by applying 1, then turn off the other by giving them value 0
42	LBG_Use_Bus	0	Liquid biomethane applied as bus fuel, substituting diesel; Turn on by applying 1, then turn off the other by giving them value 0
43	LBG_Use_Gasoline_car	0	Liquid biomethane applied as car fuel, substituting gasoline; Turn on by applying 1, then turn off the other by giving them value 0
44	LBG_Use_diesel_car	0	Liquid biomethane applied as car fuel, substituting diesel; Turn on by

			applying 1, then turn off the other by giving them value 0
45	CO2_Capture_Cyrogenic	0	Capture and purification of CO2 from the Cryogenic biogas cleaning technology; Bauer et al. 2013; Turn on by applying 1, then turn off the other by giving them value 0
46	CO2_Capture_Cyrogenic_Effecti vity	0.25	Capture efficiently of the CO2 from Cryogenic biogas cleaning; Give value in numeric %; Bauer et al. 2013
47	Bm_Compression200	1	Compression of biomethane to storage tanks, 200 bar; Turn on by applying 1, then turn off the other by giving them value 0; Bauer et al. 2013
48	Bm_Compression300	0	Compression of biomethane to storage tanks, 300 bar; Turn on by applying 1, then turn off the other by giving them value 0; Bauer et al. 2013
49	Bm_Compression45_50	0	Compression of biomethane to gas grid network, 45 - 50 bar; Turn on by applying 1, then turn off the other by giving them value 0; Bauer et al. 2013
50	E_Use_Diesel_car	2.864	MJ/ km; (BMW 1 series, 2008) 1.79 MJ / km =(0.05l/km*36.2MJ/l) or (0.08l/km*36.2MJ/l) =2.864 MJ/km accounted for the extra weight of biogas tanks.
51	E_Use_Gasoline_car	3.24	MJ/ km; (Peugeot 307 2002) 2.268 MJ / km =(0.07l/km*32.4MJ/l) or (0.1l/km*32.4MJ/l) =3.24 MJ/km accounted for the extra weight of biogas tanks.

52	E_Use_Bus	15.1848	MJ / vkm; Hung & Solli 2011
53	Person_Bus	12	Persons per bus on average in Norway; M. Simonsen 2012, Toutain et al. 2008
54	l_LBG_Nm3	1.7	liter LBG per Nm3 biomethane; Bauer et al. 2013
55	Torch	0	numeric % of biogas being torched at biogas plant; add value that is true for the given case in numeric %
56	UpT_Cleaning	0	Cleaning of biogas to meet H2S, SO2 and H2O requirements for CHP utilization; Bauer et al. 2013
57	UpT_WS	1	Upgrading technology Water scrubber; Turn on by applying 1, then turn off the other by giving them value 0; Water Scrubbing; Bauer et al. 2013
58	UpT_ChS	0	Upgrading technology Chemical scrubber; Turn on by applying 1, then turn off the other by giving them value 0; Chemical scrubber
59	UpT_PSA	0	Upgrading technology Pressure Swing Absorption; Turn on by applying 1, then turn off the other by giving them value 0; Pressure Swing Absorption
60	UpT_Membrane	0	Upgrading technology Membrane separation; Turn on by applying 1, then turn off the other by giving them value 0; Membrane filtering system
61	UpT_Cyrogenic	0	Upgrading technology Cryogenic separation; Turn on by applying 1, then turn off the other by giving them value 0; Cryogenic separation

62	HM_0	0	Heavy metal concentration class zero - Agriculture non restrictions; Gjødsels forskriften §27
63	HM_1	1	Heavy metal concentration class one - Agriculture restricted to maximum spreading:5cm/ 10 yr, Gjødsels forskriften §27
64	HM_2	0	Heavy metal concentration class two - Agriculture restricted to maximum spreading:5cm/ 10 yr, Gjødsels forskriften §27
65	HM_3	0	Heavy metal concentration class three - non agriculture or cover landfill, Gjødsels forskriften §27
66	Lr_CH4_Cleaning	0.02	Methane loss to atmosphere in numeric by biogas cleaning % ; No data found! so an assumption where cleaning = Water scrubbing have been applied
67	Lr_Ch4_UpT_WS	0.02	Methane loss to atmosphere in numeric by water scrubbing % ; Baurer et al. 2013
68	Lr_Ch4_UpT_ChS	0.001	Methane loss to atmosphere in numeric % by chemical scrubbing ; Baurer et al. 2013
69	Lr_Ch4_UpT_PSA	0.02	Methane loss to atmosphere in numeric % by Pressure swing absorption; Baurer et al. 2013
70	Lr_Ch4_UpT_Membrane	0.03	Methane loss to atmosphere in numeric % by membrane separation ; Baurer et al. 2013
71	Lr_Ch4_UpT_Cyrogenic	0.05	Methane loss to atmosphere in numeric % by cryogenic separation ; Baurer et al. 2013

72	Lr_CO2_Cleaning	0.02	Carbon dioxide loss to atmosphere in numeric by biogas cleaning % ; No data found! so an assumption where cleaning = Water scrubbing have been applied
73	Lr_CO2_UpT_WS	0.98	Carbon dioxide loss to atmosphere in numeric % by using water scrubber ; Baurer et al. 2013
74	Lr_CO2_UpT_ChS	0.998	Carbon dioxide loss to atmosphere in numeric % by using chemical scrubber ; Baurer et al. 2013
75	Lr_CO2_UpT_PSA	0.98	Carbon dioxide loss to atmosphere in numeric % by using ressure swing absorption ; Baurer et al. 2013
76	Lr_CO2_UpT_Membrane	0.8	Carbon dioxide loss to atmosphere in numeric % by using membrane separation; Baurer et al. 2013
77	Lr_CO2_UpT_Cyrogenic	1	Carbon dioxide loss to atmosphere in numeric % by using cryogenic separation ; Baurer et al. 2013
78	Digestate_Dry_DM	0.3	Numeric % of dry matter content in dry bioresidual ; Sande et al. 2008
79	Compost_Soil	0.3	Numeric % of dry bioresidual content in compost ; Sande et al. 2008
80	DMC_Cattle	0.09	Dry matter content of catle manure; Carlsson & Uldal 2009
81	DMC_Pig	0.08	Dry matter content of pig manure; Carlsson & Uldal 2009
82	DMC_Fat	0.9	Dry matter content of fried fat; Carlsson & Uldal 2009

83	DMC_OMW	0.33	Dry matter content of organic municipal waste; Carlsson & Uldal 2009
84	DMC_Animal_Fat	0.04	Dry matter content of animal fats; Carlsson & Uldal 2009
85	DMC_Fish_Waste	0.42	Dry matter content fish wastes; Carlsson & Uldal 2009
86	DMC_Resturant_Waste	0.27	Dry matter content resturant waste; Carlsson & Uldal 2009
87	DMC_Slaughter_Blood	0.1	Dry matter content of blood from slaughter house ; Carlsson & Uldal 2009
88	DMC_Slaughter_Entrails	0.16	Dry matter content of entrails from slaughter house ; Lyng et al. 2011
89	DMC_Slaughter_Offal	0.3	Dry matter content of offal from slaughter house ; Carlsson & Uldal 2009
90	DMC_Diary	0.2	Dry matter content of average diary products ; Carlsson & Uldal 2009
91	DMC_Fruit_Vegetable	0.15	Dry matter content average from fruits and vegetables ; Carlsson & Uldal 2009
92	DMC_SwSl	0.17	Dry matter content; Carlsson & Uldal 2009 , Stian Wadahl (2014) (DMC 15 - 17%), Tore Fløan (2015) (40%)
93	DMC_ProductX	0.17	Dry matter content; Insert own measured value for total mix
94	VS_Cattle	0.8	Volatile solids of DM cattle manure ; Carlsson & Udal 2009
95	VS_Pig	0.8	Volatile solids of DM pig manure ; Carlsson & Udal 2009

96	VS_Fat	1	Volatile solids of DM fired fatts ; Carlsson & Udal 2009
97	VS_OMW	0.85	Volatile solids of DM organic municipal waste ; Carlsson & Udal 2009
98	VS_Animal_Fat	0.95	Volatile solids of DM animal fatts ; Carlsson & Udal 2009
99	VS_Fish_Waste	0.98	Volatile solids of DM fish wastes ; Carlsson & Udal 2009
100	VS_Resturant_Waste	0.87	Volatile solids of DM resturant wastes ; Carlsson & Udal 2009
101	VS_Slaughter_Blood	0.95	Volatile solids of DM blood from slaughter houses ; Carlsson & Udal 2009
102	VS_Slaughter_Entrails	0.83	Volatile solids of DM entrails from slaughter houses ; Lyng et al. 2011, s.22
103	VS_Slaughter_Offal	0.83	Volatile solids of DM offal from slaughter houses ; Carlsson & Udal 2009
104	VS_Diary	0.82	Volatile solids of DM diary average ; Hamelin et al. 2014
105	VS_Fruit_Vegetable	0.95	Volatile solids of DM average from fruits and vegetables ; Carlsson & Udal 2009
106	VS_SwSl	0.8	Volatile solids of DM sewage sludge ; Wadahl 2014
107	VS_ProductX	0.9	Volatile solids of DM ; Insert own measured value for total mix

108	Share_Cattle	0.7	Numeric % DM share of cattle manure in the average manure mix ; Calculated from Morken et al. 2008
109	Share_Pig	0.3	Numeric % DM share of pig manure in the average manure mix ; Calculated from Morken et al. 2008
110	Share_Fat	1	Numeric % DM share of fried fats in the average fat mix ; Calculated from Morken et al. 2008
111	Share_OMW	1	Numeric % DM share of organic municipal waste in the average organic municipal waste mix ; Calculated from Morken et al. 2008
112	Share_Animal_Fat	0	Numeric % DM share of animal fat waste in the average organic industrial waste mix ; Calculated from Morken et al. 2008
113	Share_Fish_Waste	0.49	Numeric % DM share of fish waste in the average organic industrial waste mix ; Calculated from Morken et al. 2008
114	Share_Resturant_Waste	0.1481	Numeric % DM share of restaurant waste in the average organic industrial waste mix ; Calculated from Morken et al. 2008
115	Share_Slaughter_Blood	0	Numeric % DM share of blood from slaughterhouse waste in the average organic industrial waste mix ; Calculated from Morken et al. 2008
116	Share_Slaughter_Offal	0.245	Numeric % DM share of offal from slaughterhouse waste in the average organic industrial waste mix ; Calculated from Morken et al. 2008

117	Share_Slaughter_Entrails	0	Numeric % DM share of entrails from slaughterhouse waste in the average organic industrial waste mix ; Calculated from Morken et al. 2008
118	Share_Diary	0.117	Numeric % DM share of diary average waste in the average organic industrial waste mix ; Calculated from Morken et al. 2008
119	Share_Fruit_Vegetable	0	Numeric % DM share of average from fruits and vegetable waste in the average organic industrial waste mix ; Calculated from Morken et al. 2008
120	CH4_Share_Pig	0.65	Methane share in numeric % of the produced biogas from cattle manure ; Carlsson & Udal 2009
121	CH4_Share_Cattle	0.65	Methane share in numeric % of the produced biogas from pig manure ; Carlsson & Udal 2009
122	CH4_Share_Fat	0.68	Methane share in numeric % of the produced biogas from fried fats ; Carlsson & Udal 2009
123	CH4_Share_OMW	0.63	Methane share in numeric % of the produced biogas from organic municipal wastes ; Carlsson & Udal 2009
124	CH4_Share_Animal_Fat	0.6	Methane share in numeric % of the produced biogas from animal fats ; assumed from qualitative data ; Morken et al. 2008 , Carlsson & Udal 2009
125	CH4_Share_Fish_Waste	0.71	Methane share in numeric % of the produced biogas from fish wastes ; Calrsson & Udal 2009

126	CH4_Share_Resturant_Waste	0.63	Methane share in numeric % of the produced biogas from restaurant wastes ; Calrsson & Udal 2009
127	CH4_Share_Slaughter_Blood	0.63	Methane share in numeric % of the produced biogas from blood from slaughter house waste ; Calrsson & Udal 2009
128	CH4_Share_Slaughter_Entrails	0.63	Methane share in numeric % of the produced biogas from entrails from slaughter house waste ; Lyng et al. 2011
129	CH4_Share_Slaughter_Offal	0.68	Methane share in numeric % of the produced biogas from offal from slaughter house waste ; Calrsson & Udal 2009
130	CH4_Share_Diary	0.59	Methane share in numeric % of the produced biogas from diary average Hamelin et al. 2014
131	CH4_Share_Fruit_Vegetable	0.6	Methane share in numeric % of the produced biogas from average from fruits and vegetable waste ; Carlsson & Udal 2014
132	CH4_Share_SwSl	0.6	Methane share in numeric % of the produced biogas from sewage sludge ; Wadahl 2014
133	CH4_Share_ProductX	0.6	Methane share in numeric % of the produced biogas ; Insert own measured value for total mix
134	MY_Cattle	213	Methane yield; m3/ ton volatile solids (VS) cattle manure ; Carlsson & Udal 2009
135	MY_Pig	268	Methane yield; m3/ ton VS pig manure ; Carlsson & Udal 2009

136	MY_Fat	757	Methane yield; m3/ ton VS fried fats ; Carlsson & Udal 2009
137	MY_OMW	461	Methane yield; m3/ ton VS organic municipal waste ; Carlsson & Udal 2009
138	MY_Animal_Fat	682	Methane yield; m3/ ton VS animal fats ; Carlsson & Udal 2009
139	MY_Fish_Waste	930	Methane yield; m3/ ton VS fish waste ; Carlsson & Udal 2009
140	MY_Resturant_Waste	461	Methane yield; m3/ ton VS restaurant waste ; Carlsson & Udal 2009
141	MY_Slaughter_Blood	547	Methane yield; m3/ ton VS blood from slaughterhouse ; Carlsson & Udal 2009
142	MY_Slaughter_Entrails	688	Methane yield; m3/ ton VS entrails from slaughterhouse ; Lyng et al. 2011
143	MY_Slaughter_Offal	664	Methane yield; m3/ ton VS offal from slaughterhouse ; Hamelin et al. 2014
144	MY_Diary	277	Methane yield; m3/ ton VS diary average ; Carlsson & Udal 2009
145	MY_Fruit	666	Methane yield; m3/ ton VS average fruits and vegetable ; Carlsson & Udal 2009
146	MY_SwSl	336	Methane yield; m3/ ton VS sewage sludge ; Wadahl 2014
147	MY_ProductX	500	Methane yield; m3/ ton VS ; Insert own measured value for total mix
148	DS_Cattle	0.504	Remaining solids in numeric % of DM cattle manure ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Carlsson & Udal 2009

149	DS_Pig	0.504	Remaining solids in numeric % of DM pig manure ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Carlsson & Udal 2009
150	DS_Fat	0	Remaining solids in numeric % of DM fried fats ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Carlsson & Udal 2009
151	DS_OMW	0.46	Remaining solids in numeric % of DM organic municipal waste ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Carlsson & Udal 2009
152	DS_Animal_Fat	0.05	Remaining solids in numeric % of DM animal fat ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Carlsson & Udal 2009
153	DS_Fish_Waste	0.36	Remaining solids in numeric % of DM fish wastes ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Carlsson & Udal 2009
154	DS_Resturant_Waste	0.3	Remaining solids in numeric % of DM restaurant wastes ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Carlsson & Udal 2009
155	DS_Slaughter_Blood	0.38	Remaining solids in numeric % of DM blood from slaughterhouse ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Carlsson & Udal 2009
156	DS_Slaughter_Entrails	0.48	Remaining solids in numeric % of DM entrails from slaughterhouse ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Carlsson & Udal 2009

157	DS_Slaughter_Offal	0.46	Remaining solids in numeric % of DM offal from slaughterhouse ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Lyng et al. 2011, s.22
158	DS_Diary	0.53	Remaining solids in numeric % of DM average diary products ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Carlsson & Udal 2009
159	DS_Fruit_Vegetable	0.46	Remaining solids in numeric % of DM average fruit and vegetable waste ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Hamelin et al. 2014
160	DS_SwSl	0.6	Remaining solids in numeric % of DM sewage sludge ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Carlsson & Udal 2009
161	DS_ProductX	0.5	Remaining solids in numeric % of DM ; Remaining solids = Degrability coefficient of VS * (VS/DM) ; Insert own measured value for total mix
162	N_Cattle	44.8	kg N/ ton DM cattle manure; Poeschl et al. 2012a
163	N_Pig	99.38	kg N/ ton DM pig manure; Karlengen et al 2012
164	N_Fat	35	kg N/ ton DM fat; assumed the same as for animal fat ; Poeschl et al. 2012a via Helm 2009
165	N_OMW	40	kg N/ ton DM organic municipal waste; Poeschl et al. 2012a via Helm 2009
166	N_Animal_Fat	35	kg N/ ton DM animal fat; Poeschl et al. 2012a via Helm 2009

167	N_Fish_Waste	39.23	kg N/ ton DM fish waste; Gebauer & Eikebrokk 2005
168	N_Resturant_Waste	6	kg N/ ton DM resturant waste; Carlsson & Udal 2009
169	N_Slaughter_Blood	15	kg N/ ton DM slaughter house blood; Alvarez & Lidén 2007; (15 for cattle blood and 8.3 for pig blood) table 2 in the given paper
170	N_Slaughter_Entrails	25	kg N/ ton DM slaughter house entrails; Lyng et al. 2011 s. 22
171	N_Slaughter_Offal	59	kg N/ ton DM slaughter house offal; Lyng et al. 2011 s. 21
172	N_Diary	8.06	kg N/ ton DM diary products; Hamelin et al. 2014
173	N_Fruit_Vegetable	1.1	kg N/ ton DM Fruit and vegetables; Alvarez & Lidén 2007
174	N_SwSl	17.5	kg N/ ton DM sewage sludge ; Yara 2011
175	N_ProductX	0	kg N/ ton DM ; Insert own measured value for total mix
176	P_Cattle	4.56	kg P/ ton DM cattle manure ; Poeshl et al. 2012a via Helm 2009
177	P_Pig	13.98	kg P/ ton DM pig manure ; Karlengen et al. 2012
178	P_Fat	3.8	kg P/ ton DM frying fat Assumed the same as separated fats ; Poeshl et al. 2012a via Helm 2010
179	P_OMW	0.59	kg P/ ton DM organic municipal waste ; Poeshl et al. 2012a via Helm 2010
180	P_Animal_Fat	3.8	kg P/ ton DM animal fat ; Poeshl et al. 2012a via Helm 2010

181	P_Fish_Waste	1.1	kg P/ ton DM fish wastes ; Carlsson & Udal 2009 , Genauer & Eikebrokk 2005
182	P_Resturant_Waste	0.9	kg P/ ton DM restaurant wastes ; Poeshl et al. 2012a via Helm 2010
183	P_Slaughter_Blood	0.1	kg P/ ton DM blood from slaughterhouse ; Alvarez & Lidén 2007
184	P_Slaughter_Entrails	10.5	kg P/ ton DM entrails from slaughterhouse ; Lyng et al. 2011 s. 22
185	P_Slaughter_Offal	40	kg P/ ton DM offal from slaughterhouse waste ; Lyng et al. 2011 s. 21
186	P_Diary	1.12	kg P/ ton DM average diary products ; Hamelin et al. 2014
187	P_Fruit_Vegetable	0.2	kg P/ ton DM average of fruits and vegetable waste ; Alvarez & Lidén 2007
188	P_SwSl	16	kg P/ ton DM sewage sludge ; Yara 2011
189	P_ProductX	0	kg P/ ton DM ; Insert own measured value for total mix
190	Digestate_ava_P	1	Plant availability phosphorus (P) for digested organic wastes ; Lyng et al 2011
191	Digestate_ava_N	0.85	Plant availability nitrogen (N) for digested organic wastes ; Luostarinen et al 2011, s41
192	Manure_ava_P	1	Plant availability phosphorus (P) for manure; Lyng et al 2011
193	Manure_ava_N	0.3	Plant availability nitrogen (N) for manure; Luostarinen et al 2011, s 41

194	HHV_Cattle	15358	MJ / ton DM cattle manure Higher heating value (HHV) ; Annamali & Sweeten 1987, calculated the average for all manure types in table 1, s 1206
195	HHV_Pig	15358	MJ / ton DM pig manure (HHV) ; Assumed the same as for Cattle manure ; Annamali & Sweeten 1987
196	HHV_Fat	37550	MJ / ton DM frying fat (HHV) ; Metha & Anand 2009
197	HHV_OMW	18500	MJ / ton DM organic municipal waste (HHV) ; Wirsenius 2000
198	HHV_Animal_Fat	35550	MJ / ton DM animal fat (HHV) ; Assumed the same as for fat - 2000 MJ so as to account for the impurity ; qualitative assumption , Carlsson & Udal 2009
199	HHV_Fish_Waste	20000	MJ / ton DM fish waste (HHV); Wirsenius 2000
200	HHV_Resturant_Waste	18500	MJ / ton DM; Wirsenius 2000
201	HHV_Slaughter_Blood	18000	MJ / ton DM blood from slaughterhouse ; Assumed same as Offal , Wirsenius 2000
202	HHV_Slaughter_Entrails	18000	MJ / ton DM entrails from slaughterhouse; Assumed same as Offal , Wirsenius 2000
203	HHV_Slaughter_Offal	17500	MJ / ton DM offal from slaughterhouse ; Wirsenius 2000
204	HHV_Diary	15650	MJ / ton DM average diary products: 3.13 kj/ wet weight yogurt; Matvaretabelen.no

205	HHV_Fruit_Vegetable	17000	MJ / ton DM average fruit and vegetable waste: Assumed the same as for uneaten food , Wirsenius 2000
206	HHV_SwSl	15000	MJ / ton DM sewage sludge ; Fryba et al. 2014
207	HHV_ProductX	0	MJ / ton DM ; Insert own meassured value for total mix
208	Pl_Avg	0.1	Phosphorus loss (P) during digestion, assumed 10% for all substrates ; Möller & Müller 2012 , Hospido et al. 2005 (Sewage sludge)
209	Molar_Mass_CH4	0.7143	kg/Nm3 ; Mass density for CH4 per m3 - found by applying the ideal gas law m/V = (P*M)/(R*T)
210	Molar_Mass_CO2	1.9642	kg/Nm3 ; Mass density for CO2 per m3 - found by applying the ideal gas law m/V = (P*M)/(R*T)
211	SHC	4.18	MJ/ (ton*C) ; Specific heat capacity for water - assumed the same fro water and organic material ; Coultry et al. 2013
212	FSH	14	Celsius, organic waste substrate temperature in Celsius in to treatment plant, pasteurization start temperature ; Jørgensen 2015, Bauer et al.2013
213	AFSH	44	Celsius, organic waste substrate temperature in Celsius after heat recovery, pasteurization start temperature ; Jørgensen 2015 ; Calculator follows in Appendix XX
214	PH	70	Pasteurization treatment heat for one hour; Jørgensen 2015, Ecopro (2015), Morken et al. 2008 ; Calculator follows in Appendix XX

215	Temp_Incinerated_Water	120	Temperatur absorbed by the water within the incinerated material - resulting in a lesser energy output than the HHV suggests ; Jørgensen 2015
216	Lr_HP	0.05	Loss rate Heating Pretreatment ; Hamelin et al. 2014
217	HV_W	2260	MJ/ton water to steam - energy requirements in conversion from liquid to gaseous state ; Heat of vaporization of water; Wikipedia: "Enthalpy of vaporization"
218	Energy_Methane	37.5	MJ/Nm3 ; energy density methane (CH4) per m3; Morken et al. 2008
219	Energy_Efficiency_e_CHP	0.12	Energy efficiency for electricity in biogas combined heat and power plant (CHP), biogas utilization : Hung & Solli 2011
220	Energy_Efficiency_H_CHP	0.8	Energy efficiency for heat in CHP, biogas utilization ; Hung & Solli 2011
221	Energy_Efficiency_e_Incineration	0	Energy efficiency for electricity production in incineration plant ; Hung & Solli 2011
222	Energy_Efficiency_H_Incineration	0.8	Energy efficiency for heat production in incineration plant ; Hung & Solli 2011
223	kWh_to_MJ	3.6	MJ converted to kWh
224	Share_Fly_ash	0.1	Dry matter that goes to fly ash ; Boesch et al. 2014
225	Share_HM_Fly_ash	0.3	Amount of heavy metals that foes to fly ash ; Boesch et al. 2014
226	Share_Bottom_ash	0.9	Dry matter that goes to fly ash ; Boesch et al. 2014

227	Share_HM_Bottom_ash	0.7	Amount of heavy metals that foes to fly ash ; Boesch et al. 2014
228	Cd0	0.4	mg/kg DM bioresidual Cadmium (Cd) class 0; Gjødselsforskriften §10
229	Cd1	0.8	mg/kg DM bioresidual Cadmium (Cd) class 1; Gjødselsforskriften §10
230	Cd2	2	mg/kg DM bioresidual Cadmium (Cd) class 2; Gjødselsforskriften §10
231	Cd3	5	mg/kg DM bioresidual Cadmium (Cd) class 3; Gjødselsforskriften §10
232	Pb0	40	mg/kg DM bioresidual Lead (Pb) class 0; Gjødselsforskriften §10
233	Pb1	60	mg/kg DM bioresidual Lead (Pb) class 1; Gjødselsforskriften §10
234	Pb2	80	mg/kg DM bioresidual Lead (Pb) class 2; Gjødselsforskriften §10
235	Pb3	200	mg/kg DM bioresidual Lead (Pb) class 3; Gjødselsforskriften §10
236	Hg0	0.2	mg/kg DM bioresidual Mercury (Hg) class 0; Gjødselsforskriften §10
237	Hg1	0.6	mg/kg DM bioresidual Mercury (Hg) class 1; Gjødselsforskriften §10
238	Hg2	3	mg/kg DM bioresidual Mercury (Hg) class 2; Gjødselsforskriften §10
239	Hg3	5	mg/kg DM bioresidual Mercury (Hg) class 3; Gjødselsforskriften §10
240	Ni0	20	mg/kg DM bioresidual Nickel (Ni) class 0; Gjødselsforskriften §10
241	Ni1	30	mg/kg DM bioresidual Nickel (Ni) class 1; Gjødselsforskriften §10

242	Ni2	50	mg/kg DM bioresidual Nickel (Ni) class 2; Gjødselsforskriften §10
243	Ni3	80	mg/kg DM bioresidual Nickel (Ni) class 3; Gjødselsforskriften §10
244	Zn0	150	mg/kg DM bioresidual Zinc (Zn); Gjødselsforskriften §10
245	Zn1	400	mg/kg DM bioresidual Zinc (Zn); Gjødselsforskriften §10
246	Zn2	800	mg/kg DM bioresidual Zinc (Zn); Gjødselsforskriften §10
247	Zn3	1500	mg/kg DM bioresidual Zinc (Zn); Gjødselsforskriften §10
248	Cu0	50	mg/kg DM bioresidual Copper (Cu); Gjødselsforskriften §10
249	Cu1	150	mg/kg DM bioresidual Copper (Cu); Gjødselsforskriften §10
250	Cu2	650	mg/kg DM bioresidual Copper (Cu); Gjødselsforskriften §10
251	Cu3	1000	mg/kg DM bioresidual Copper (Cu); Gjødselsforskriften §10
252	Cr0	50	mg/kg DM bioresidual Chromium (Cr); Gjødselsforskriften §10
253	Cr1	60	mg/kg DM bioresidual Chromium (Cr); Gjødselsforskriften §10
254	Cr2	100	mg/kg DM bioresidual Chromium (Cr); Gjødselsforskriften §10
255	Cr3	150	mg/kg DM bioresidual Chromium (Cr); Gjødselsforskriften §10

256	E_Sorting	4.16	kWh electricity (e-)/ ton sorted organic waste; Jørgensen 2015, Composed of Sorting (3.12) and crushing (1.04).
257	E_Pasteurization	1.733	kWh (e-) /ton pasteurized organic waste; Jørgensen 2015
258	E_AD	2.08	kWh (e-)/ton organic waste to treatment; Jørgensen 2015
259	E_Dewatering	2.4	kWh (e-)/m3 bioresidual slurry to dewatering ; Rehl & Müller 2011
260	E_WT	0.42889	kWh (e-)/ m3 (ton) cleaned water to treatment system ; Ecoinvent 3.0
261	E_WWT	0.3997	kWh (e-)/ m3 waste water to waste water treatment plant (WWTP) ; Ecoinvent 3.0
262	E_Spreading_Dry	0.16	kWh (diesel) / ton spreader dry fertilizer (Water content >25%) ;
263	E_Spreading_Wet	0.8	kWh (diesel) / ton spreader wet fertilizer (Water content <75%) ;
264	E_Tanking	0.16	kWh (e-) / m3 tanking of biomethane ; Soli et al.2011
265	E_Cleaning	0	kWh (e-)/m3 raw biogas cleaned ; Bauer et al. 2013
266	E_UpT_WS	0.23	kWh (e-) /m3; Bauer et al. 2013
267	E_UpT_ChS	0.13	kWh (e-) /m3; Bauer et al. 2013
268	E_UpT_PSA	0.25	kWh (e-) /m3 ; Bauer et al. 2013
269	E_UpT_Membrane	0.3	kWh (e-) /m3; Bauer et al. 2013
270	E_UpT_Cyrogenic	0.4564	kWh (e-)/m3; Bauer et al. 2013
271	H_UpT_ChS	1.96	MJ heat (H)/m3; Bauer et al. 2013
272	E_LBG_Process	0.75	kWh (e-)/m3; Bauer et al. 2013

273	E_Compression45_50	0.16	kWh (e-)/m3 biomethane compressed to 45 - 50 bar ; Bauer et al. 2013
274	E_Compression200	0.21	kWh (e-)/m3 biomethane compressed to 20 bar ; Bauer et al. 2013
275	E_Compression300	0.25	kWh (e-)/ m3 biomethane compressed to 300 bar ; Bauer et al. 2013
276	E_Incineration	65.7	kWh (e-)/ ton waste Hospido et al. 2005
277	E_CHP	0.04	% of total energy content in
278	km_Manure	50	Average transport distance for manure to plant ; Part of the assumption in the LCA model
279	km_Fat	50	Average transport distance for frying fat to plant ; Part of the assumption in the LCA model
280	km_OMW	19	Average transport distance for organic municipal waste to plant ; Part of the assumption in the LCA model
281	km_OIW	50	Average transport distance for organic industrial waste to plant ; Part of the assumption in the LCA model
282	km_SwSl	0	Average transport distance for sewage sludge to plant ; Part of the assumption in the LCA model
283	km_ProductX	0	Average transport distance for self defined organic water mix to plant ; Part of the assumption in the LCA model
284	km_Dg	50	Average transport distance for bioresidual to application area ; Part of the assumption in the LCA model

285	km_Compost	50	Average transport distance for composted bioresidual to application area ; Part of the assumption in the LCA model
286	km_Bm_LBG	10	Average transport distance for liquid biogas (LBG) to filling station ; Part of the assumption in the LCA model
287	km_Fly_Ash	250	Average transport distance for fly ash to hazardous landfill ; Part of the assumption in the LCA model
288	km_Bottom_Ash	1	Average transport distance for bottom ash to land fill ; Part of the assumption in the LCA model
289	Diesel_Consumption	0.022	kg diesel / tkm or (43,1 MJ diesel / tkm) ; Ecoinvent 3.0 ; MJ / kg diesel https://snl.no/energivare
290	E_Transport_EUR5	1.056	MJ diesel per tkm: Energy per transport unit
291	E_Transport_Municipal_Collection	16.12	MJ diesel per tkm: Energy per transport unit
292	Share_NH4	0.75	numeric % share of nitrogen (N) bound as NH4 - N in the bioresidual ; Bernstad & Jansen 2011 via Svensson et al. 2004, Britto & Kronzucker 2002
293	Share_NO3	0.018	numeric % share of nitrogen (N) bound as NO3 - N in the bioresidual ; Bernstad & Jansen 2011 via Svensson et al. 2004
294	Share_N_Org	0.232	numeric % share of nitrogen (N) bound as organic N - N in the bioresidual ; Bernstad & Jansen 2011 via Svensson et al. 2004

295	N_N2	1	Mass (kg) of N per N-compound; Bernstad & Jansen 2011
296	N_N2O	0.636	Mass (kg) of N per N-compound; Bernstad & Jansen 2011
297	N_NH3	0.824	Mass (kg) of N per N-compound; Bernstad & Jansen 2011
298	NH3_dig	0.96	numeric % of N-total loss converted to ammonia (NH3) ; Bernstad & Jansen 2011
299	N2O_dig	0.02	numeric % of N-total loss converted to dinitrogen monoxide (N2O) ; Bernstad & Jansen 2011
300	Nl_Waste	0.17	Numeric % of N-total loss from digested organic waste ; Bernstad & Jansen 2011, table 5, their reference Sonesson (1996) (17.0% loss)
301	Nl_Manure	0.179	Numeric % of N-total loss from untreated manure ; Bernstad & Jansen 2011, table 5, their reference Sonesson (1996) (17.9% loss)
302	Nl_Manure_Dig	0.218	Numeric % of N-total loss from digested manure; Bernstad & Jansen 2011, table 5, their reference Sonesson (1996) (21.8% loss)
303	Nl_undig_NH3	0.99	Numeric % production of ammonia (NH3) from undigested manure relative to digested manure ; Assumed from Amon et al. 2006
304	Nl_dig_NH3	1	Numeric % production of ammonia (NH3) from digested manure relative to digested manure ; Assumed from Amon et al. 2006

305	Nl_sepa_NH3	1.78	Numeric % production of ammonia (NH3) from digested and separated manure relative to digested manure ; Assumed from Amon et al. 2006
306	Nl_dig_N2O	1	Numeric % production of dinitrogen monoxide (N2O) from undigested manure relative to digested bioresidual ; Calculated from Amon et al. 2006
307	Nl_undig_N2O	0.77	Numeric % production of dinitrogen monoxide (N2O) from undigested manure relative to digested bioresidual ; Calculated from Amon et al. 2006
308	Nl_sepa_N2O	1.19	Numeric % production of dinitrogen monoxide (N2O) produced from separated bioresidual relative to digested bioresidual ; Calculated from Amon et al. 2006
309	Nl_dig_N2	1	Numeric % production of dinitrogen (N2) from digested bioresidual relative to digested bioresidual ; Calculated from Amon et al. 2006
310	Nl_undig_N2	1	Numeric % production of dinitrogen (N2) from undigested manure relative to digested bioresidual ; Calculated from Amon et al. 2006
311	Nl_Sepa_N2	1	Numeric % production of dinitrogen (N2) from separated bioresidual relative to digested bioresidual ; Calculated from Amon et al. 2006
312	Storage_Untreated_NH3	0.18	Numeric % ammonia (NH3) produced by storage of untreated manure ; Calculated from Amon et al. 2006

313	Storage_Sepa_Dry_NH3	0.71	Numeric % ammonia (NH3) produced by storage of dewatered bioresidual ; Calculated from Amon et al. 2006
314	Storage_Sepa_Wet_NH3	0.1	Numeric % ammonia (NH3) produced by storage of liquid fraction of dewatered bioresidual ; Calculated from Amon et al. 2006
315	Storage_Digested_NH3	0.04	Numeric % ammonia (NH3) produced by storage of digested organic wastes substrates ; Calculated from Amon et al. 2006
316	Storage_Untreated_N2O	0.84	Numeric % dinitrogen monoxide (N2O) produced by storage of untreated manure ; Calculated from Amon et al. 2006
317	Storage_Sepa_Dry_N2O	0.31	Numeric % dinitrogen monoxide (N2O) produced by storage of dewatered bioresidual ; Calculated from Amon et al. 2006
318	Storage_Sepa_Wet_N2O	0.46	Numeric % dinitrogen monoxide (N2O) produced by storage of liquid fraction of dewatered bioresidual ; Calculated from Amon et al. 2006
319	Storage_Digested_N2O	0.91	Numeric % dinitrogen monoxide (N2O) produced by storage of digested organic wastes substrates ; Calculated from Amon et al. 2006
320	Storage_Untreated_CH4	1	Numeric % methane (CH4) produced by storage of untreated manure ; Calculated from Amon et al. 2006
321	Storage_Sepa_Dry_CH4	0.215	Numeric % methane (CH4) produced by storage of dewatered bioresidual ; Calculated from Amon et al. 2006

322	Storage_Sepa_Wet_CH4	0.785	Numeric % methane (CH4) produced by storage of liquid fraction of dewatered bioresidual ; Calculated from Amon et al. 2006
323	Storage_Digested_CH4	1	Numeric % methane (CH4) produced by storage of digested organic wastes substrates ; Calculated from Amon et al. 2006
324	App_Sepa_Dry_NH3	0.1654	Numeric % ammonia (NH3) produced by application of dewatered bioresidual ; Calculated from Amon et al. 2006
325	App_Sepa_Wet_NH3	0.02258	Numeric % ammonia (NH3) produced by application of liquid fraction of dewatered bioresidual ; Calculated from Amon et al. 2006
326	App_Sepa_Dry_N2O	0.1331	Numeric % dinitrogen monoxide (N2O) produced by application of dewatered bioresidual ; Calculated from Amon et al. 2006
327	App_Sepa_Wet_N2O	0.09	Numeric % dinitrogen monoxide (N2O) produced by application of liquid fraction of dewatered bioresidual ; Calculated from Amon et al. 2006
328	App_Sepa_Dry_CH4	0	Numeric % methane (CH4) produced by application of liquid fraction of dewatered bioresidual ; Calculated from Amon et al. 2006
329	App_Sepa_Wet_CH4	0	Numeric % methane (CH4) produced by storage of liquid fraction of dewatered bioresidual ; Calculated from Amon et al. 2006
330	CH4_Undig	3.01	Numeric % methane (CH4) produced from undigested manure relative to

			digested bioresidual ; Calculated from Amon et al. 2006
331	CH4_Wet	1	Numeric % methane (CH4) produced from digested bioresidual relative to digested bioresidual ; Calculated from Amon et al. 2006
332	CH4_Sepa	0.58	Numeric % methane (CH4) produced from separated manure relative to digested bioresidual ; Calculated from Amon et al. 2006
333	N_Runoff_SW	0	Nitrogen (N) runoff to surface water ; Bernstad & Jansen 2011
334	N_Runoff_GW	0.22	Nitrogen (N) runoff to ground water ; Bernstad & Jansen 2011
335	P_Runoff_SW	0.107	Phosphorus (P) runoff to surface waters; Hamilton et al. 2015

Appendix 22 – Inventory calculation list from LCA model developed in Simapro 8

Calculated parameters		
Mass_Pt	Mass_Manure*(1-MDF)+Mass_OMW+Mass_OIW+Mass_SwSl+Mass_ProductX	Ton / FU , organic waste substrate wet wheight (WW)
DM_AD_Balance	Manure+Fat+OMW+OIW+SwSL+ProductX	Functional unit balance: Warning: If > or < 1; then an error have occured and the resulst are not correct.
Mass_Cattle	((DM_AD/DMC_Cattle)*Manure*Share_Cattle)	Ton / FU , total cattle manure (WW) assessed in the LCA

Mass_Pig	(DM_AD/DMC_Pig)*Manure*Share_Pig	Ton / FU , total pig manure (WW) assessed in the LCA
Mass_Manure	(Mass_Cattle+Mass_Pig)	Ton / FU , total manure (WW) assessed in the LCA
Mass_Fat	(DM_AD/DMC_Fat)*Fat*Share_Fat	Ton / FU , total fat (WW) assessed in the LCA
Mass_OMW	((DM_AD/DMC_OMW)*OMW*Share_OMW)	Ton / FU , total organic municipal waste (WW) assessed in the LCA
Mass_Animal_Fat	OIW*((DM_AD/DMC_Animal_Fat)*Share_Animal_Fat)	Ton / FU , total animal fat (WW) assessed in the LCA
Mass_Fish_Waste	OIW*((DM_AD/DMC_Fish_Waste)*Share_Fish_Waste)	Ton / FU , total fish waste (WW) assessed in the LCA
Mass_Resturant_Waste	OIW*((DM_AD/DMC_Resturant_Waste)*Share_Resturant_Waste)	Ton / FU , total restaurant waste (WW) assessed in the LCA
Mass_Slaughter_Blood	OIW*((DM_AD/DMC_Slaughter_Blood)*Share_Slaughter_Blood)	Ton / FU , total blood from slaughterhouse (WW) assessed in the LCA
Mass_Slaughter_Entrails	OIW*((DM_AD/DMC_Slaughter_Entrails)*Share_Slaughter_Entrails)	Ton / FU , total entrails from slaughterhouse (WW) assessed in the LCA
Mass_Slaughter_Offal	OIW*((DM_AD/DMC_Slaughter_Offal)*Share_Slaughter_Offal)	Ton / FU , total offal from slaughterhouse (WW) assessed in the LCA
Mass_Diary	OIW*((DM_AD/DMC_Diary)*Share_Diary)	Ton / FU , total diary (WW) assessed in the LCA
Mass_Fruit_Vegetable	OIW*((DM_AD/DMC_Fruit_Vegetable)*Share_Fruit_Vegetable)	Ton / FU , total fruit and vegetables (WW) assessed in the LCA
Mass_OIW	(Mass_Animal_Fat+Mass_Fish_Waste+Mass_Resturant_Waste+Mass_Slaughter_Blood+Mass_Slaughter_Entrails+Mass_Slaughter_Offal+Mass_Diary+Mass_Fruit_Vegetable)	Ton / FU , total indsutrial waste (WW) assessed in the LCA

Mass_SwSl	SwSl*(DM_AD/DMC_SwSl)	Ton / FU , total sewage sludge (WW) assessed in the LCA
Mass_ProductX	ProductX*(DM_AD/DMC_ProductX)	Ton / FU , total self defined substrate
Mass_AD	Mass_Pt+Water_To_Pretreatment	Ton / FU , total mass to treatment anaerobic treatment facility
Mass_Bg_AD	((Methane_ADpot*Molar_Mass_CH4)+(CO2_AD*Molar_Mass_CO2))	Kg / FU , biogass out from anaerobic treatment facility
Mass_Methane	(Methane_ADpot*Molar_Mass_CH4)	Kg / FU , methane (CH4) out from anaerobic treatment facility
Mass_Carbondioxide	(CO2_AD*Molar_Mass_CO2)	Kg / FU , carbon dioxide (CO2) out from anaerobic treatment facility
Mass_Balance_System	(Mass_Bg_AD+Bg_Pot_Mass_Storage+N_Lost+(Dg_Raw_DM*1000))/1000	Warning: If > or < 1; then an error have occured and the resulst are not correct.
Mass_OMW_Inorganic	Mass_OMW+(1/(1-Inorganic_OMW)-1)	Kg / FU , organic municipal waste included inorganic wastes [important for transport]
Mass_OIW_Inorganic	Mass_OIW+(1/(1-Inorganic_OIW)-1)	Kg / FU , organic industrial waste included inorganic wastes [important for transport]
Mass_Inorganic_OMW	1/(1-Inorganic_OMW)-1	Kg / FU , inorganic waste that needs to be separated from organic municipal waste
Mass_Inorganic_OIW	1/(1-Inorganic_OIW)-1	Kg / FU , inorganic waste that needs to be separated from organic industrial waste
Water_Balance	(DM_AD/Mass_Pt)	numeric % of dry matter in feedstock of organic waste substrates
Water_To_Pretreatment	((DM_AD-(DM_Manure*MDF))/(1-OWc_AD)-Mass_Pt)+(Dg_Raw_DM-DM_AD/((1-Owc_AD)/DM_AD))*Recycle_W_AD	Raw data water to pretreatment to fulfill the optimal consentration water; If water is recyceled a change in water source is apparent in the pretreatmetn process in the"Process" inventory

Water_Added	Iff(Water_To_Pretreatment<0, 0, Water_To_Pretreatment)	Ton / FU , water added to the feedstock to anaerobic digestion [fullfillment of OWc]
Water_DW	Iff(-Water_To_Pretreatment<0, 0, -Water_To_Pretreatment)	Ton / FU , water dewatered from the feedstock to the anaerobic digestion [fullfillment of OWc]
Balance_Inputmix	Manure+Fat+OMW+OIW+SwSl+ProductX	Warning: If > or < 1; then an error have occured and the resulst are not correct. Balance of organic waste sectors
Balance_Manure	Share_Cattle+Share_Pig	Warning: If > or < 1; then an error have occured and the resulst are not correct. Balance of manure composition in manure sector
Balance_OIW	Share_Animal_Fat+Share_Resturant_Waste+Share_Fish_Waste+Share_Slaughter_Blood+Share_Slaughter_Entrails+Share_Slaughter_Offal+Share_Diary	Warning: If > or < 1; then an error have occured and the resulst are not correct. Balance of organic waste composition in OIW sector
Methane_Cattle	(DMC_Cattle*Mass_Cattle*(1-DS_Cattle)*MY_Cattle)*(1-MDF)	Nm3 / FU , Methane (CH4) produced from digestion of cattle manure
Methane_Pig	(DMC_Pig*Mass_Pig*(1-DS_Pig)*MY_Pig)*(1-MDF)	Nm3 / FU , Methane (CH4) produced from digestion of pig manure
Methane_Manure	(Methane_Cattle+Methane_Pig)-Methane_Manure_Storage_AD	Nm3 / FU , Methane (CH4) produced as sum of digested manures
Methane_Fat	(DMC_Fat*Mass_Fat*(1-DS_Fat)*MY_Fat)	Nm3 / FU , Methane (CH4) produced from digestion of frying fats
Methane_OMW	(DMC_OMW*Mass_OMW*(1-DS_OMW)*MY_OMW)	Nm3 / FU , Methane (CH4) produced from digestion of organic municipal waste
Methane_Animal_Fat	(DMC_Animal_Fat*Mass_Animal_Fat*(1-DS_Animal_Fat)*MY_Animal_Fat)	Nm3 / FU , Methane (CH4) produced from digestion of animal fats
Methane_Fish_Waste	(DMC_Fish_Waste*Mass_Fish_Waste*(1-DS_Fish_Waste)*MY_Fish_Waste)	Nm3 / FU , Methane (CH4) produced from digestion of fish waste

Methane_Resturant	(DMC_Resturant_Waste*Mass_Resturant_Waste*(1-DS_Resturant_Waste)*MY_Resturant_Waste)	Nm3 / FU , Methane (CH4) produced from digestion of resturant waste
Methane_Slaughter_Blood	(DMC_Slaughter_Blood*Mass_Slaughter_Blood*(1-DS_Slaughter_Blood)*MY_Slaughter_Blood)	Nm3 / FU , Methane (CH4) produced from digestion of blood from slaughterhouse
Methane_Slaughter_Entrails	(DMC_Slaughter_Entrails*Mass_Slaughter_Entrails*(1-DS_Slaughter_Entrails)*MY_Slaughter_Entrails)	Nm3 / FU , Methane (CH4) produced from digestion of entrails from slaugtherhouse
Methane_Slaughter_Offal	(DMC_Slaughter_Offal*Mass_Slaughter_Offal*(1-DS_Slaughter_Offal)*MY_Slaughter_Offal)	Nm3 / FU , Methane (CH4) produced from digestion of offal from slaugherhouse
Methane_Diary	(DMC_Diary*Mass_Diary*(1-DS_Diary)*MY_Diary)	Nm3 / FU , Methane (CH4) produced from digestion of diary products
Methane_Fruit_Vegetable	(DMC_Fruit_Vegetable*Mass_Fruit_Vegetable*(1-DS_Fruit_Vegetable)*MY_Fruit)	Nm3 / FU , Methane (CH4) produced from digestion of fruits and vegetables
Methane_OIW	(Methane_Animal_Fat+Methane_Fish_waste+Methane_Resturant+Methane_Slaughter_Blood+Methane_Slaughter_Entrails+Methane_Slaughter_Offal+Methane_Diary+Methane_Fruit_Vegetable)	Nm3 / FU , Methane (CH4) produced as sum digestion of organic industrial waste
Methane_SwSl	DMC_SwSl*Mass_SwSl*(1-DS_SwSl)*MY_SwSl	Nm3 / FU , Methane (CH4) produced from digestion of sewagesludge
Methane_ProductX	DMC_ProductX*Mass_ProductX*(1-DS_ProductX)*MY_ProductX	Nm3 / FU , Methane (CH4) produced from digestion of self defined organic waste substrate
CH4_Manure_MAX	((DMC_Cattle*Mass_Cattle*(VS_Cattle)*MY_Cattle)+(DMC_Pig*Mass_Pig*(VS_Pig)*MY_Pig))	Nm3 / FU , Maximum methane yield based on VS from digestion of manure

CH4_Fat_Max	(DMC_Fat*Mass_Fat*(VS_Fat)*MY_Fat)	Nm3 / FU , Maximum methane yield based on VS from digestion of frying fats
CH4_OMW_Max	(DMC_OMW*Mass_OMW*(VS_OMW)*MY_OMW)	Nm3 / FU , Maximum methane yield based on VS from digestion of organic municipal waste
CH4_OIW1_Max	(DMC_Animal_Fat*Mass_Animal_Fat*(VS_Animal_Fat)*MY_Animal_Fat)+(DMC_Fish_Waste*Mass_Fish_Waste*(VS_Fish_Waste)*MY_Fish_Waste)+(DMC_Resturant_Waste*Mass_Resturant_Waste*(VS_Resturant_Waste)*MY_Resturant_Waste)+(DMC_Slaughter_Blood*Mass_Slaughter_Blood*(VS_Slaughter_Blood)*MY_Slaughter_Blood)	Nm3 / FU , Maximum methane yield based on VS from digestion of organic industriall waste [Equation part 1]
CH4_OIW2_MAX	(DMC_Slaughter_Entrails*Mass_Slaughter_Entrails*(VS_Slaughter_Entrails)*MY_Slaughter_Entrails)+(DMC_Slaughter_Offal*Mass_Slaughter_Offal*(VS_Slaughter_Offal)*MY_Slaughter_Offal)+(DMC_Diary*Mass_Diary*(VS_Diary)*MY_Diary)	Nm3 / FU , Maximum methane yield based on VS from digestion of organic industriall waste [Equation part 2]
CH4_OIW_MAX	CH4_OIW1_Max+CH4_OIW2_Max+(DMC_Fruit_Vegetable*Mass_Fruit_Vegetable*(VS_Fruit_Vegetable)*MY_Fruit)	Nm3 / FU , Maximum methane yield based on VS from digestion of organic industriall waste [Equation part 3, finished]
CH4_SwSl_Max	DMC_SwSl*Mass_SwSl*(VS_SwSl)*MY_SwSl	Nm3 / FU , Maximum methane yield based on VS from digestion of sewage sludge
CH4_ProductX_Max	DMC_ProductX*Mass_ProductX*(VS_ProductX)*MY_ProductX	Nm3 / FU , Maximum methane yield based on VS from digestion of self defined organic waste substrate
Methane_Max	CH4_Manure_MAX+CH4_Fat_Max+CH4_OMW_Max+CH4_OIW_Max+CH4_SwSl_Max+CH4_ProductX_Max	Nm3 / FU , Maximum methane prodction potential based on VS from digestion of all substrates

Methane_ADpot	(Methane_Manure+Methane_Fat+Methane_OMW+Methane_OIW+Methane_SwSl+Methane_ProductX)*Codigestion_Benefit	Nm3 / FU , Methane prodction potential based on degrability rate of digestion of all substrates
Com	Digestate_Use_Compost	Compost to fit formula "Methane_Storage_pot"
Methane_Storage_pot	((CH4_Manure_MAX-Methane_Manure)*MDF+(CH4_Fat_Max-Methane_Fat)+(CH4_OMW_Max-Methane_OMW)+(CH4_OIW_MAX-Methane_OIW)+(CH4_SwSl_Max-Methane_SwSl)+(CH4_ProductX_Max-Methane_ProductX))*AD_SL*((Digestate_Use_Wet+Com)*CH4_Wet+Digestate_Use_Separated*CH4_Sepa+Digestate_Use_Dry*CH4_Sepa*Storage_Sepa_Dry_CH4)	Nm3 / FU , Methane ptotentially produced in bioresidual storage
Methane_Storage	Methane_Storage_pot*((1-Lr_SC)*SC+(1-SC))*((1-Lr_ST)*ST+(1-ST))	Nm3 / FU , Methane actually produced in bioresidual storage
Methane_Storage_Mass	Methane_Storage*Molar_Mass_CH4	Kg / FU , Methane actually produced in bioresidual storage
Methane_Manure_Storage	(((DMC_Cattle*Mass_Cattle*(VS_Cattle)*MY_Cattle)+(DMC_Pig*Mass_Pig*(VS_Pig)*MY_Pig))*(MDF)*AD_SL*CH4_Undig)	Nm3 / FU , Methane actually produced from manure storage before direct spreading [MDF option]
Methane_Manure_Storage_Mass	Methane_Manure_Storage*Molar_Mass_CH4	Kg / FU , Methane actually produced in manure storage before direct spreading [MDF option]
Methane_Manure_Storage_AD	(Methane_Cattle+Methane_Pig)*Storage_Emission_BD	Nm3 / FU , Methane actually produced from manure storage before anaerobic digestion
Methane_Manure_Storage_AD_Mass	(Methane_Cattle+Methane_Pig)*Storage_Emission_BD*Molar_Mass_CH4	Kg / FU , Methane actually produced in manure storage before direct spreading

Methane_AD	Methane_ADpot*(1-Lr_ADg)*(1-Torch*Bg_pot_CH4_Share)	Nm3 / FU , Methane available afer digestion based on degrability rate of digestion of all substrates
Methane_AD_Lr	Methane_ADpot*(Lr_ADg)*(1-Torch*Bg_pot_CH4_Share)	Nm3 / FU , Methane available afer digestion based on degrability rate of digestion of all substrates
Torch_CH4	Methane_ADpot*(1-Lr_ADg)*(Torch*Bg_pot_CH4_Share)	Nm3 / FU , Methane torched due to over production
Torch_CO2	(Torch_CH4/Bg_pot_CH4_Share)*(1-Bg_pot_CH4_Share)	Nm3 / FU , Carbon dioxide (CO2) torched due to over production
CO2_Manure	Bg_Manure-Methane_Manure	Nm3 / FU , Carbon dioxide (CO2) produced as sum of digested manures
CO2_Fat	Bg_Fat-Methane_Fat	Nm3 / FU , Carbon dioxide (CO2) produced from digestion of frying fats
CO2_OMW	Bg_OMW-Methane_OMW	Nm3 / FU , Carbon dioxide (CO2) produced from digestion of organic municipal waste
CO2_OIW	Bg_OIW-Methane_OIW	Nm3 / FU , Carbon dioxide (CO2) produced as sum digestion of organic industrial waste
CO2_SwSl	Bg_SwSl-Methane_SwSl	Nm3 / FU , Carbon dioxide (CO2) produced from digestion of sewagesludge
CO2_ProductX	Bg_ProductX-Methane_ProductX	Nm3 / FU , Carbon dioxide (CO2) produced from digestion of self defined organic waste substrate
CO2_MAX	(Methane_MAX*(1-CO2_Share_AD))*CO2_Share_AD	Nm3 / FU , Carbon dioxide (CO2) prodction potential based on VS from digestion of all substrates
CO2_ADpot	CO2_Manure+CO2_Fat+CO2_OMW+CO2_OIW+CO2_SwSl+CO2_ProductX*(1-Codigestion_benefit)	Nm3 / FU , Carbon dioxide (CO2) potential based on degrability rate of digestion of all substrates

CO2_Share_AD	CO2_ADpot/(Methane_ADpot+CO2_ADpot)	Numeric % , Carbon dioxide (CO2)content in the biogas produced from anaerobic digestion of potential biogas (Bg_Pot_AD)
CO2_AD	(Methane_AD/(1-CO2_Share_AD))*CO2_Share_AD	Nm3 / FU , Carbon dioxide (CO2) available afer digestion based on degrability rate of digestion of all substrates
CO2_Storage	((Methane_Storage/Bg_CH4_Share)*CO2_Share_AD)	Nm3 / FU ,Carbon dioxide (CO2) actually produced in bioresidual storage
CO2_Storage_Mass	((Methane_Storage/Bg_CH4_Share)*CO2_Share_AD)*Molar_Mass_CO2	Kg / FU , Carbon dioxide (CO2) actually produced in bioresidual storage
Bg_Manure	(Methane_Cattle/CH4_Share_Cattle)+(Methane_Pig/CH4_Share_Pig)	Nm3 / FU , biogas produced from digestion of manure
Bg_Fat	Methane_Fat/CH4_Share_Fat	Nm3 / FU , biogas produced from digestion of fying fats
Bg_OMW	Methane_OMW/CH4_Share_OMW	Nm3 / FU , biogas produced from digestion of organic municipal waste
Bg_OIW1	Methane_Animal_Fat/CH4_Share_Animal_Fat+Methane_Fish_Waste/CH4_Share_Fish_Waste+Methane_Resturant/CH4_Share_Resturant_Waste+Methane_Slaughter_Blood/CH4_Share_Slaughter_Blood+Methane_Slaughter_Entrails/CH4_Share_Slaughter_Entrails	Nm3 / FU , biogas produced from digestion of organic industrial waste [Equation part 1]
Bg_OIW	Bg_OIW1+Methane_Slaughter_Offal/CH4_Share_Slaughter_Offal+Methane_Diary/CH4_Share_Diary+Methane_Fruit_Vegetable/CH4_Share_Fruit_Vegetable	Nm3 / FU , biogas produced from digestion of organic industrial waste [Equation part 2, finished]
Bg_SwSl	Methane_SwSl/CH4_Share_SwSl	Nm3 / FU , biogas produced from digestion of sewage sludge

Bg_ProductX	Methane_ProductX/CH4_Share_ProductX	Nm3 / FU , biogas produced from digestion of defined organic waste substrate
Bg_pot_AD	Methane_ADpot+CO2_ADpot	Nm3 / FU , biogas produced afer digestion based on degrability rate of digestion of all substrates
Bg_pot_CH4_Share	Methane_ADpot/Bg_pot_AD	numeric % of the average methaneshare of the produced raw biogas
Bg_pot_Storage	Methane_Storage+((Methane_Storage/Bg_CH4_Share)*(1-Bg_CH4_Share))	Nm3 / FU , volume of potential biogas produced during storage of bioresidual
Bg_pot_Mass_Storage	(Methane_Storage*Molar_Mass_CH4+((Methane_Storage/Bg_CH4_Share)*(1-Bg_CH4_Share)*Molar_Mass_CO2))*(1-MDF)	Kg / FU , mass of actual biogas produced during storage of bioresidual
Bg_Mass_Manure_Storage	(Methane_Manure_Storage*Molar_Mass_CH4+(Methane_Manure_Storage/(CH4_Share_Cattle*Share_Cattle+CH4_Share_Pig*Share_Pig)*(1-(CH4_Share_Cattle*Share_Cattle+CH4_Share_Pig*Share_Pig)))*Molar_Mass_CO2)*MDF	Kg / FU , mass of actual biogas produced during storage of manure and bioresidual
Bg_Out	Methane_AD+CO2_AD	Nm3 / FU , volume of actually produced raw biogas
Bg_pot_Mass	(Methane_ADpot*Molar_Mass_CH4)+(CO2_ADpot*Molar_Mass_CO2)	Kg / FU , mass of potentially produced biogas
Bg	(Bg_Out*(1-CO2_Share_AD)*(1-Lr_CH4_Cleaning))+(Bg_Out*CO2_Share_AD*(1-Lr_CO2_Cleaning))	Nm3 / FU , volume of cleaned biogas
Bg_CH4_Share	(Bg_Out*(1-CO2_Share_AD)*(1-Lr_CH4_Cleaning))/((Bg_Out*(1-CO2_Share_AD)*(1-Lr_CH4_Cleaning))+(Bg_Out*CO2_Share_AD*(1-Lr_CO2_Cleaning)))	Numeric % methane content of biogas

Bg_Mass	Bg*Bg_Ch4_Share*Molar_Mass_CH4+ Bg*(1- Bg_CH4_Share)*Molar_Mass_CO2	kg / FU , total mass of cleaned biogas
Bm	((Bg_Out*(1-CO2_Share_AD)*(1- Lr_UpT_CH4))+(Bg_Out*CO2_Share_ AD*(1-Lr_UpT_CO2)))*(1- UpT_Cleaning)	kg / FU , total mass of LBG
Bm_CH4_Share	Bg_Out*(1-CO2_Share_AD)*(1- Lr_UpT_CH4)/((Bg_Out*(1- CO2_Share_AD)*(1- Lr_UpT_CH4))+(Bg_Out*CO2_Share_ AD*(1-Lr_UpT_CO2)))	Numeric % methane content of biomethane
Bm_Mass	Bm*Bm_Ch4_Share*Molar_Mass_CH4 +Bm*(1- Bm_CH4_Share)*Molar_Mass_CO2	kg / FU , total mass of LBG
LBG_m3	(Bm*(1-Lr_LBG))	Nm3 / FU , LBG in un-liquified m3 (gasuous uantity) [to compare the loss to biomethane]
LBG_CH4_Share	Bm_CH4_Share	Numeric % methane content of LBG
LBG_l	LBG_m3*l_LBG_Nm3	l / FU , total volume of LBG
LBG_Mass	LBG_m3*LBG_Ch4_Share*Molar_Mas s_CH4+LBG_m3*(1- LBG_CH4_Share)*Molar_Mass_CO2	kg / FU , total mass of LBG
E_Bg_Max	Methane_Max*Energy_Methane	MJ / FU , total energy content in maximal potential biogas
E_Bg	Bg*Bg_Ch4_Share*Energy_Methane	MJ / FU , total energy content in actual biogas produced
E_Bm	Bm*Bm_CH4_Share*Energy_Methane	MJ / FU , total energy content in actual biomethane produced
E_LBG	LBG_m3*Bm_CH4_Share*Energy_Met hane	MJ / FU , total energy content in LBG
E_Bg_Efficiency_F eedstock	E_Bg/E_Feedstock	Numeric % of energy efficiency of energy output in form of produced

		biogas over the feedstock energy content
E_Bg_Max_Efficiency_Feedstock	E_Bg_Max/E_Feedstock	Numeric % of energy efficiency of energy output in form of maximum biogas potential over the feedstock energy content
E_Bm_Efficiency_Feedstock	E_Bm/E_Feedstock	Numeric % of energy efficiency of energy output in form of biomethane over the feedstock energy content
E_LBG_Efficiency_Feedstock	E_LBG/E_Feedstock	Numeric % of energy efficiency of energy output in form of LBG over the feedstock energy content
E_Total_Efficiency_Bg	E_Bg/(((E_Sorting_Tot+E_Pasteurization_e+E_AD_Tot+E_Dewatering_Tot+E_WWT_Tot+E_UpT_or_Cleaning_e+E_Spreading_Tot+E_CHP_Tot)*kWh_To_MJ)+(E_UpT_or_Cleaning_h+Ep_Pt)+E_T_Tot+E_Feedstock)	Numeric % of energy efficiency of energy output in form of biogas over the total energy input
E_Process_Bg	E_Bg/(((E_Sorting_Tot+E_Pasteurization_e+E_AD_Tot+E_Dewatering_Tot+E_WWT_Tot+E_UpT_or_Cleaning_e+E_Spreading_Tot+E_CHP_Tot)*kWh_To_MJ)+(E_UpT_or_Cleaning_h+Ep_Pt)+E_T_Tot)	Numeric % of energy efficiency of energy output in form of biogas over the process energy input to produce and handle the LBG
E_Total_Efficiency_Bm	E_Bm/(((E_Sorting_Tot+E_Pasteurization_e+(Ep_Pt/kWh_to_MJ)+E_AD_Tot+E_Dewatering_Tot+E_WWT_Tot+E_LBG_Compression_Tot+E_UpT_or_Cleaning_e+(E_UpT_or_Cleaning_h/kWh_to_MJ)+E_Tanking_Tot+E_Spreading_Tot)*kWh_To_MJ)+E_Feedstock+E_T_Tot)	Numeric % of energy efficiency of energy output in form of biomethane over the total energy input
E_Process_Bm	E_Bm/(((E_Sorting_Tot+E_Pasteurization_e+(Ep_Pt/kWh_to_MJ)+E_AD_Tot+E_Dewatering_Tot+E_WWT_Tot+E_LBG_Compression_Tot+E_UpT_or_Cleaning_e+(E_UpT_or_Cleaning_h/kWh_to_	Numeric % of energy efficiency of energy output in form of biomethane over the process energy input to produce and handle the LBG

	MJ)+E_Tanking_Tot+E_Spreading_Tot)*kWh_To_MJ)+E_T_Tot)	
E_Total_Efficiency_LBG	E_LBG/(((E_Sorting_Tot+E_Pasteurization_e+(Ep_Pt/kWh_to_MJ)+E_AD_Tot+E_Dewatering_Tot+E_WWT_Tot+E_LBG_Compression_Tot+E_UpT_or_Cleaning_e+(E_UpT_or_Cleaning_h/kWh_to_MJ)+E_Spreading_Tot)*kWh_To_MJ)+E_Feedstock+E_T_Tot)	Numeric % of energy efficiency of energy output in form of LBG over the total energy input
E_Process_LBG	E_LBG/(((E_Sorting_Tot+E_Pasteurization_e+(Ep_Pt/kWh_to_MJ)+E_AD_Tot+E_Dewatering_Tot+E_WWT_Tot+E_LBG_Compression_Tot+E_UpT_or_Cleaning_e+(E_UpT_or_Cleaning_h/kWh_to_MJ)+E_Spreading_Tot)*kWh_To_MJ)+E_T_Tot)	Numeric % of energy efficiency of energy output in form of LBG over the process energy input to produce and handle the LBG
E_Bg_Nm3	E_Bg/Bg	MJ / Nm3 , energy content per normal cubic of biogas
E_Bm_Nm3	E_Bm/Bm	MJ / Nm3 MJ/Nm3 , energy content per normal cubic of biomethane
E_LBG_l	E_LBG/LBG_l	MJ / l LBG , energy dencity of liquid biogas (LBG)
E_Feedstock	HHV_Manure_In+HHV_Fat_In+HHV_OMW_In+HHV_OIW_In+HHV_SwSl_In+HHV_ProductX_In	MJ / FU , total chemical energy contained in the organic waste substates
E_Process	(E_Sorting_Tot+E_Pasteurization_e+(Ep_Pt/kWh_to_MJ)+E_AD_Tot+E_Dewatering_Tot+E_WWT_Tot+E_LBG_Compression_Tot+E_UpT_or_Cleaning_e+(E_UpT_or_Cleaning_h/kWh_to_MJ)+E_Tanking_Tot+E_Spreading_Tot+E_CHP_Tot)*kWh_To_MJ	MJ / FU , total process energy requiered due to fullfillment of the FU
E_Dg_Raw	E_Feedstock-((Methane_ADpot+Methane_Storage)*Energy_Methane)	MJ / FU , total chemical energy remaining in bioresidual after digestion

DM_Cattle	DMC_Cattle*Mass_Cattle	Ton / FU , dry matter cattle manure , balance check
DM_Pig	DMC_Pig*Mass_Pig	Ton / FU , dry matter pig manure , balance check
DM_Manure	DM_Cattle+DM_Pig	Ton / FU , dry mattersum manure , balance check
DM_Fat	DMC_Fat*Mass_Fat	Ton / FU , dry matter frying fat to AD , balance check
DM_OMW	DMC_OMW*Mass_OMW	Ton / FU , dry matter organic municipal waste to AD , balance check
DM_Animal_Fat	DMC_Animal_Fat*Mass_Animal_Fat	Ton / FU , dry matter animal fat to AD , balance check
DM_Fish_Waste	DMC_Fish_Waste*MAss_Fish_Waste	Ton / FU , dry matter fish waste to AD , balance check
DM_Resturant_Waste	DMC_Resturant_Waste*Mass_Resturant_Waste	Ton / FU , dry matter resturant waste to AD , balance check
DM_Slaughter_Blood	DMC_Slaughter_Blood*Mass_Slaughter_Blood	Ton / FU , dry matter blood from slaughterhouse to AD , balance check
DM_Slaughter_Entrails	DMC_Slaughter_Entrails*Mass_Slaughter_Entrails	Ton / FU , dry matter entrails from slaughterhouse to AD , balance check
DM_Slaughter_Offal	DMC_Slaughter_Offal*Mass_Slaughter_Offal	Ton / FU , dry matter offal from slaughterhouse , balance check
DM_Diary	DMC_Diary*Mass_Diary	Ton / FU , dry matter diary products , balance check
DM_Fruit_Vegetable	DMC_Fruit_Vegetable*Mass_Fruit_Vegetable	Ton / FU , dry matter fruit and vegetable waste to AD , balance check
DM_OIW	DM_Animal_Fat+DM_Fish_Waste+DM_Resturant_Waste+DM_Slaughter_Blood+DM_Slaughter_Entrails+DM_Slaughter_Offal+DM_Diary+DM_Fruit_Vegetable	Ton / FU , sum dry matter organic industrial waste to AD , balance check

DM_SwSl	DMC_SwSl*Mass_SwSl	Ton / FU , dry matter sewage sludge to AD , balance check
DM_ProductX	DMC_ProductX*Mass_ProductX	Ton / FU , dry matter self defined substrate to AD , balance check
DM_Balance	DM_Manure+DM_Fat+DM_OMW+DM_OIW+DM_SwSl+DM_ProductX	Warning: If > or < 1; then an error have occured and the resulst are not correct. Dry matter balance.
Dg_Raw_DM	DM_AD-((Bg_Pot_Mass+Bg_pot_Mass_Storage+N_Lost)/1000)	Ton / FU , raw dry mater bioresidual after digestion
Dg_Raw	(Mass_AD-(1-Dg_Raw_DM))	Ton / FU , raw bioresidual after digestion
Dg_Reject_Water_Dry	(Dg_Raw-Dg_Dry)*(Digestate_Use_Separated+Digestate_Use_Dry)+(Dg_Raw-Dg_Compost)*Digestate_Use_Compost	Ton / FU , rejected water by dewatering of bioresidual
Dg_Reject_Water_Out	(Dg_Raw-Dg_Dry-Dg_Reject_Water_Dry*Recycle_W_AD)*(Digestate_Use_Dry+Digestate_Use_Separated)+(Dg_Raw-Dg_Compost-Dg_Reject_Water_Dry*Recycle_W_AD)*Digestate_Use_Compost	Ton / FU , rejected water to WWT by dewatering of bioresidual
Recycled_Water	Dg_Reject_Water_Dry-Dg_Reject_Water_Out	Ton / FU , water recyceled for usage to add water to organic waste substrate feedstock into anaerobic digester
Dg_Reject_DM	(Dg_Raw_DM)*(Lr_DW)*(Digestate_Use_Separated+Digestate_Use_Dry+Digestate_Use_Compost)	Ton / FU , dry matter bioresidual transfered into the water compartment by dewatering bioresidual
Dg_Untreated	Dg_Raw	ton / FU , untreated bioresidual
Dg_Separated	(Dg_Raw-(N_Lost/1000))	ton / FU , separated bioresidual into watery compartment and dewatered bioresidual
Dg_DM	Dg_Raw_DM-Dg_Reject_DM	ton / FU , remaining dry matter after digestion

Dg_Dry	((Dg_Raw_DM-Dg_Reject_DM)/Digestate_Dry_DM)	ton / FU , dewatered bioresidual
Dg_Compost	(((Dg_Raw_DM-Dg_Reject_DM)/Compost_Soil)/Digestate_Dry_DM)	ton / FU , composted bioresidual
P_Manure	P_Cattle*DM_Manure+P_Pig*DM_Pig	kg P / FU , phosporus (P) in manure
P_OIW	P_Animal_Fat*DM_Animal_Fat+P_Fish_waste*DM_Fish_Waste+P_Resturant_Waste*DM_Resturant_Waste+P_Slaughter_Blood*DM_Slaughter_Blood+P_Slaughter_Entrails*DM_Slaughter_Entrails+P_Slaughter_Offal*DM_Slaughter_Offal+P_Diary*DM_Diary+P_Fruit_Vegetable*DM_Fruit_Vegetable	kg P / FU , phosporus (P) in organic waste
P_Out	P_Manure*(1-MDF)+(P_Fat*DM_Fat+P_OMW*DM_OMW+P_OIW+P_SwSl*DM_SwSl+ProductX*DM_ProductX)	kg P / FU , phosporus (P) in bioresidual
P_Water	P_Out*(1-P_DM)	kg P / FU , phosporus (P) in watery compartment of bioresidual
P_DW_WWT	(((Dg_Reject_Water_Dry/(Dg_Raw-Dg_Raw_DM)))*P_Water)+(P_Out*Lr_Dw*P_DM)	kg P / FU , phosporus (P) in watery compartment from dewatered bioresidual
P_DW_Dg	P_Out-P_DW_WWT	kg P / FU , phosporus (P) in dewatered bioresidual
P_Available	(((P_Out*Digestate_Use_Wet)+(P_DW_Dg*(Digestate_Use_Dry+Digestate_Use_Compost))+((P_DW_WWT+P_DW_Dg)*Digestate_Use_Separated))*Digestate_Ava_P*(1-Incineration)+(MDF*P_Manure*Manure_Ava_P))*(1-Pl_Avg)	kg P / FU , total available phosporus (P) in all biofertilizers
N_Manure	(N_Cattle*DM_Cattle+N_Pig*DM_Pig)*N_Adjustment	kg N (manure) / FU , nitrogen (N) found in the manure

N_OIW	N_Animal_Fat*DM_Animal_Fat+N_Fish_waste*DM_Fish_Waste+N_Resturant_Waste*DM_Resturant_Waste+N_Slaughter_Blood*DM_Slaughter_Blood+N_Slaughter_Entrails*DM_Slaughter_Entrails+N_Slaughter_Offal*DM_Slaughter_Offal+N_Diary*DM_Diary+N_Fruit_Vegetable*DM_Fruit_Vegetable	kg N (waste) / FU , nitrogen (N) found in the industrial organic waste
N_Waste	(N_Fat*DM_Fat+N_OMW*DM_OMW+N_OIW+N_SwSl*DM_SwSl+N_ProductX*DM_ProductX)*N_Adjustment	kg N (waste) / FU , total nitrogen (N) found in the organic waste
N2_dig	1-(NH3_dig+N2O_dig)	numeric % of the N2 produsedalong with NH3 and N2O for the gasous N losses.
NH3_Manure_Dig	(((N_Manure*Nl_Manure_Dig*NH3_Dig)/N_NH3)*Storage_Digested_NH3*(1-(SC*Lr_SC+ST*Lr_St))*(NH3_Inhi*(1-NH3_Red)+(1-NH3_Inhi))+((N_Manure*Nl_Manure_Dig*NH3_Dig)/N_NH3)*(1-Storage_Digested_NH3)*(Sp_Bat*(1-Spr_NH3)+(1-SP_BAT)))*Digestate_Use_wet*(1-MDF)	kg NH3 / FU , ammonia (NH3) produced from untreated bioresidual from manure
N2O_Manure_Dig	(((N_Manure*(1-MDF)*Nl_Manure_Dig*N2O_Dig)/N_N2O)*Storage_Digested_N2O+((N_Manure*(1-MDF)*Nl_Manure_Dig*N2O_Dig)/N_N2O)*(1-Storage_Digested_N2O))*(Digestate_Use_Wet)	kg N2O / FU , dinitrogen monoxide (N2O) produced from untreated bioresidual from manure
N2_Manure_Dig	((N_Manure*(1-MDF)*Nl_Manure_Dig*N2_Dig)/N_N2)*(Digestate_Use_Wet)	kg N2 / FU , dinitrogen (N2) produced from untreated bioresidual from manure

Nl_Manure_Dige	NH3_Manure_Dig*N_NH3+N2O_Manure_Dig*N_N2O+N2_Manure_Dig*N_N2	kg N / FU , total nitrogen (N) lost due to gassous emissions from untreated bioresidual from manure
NH3_Manure_Untreated	(((N_Manure*Nl_Manure*NH3_Dig)/N_NH3)*Nl_Undig_NH3*Storage_Untreated_NH3*(1-(SC*Lr_SC+ST*Lr_St))*(NH3_Inhi*(1-NH3_Red)+(1-NH3_Inhi))+((N_Manure*Nl_Manure*NH3_Dig)/N_NH3)*Nl_Undig_NH3*(1-Storage_Untreated_NH3)*(Sp_Bat*(1-Spr_NH3)+(1-SP_BAT)))*MDF	kg NH3 / FU , ammonia (NH3) produced from untreated manure
N2O_Manure_Untreated	(((N_Manure*Nl_Manure*N2O_Dig)/N_N2O)*Nl_Undig_N2O*Storage_Untreated_N2O+((N_Manure*Nl_Manure*N2O_Dig)/N_N2O)*Nl_Undig_N2O*(1-Storage_Untreated_N2O))*MDF	kg N2O / FU , dinitrogen monoxide (N2O) produced from untreated manure
N2_Manure_Untreated	((N_Manure*(MDF)*Nl_Manure*N2_Dig)/N_N2)*Nl_Undig_N2	kg N2 / FU , dinitrogen (N2) produced untreated manure
Nl_Manure_Untreated	NH3_Manure_Untreated*N_NH3+N2O_Manure_Untreated*N_N2O+N2_Manure_Untreated*N_N2	kg N / FU , nitrogen (N) lost from untreated manure
NH3_Manure_Sepa_Wet	((((N_Manure*Nl_Manure_Dig*NH3_Dig)/N_NH3)*Nl_Sepa_NH3)*Storage_Sepa_Wet_NH3*(1-(SC*Lr_SC+ST*Lr_St))*(NH3_Inhi*(1-NH3_Red)+(1-NH3_Inhi))+(((N_Manure*Nl_Manure_Dig*NH3_Dig)/N_NH3)*Nl_Sepa_NH3)*App_Sepa_Wet_NH3)*(Digestate_Use_Separated)*(1-MDF)	kg NH3 / FU , ammonia (NH3) produced from watery comparment of dewatered bioresidual of organic waste excluded manure
N2O_Manure_Sepa_Wet	((((N_Manure*Nl_Manure_Dig*N2O_Dig)/N_N2O)*Nl_Sepa_N2O)*Storage_Sepa_Wet_N2O+(((N_Manure*Nl_Manure_Dig*N2O_Dig)/N_N2O)*Nl_Sepa_N2O)*App_Sepa_Wet_N2O)*(Digestate_Use_Separated)*(1-MDF)	kg N2O / FU , dinitrogen monoxide (N2O) produced from watery compartment of dewatered bioresidual from manure

N2_Manure_Sepa_Wet	((N_Manure*Nl_Manure_Dig*N2_Dig)/N_N2)*Nl_Undig_N2*(Digestate_Use_Separated)*(1-MDF)*0	kg N2 / FU , dinitrogen (N2) produced from watery compartment of dewatered bioresidual from manure
Nl_Manure_Sepa_Wet	NH3_Manure_Sepa_Wet*N_NH3+N2O_Manure_Sepa_Wet*N_N2O+N2_Manure_Sepa_Wet*N_N2	kg N / FU , nitrogen (N) lost from watery compartment of dewatered bioresidualfrom manure
NH3_Manure_Sepa_Dry	((((N_Manure*Nl_Manure_Dig*NH3_Dig)/N_NH3)*Nl_Sepa_NH3)*Storage_Sepa_Dry_NH3*(1-(SC*Lr_SC+ST*Lr_St))*(NH3_Inhi*(1-NH3_Red)+(1-NH3_Inhi))+(((N_Manure*Nl_Manure_Dig*NH3_Dig)/N_NH3)*Nl_Sepa_NH3)*App_Sepa_Dry_NH3)*(1-Digestate_Use_Wet)*(1-MDF)	kg NH3 / FU , ammonia (NH3) produced from dewatered bioresidual from manure
N2O_Manure_Sepa_Dry	((((N_Manure*Nl_Manure_Dig*N2O_Dig)/N_N2O)*Nl_Sepa_N2O)*Storage_Sepa_Dry_N2O+(((N_Manure*Nl_Manure_Dig*N2O_Dig)/N_N2O)*Nl_Sepa_N2O)*App_Sepa_Dry_N2O)*(1-Digestate_Use_Wet)*(1-MDF)	kg N2O / FU , dinitrogen monoxide (N2O) produced from dewatered bioresidual from manure
N2_Manure_Sepa_Dry	((N_Manure*Nl_Manure_Dig*N2_Dig)/N_N2)*Nl_Undig_N2*(1-Digestate_Use_Wet)*(1-MDF)	kg N2 / FU , dinitrogen (N2) produced from dewatered bioresidual from manure
Nl_Manure_Sepa_Dry	(NH3_Manure_Sepa_Dry*N_NH3)+(N2O_Manure_Sepa_Dry*N_N2O)+(N2_Manure_Sepa_Dry*N_N2)	kg N / FU , nitrogen (N) lost from dewatered bioresidual from manure
Nl_Manure_Sepa	Nl_Manure_Sepa_Dry+Nl_Manure_Sepa_Wet	kg N / FU , total nitrogen (N) lost due to separation of bioresidual from manure
NH3_Waste_Dig	(((N_Waste*Nl_Waste*NH3_Dig)/N_NH3)*Storage_Digested_NH3*(1-(SC*Lr_SC+ST*Lr_St))*(NH3_Inhi*(1-NH3_Red)+(1-NH3_Inhi))+((N_Waste*Nl_Waste*NH3_Dig)/N_NH3)*(1-Storage_Digested_NH3)*(Sp_Bat*(1-	kg NH3 / FU , ammonia (NH3) produced from untreated bioresidual from organic waste, excluded manure

	Spr_NH3)+(1-SP_BAT)))*Digestate_Use_wet	
N2O_Waste_Dig	(((N_Waste*Nl_Waste*N2O_Dig)/N_N2O)*Storage_Sepa_Dry_N2O+((N_Waste*Nl_Waste*N2O_Dig)/N_N2O)*App_Sepa_Dry_N2O)*Digestate_Use_wet	kg N2O / FU , dinitrogen monoxide (N2O) produced from untreated bioresidual from organic waste, excluded manure
N2_Waste_Dig	((N_Waste*Nl_Waste*N2_Dig)/N_N2)*Digestate_Use_wet	kg N2 / FU , dinitrogen (N2) produced from untreated bioresidual from organic waste, excluded manure
Nl_Waste_Dig	NH3_Waste_Dig*N_NH3+N2O_Waste_Dig*N_N2O+N2_Waste_Dig*N_N2	kg N / FU , total nitrogen (N) lost due to gassous emissions from untreated bioresidual from organic waste, excluded manure
NH3_Waste_Sepa_Wet	((((N_Waste*Nl_Waste*NH3_Dig)/N_NH3)*Nl_Sepa_NH3)*Storage_Sepa_Wet_NH3*(1-(SC*Lr_SC+ST*Lr_St))*(NH3_Inhi*(1-NH3_Red)+(1-NH3_Inhi))+((App_Sepa_Wet_NH3*(N_Waste*Nl_Waste*NH3_Dig)/N_NH3)*Nl_Sepa_NH3)*(Sp_Bat*(1-Spr_NH3)+(1-SP_BAT)))*Digestate_Use_Separated	kg NH3 / FU , ammonia (NH3) produced from watery comparment of dewatered bioresidual of organic waste excluded manure
N2O_Waste_Sepa_Wet	((((N_Waste*Nl_Waste*N2O_Dig)/N_N2O)*Nl_Sepa_N2O)*Storage_Sepa_Dry_N2O+(((N_Waste*Nl_Waste*N2O_Dig)/N_N2O)*Nl_Sepa_N2O)*App_Sepa_Dry_N2O)*(Digestate_Use_Separated)	kg N2O / FU , dinitrogen monoxide (N2O) produced from watery compartment of dewatered bioresidual of organic waste, excluded manure
N2_Waste_Sepa_Wet	(((N_Waste*Nl_Waste*N2_Dig)/N_N2)*Nl_Sepa_N2)*(Digestate_Use_Separated)*0	kg N2 / FU , dinitrogen (N2) produced from watery compartment of dewatered bioresidual of organic waste, excluded manure
Nl_Waste_Sepa_Wet	NH3_Waste_Sepa_Wet*N_NH3+N2O_Waste_Sepa_Wet*N_N2O+N2_Waste_Sepa_Wet*N_N2*(Digestate_Use_Separated)	kg N / FU , nitrogen (N) lost from watery compartment of dewatered bioresidual of organic waste, excluded manure

NH3_Waste_Sepa_Dry	((((N_Waste*Nl_Waste*NH3_Dig)/N_NH3)*Nl_Sepa_NH3)*Storage_Sepa_Dry_NH3*(1-(SC*Lr_SC+ST*Lr_St))*(NH3_Inhi*(1-NH3_Red)+(1-NH3_Inhi))+(((N_Waste*Nl_Waste*NH3_Dig)/N_NH3)*Nl_Sepa_NH3)*App_Sepa_Dry_NH3*(Sp_Bat*(1-Spr_NH3)+(1-SP_BAT)))*(1-Digestate_Use_Wet)*(1-MDF)	kg NH3 / FU , ammonia (NH3) produced from dewatered bioresidual of organic waste excluded manure
N2O_Waste_Sepa_Dry	((((N_Waste*Nl_Waste*N2O_Dig)/N_N2O)*Nl_Sepa_N2O)*Storage_Sepa_Dry_N2O+(((N_Waste*Nl_Waste*N2O_Dig)/N_N2O)*Nl_Sepa_N2O)*App_Sepa_Dry_N2O)*(1-Digestate_Use_Wet)*(1-MDF)	kg N2O / FU , dinitrogen monoxide (N2O) produced from dewatered bioresidual of organic waste, excluded manure
N2_Waste_Sepa_Dry	(((N_Waste*Nl_Waste*N2_Dig)/N_N2)*Nl_Sepa_N2)*(1-Digestate_Use_Wet)*(1-MDF)	kg N2 / FU , dinitrogen (N2) produced from dewatered bioresidual of organic waste, excluded manure
Nl_Waste_Sepa_Dry	NH3_Waste_Sepa_Dry*N_NH3+N2O_Waste_Sepa_Dry*N_N2O+N2_Waste_Sepa_Dry*N_N2*(1-MDF)	kg N / FU , nitrogen (N) lost from dewatered bioresidual of organic waste, excluded manure
Nl_Waste_Sepa	Nl_Waste_Sepa_wet+Nl_Waste_Sepa_Dry	kg N / FU , total nitrogen (N) lost due to separation of bioresidual from organic waste excluded manure
N_in	N_Waste+N_Manure	kg N (in) / FU , total nitrogen (N) in organic waste substrates per FU
N_Manure_Out	(N_Manure-(Nl_Manure_Sepa_Wet+Nl_Manure_Sepa_Dry+Nl_Manure_Dige))*(1-MDF)	kg N (manure) / FU , remaining nitrogen (N) in bioresidual from manure after digestion
N_Waste_Out	N_Waste-(Nl_Waste_Dig+Nl_Waste_Sepa_Wet+Nl_Waste_Sepa_Dry)*(1-Incineration)	kg N (waste) / FU , remaining nitrogen (N) in bioresidual from organic waste after digestion
N_MDF	(N_Manure-Nl_Manure_Untreated)*MDF	kg N (manure) / FU , remaining nitrogen (N) in manure that is being applied directly as fertilizer

N_Lost	N_Waste-N_Waste_Out+(N_Manure-N_Manure_Out)*(1-MDF)+(Nl_Manure_Untreated*MDF)	kg N (tot, lost) / FU , total Nitrogen lost during anaerobic digestion, storage of bioresidual / manure and during spreading
N_Water	(N_Waste_Out+N_Manure_Out)*(1-N_DM)	kg N / FU , total nitrogen (N) in wet matter bioresidual , excluded dry matter = only water compartment
N_DW_WWT	(Dg_Reject_Water_Dry/(Dg_Raw-Dg_Raw_DM))*(N_Water)+((Dg_Reject_DM/Dg_Raw_DM)*N_Dry)	kg N / FU , total nitrogen (N) in wet matter bioresidual , included dry matter lost to water fraction by dewatering process
N_Dry	(N_Waste_Out+N_Manure_Out)*(N_DM)	kg N / FU , total nitrogen (N) in dry matter (DM) bioresidual
N_Dw_Dg	((N_Waste_Out+N_Manure_Out)-N_DW_WWT)	kg N / FU , total nitrogen (N) in dewatered bioresidual
N_Available	(N_MDF*Manure_ava_N)+(((N_Dw_Dg)*Digestate_ava_N*(Digestate_Use_Dry+Digestate_Use_Compost)+(N_Dw_Dg+N_DW_WWT)*(Digestate_Use_Separated+Digestate_Use_Wet)*Digestate_ava_N))*(1-HM_3)	kg N (available) / FU , total available nitrogen (N) in biofertilizer
N_Unavailable_Out	(N_MDF)+(((N_Dw_Dg)*(Digestate_Use_Dry+Digestate_Use_Compost)+(N_Dw_Dg+N_DW_WWT)*(Digestate_Use_Separated+Digestate_Use_Wet)))*(1-HM_3)	kg N (out) / FU , total output of nitrogen (N) for bioresidual
N_Water_Dry_Balance	(N_Water+N_Dry)-(N_DW_WWT+N_Dw_Dg)	Warning: If > or < 0; then an error have occured and the resulst are not correct. Nitrogen balance of dewatered bioresidual
Lr_UpT_CO2	Lr_CO2_UpT_WS*UpT_WS+Lr_CO2_UpT_ChS*UpT_ChS+Lr_CO2_UpT_PSA*UpT_PSA+Lr_CO2_UpT_Membrane*UpT_Membrane+Lr_CO2_UpT_Cyrogenic*UpT_Cyrogenic	Numeric % of loss rate of carbon dioxide (CO2) due to the applied biogas treatment technology

Lr_UpT_CH4	Lr_CH4_UpT_WS*UpT_WS+Lr_CH4_UpT_ChS*UpT_ChS+Lr_CH4_UpT_PSA*UpT_PSA+Lr_CH4_UpT_Membrane*UpT_Membrane+Lr_CH4_UpT_Cyrogenic*UpT_Cyrogenic	Numeric % of loss rate of methane (CH4) due to the applied biogas treatment technology
Ep_Pt_Initial	SHC*(1+Lr_HP)*(PH-FSH)*Mass_AD	MJ / FU , heat demand for the pasteurization per FU before heat recovery
Ep_Pt	(SHC*(1+Lr_HP)*(PH-AFSH)*Mass_AD)	MJ / FU , heat demand for the pasteurization per FU after heat recovery
Energy_e_CHP	(E_Bg*Energy_Efficiency_e_CHP)/kWh_to_MJ	MJ / FU , electricity output from CHP biogas
Energy_H_CHP	E_Bg*Energy_Efficiency_H_CHP	MJ / FU , heat output from CHP biogas
E_Sorting_Tot	Mass_OIW+Mass_OMW*E_Sorting	kWh / FU , electricity demand for the total requiered sorting of OMW and OIW per FU , removal of inorganic materials
E_Pasteurization_e	(Mass_Pt+Water_to_Pretreatment)*E_Pasteurization	kWh / FU , electricity demand for the pasteurization per FU
E_AD_Tot	Mass_AD*E_AD	kWh / FU , elelcticity demand for the anaerobic digester per FU
E_Dewatering_Tot	Water_Dw+(Dg_Reject_Water_Out)*E_Dewatering	kWh / FU , electricity demand for total dewatering requiered per FU
E_WT_Tot	E_WT*Water_To_Pretreatment+I_Water_In*0	kWh / FU , electricity demand for water production to cover water demand per FU
E_WWT_Tot	Water_Dw+(Dg_Reject_Water_Out)*E_WWT	kWh/ FU , lecticity demand for treatment of waste water per FU
E_LBG_Compression_Tot	((LBG_Use_Bus+LBG_Use_Gasoline_car+LBG_Use_Diesel_car)*E_LBG_Process*Bg)+(E_Compression200*Bm_Compression200+E_Compression300*Bm_C	kWh / FU , electricity demand for compression per FU

	ompression300+E_Compression45_50*Bm_Compression45_50)*Bm	
E_UpT_or_Cleaning_e	(E_UpT_WS*UpT_WS+E_UpT_ChS*UpT_ChS+E_UpT_PSA*UpT_PSA+E_UpT_Membrane*UpT_Membrane+E_UpT_Cyrogenic*UpT_Cyrogenic)*Bg	kWh/ FU , electricity reguiered to clean or upgrae the biogas per FU
E_UpT_or_Cleaning_H	(H_UpT_ChS*UpT_ChS*Bg)/kWh_to_MJ	kWh / FU , electricity reguiered to clean or upgrae the biogas per FU
E_Tanking_Tot	Bm*E_Tanking*(Biomethane_Use_Bus+Biomethane_Use_Gasoline_Car+Biomethane_Use_Diesel_Car)	kWh / FU , electricity consumption by tanking of biofuel per FU
E_Spreading_Tot	(Digestate_Use_Dry*Dg_Dry*E_Spreading_Dry)+(Digestate_Use_Wet*Dg_Raw+Digestate_Use_Separated*Dg_Raw+Digestate_USe_Compost*Dg_Compost)*E_Spreading_Wet	kWh / FU , energy consumption, undefined, for spreading bioresidual to field per FU
E_CHP_Tot	E_CHP*E_Bg*UpT_Cleaning*kWh_to_MJ	MJ / FU , electricity requiered in the CHP plant per FU
E_Incineration_Tot	E_Incineration*((Mass_Manure-(Bg_Mass_Manure_Storage/1000))*(1-MDF)*(Incineration)+(Mass_Fat+Mass_OMW+Mass_OIW+Mass_SwSl+Mass_ProductX))	kWh Incineration: Turn on by applying 1, then turn off the other by giving them value 0
Bm_Bus	E_Bm/E_Use_Bus	total km possible to drive by using biomethane produced due to treatment of FU
LBG_Bus	E_LBG/E_Use_Bus	total km possible to drive by using LBG produced due to treatment of FU
HHV_Manure_In	(HHV_Cattle*Share_Cattle+HHV_Pig*Share_Pig)*Manure*DM_AD-(Bg_Mass_Manure_Storage/1000)	MJ / FU , energy output form incineration of manure in terms of HHV
HHV_Fat_In	HHV_Fat*Fat*DM_AD	MJ / FU , energy output form incineration of frying fats in terms of HHV

HHV_OMW_In	HHV_OMW*OMW*DM_AD	MJ / FU , energy output form incineration of organic municipal waste in terms of HHV
HHV_OIW_In1	(HHV_Animal_Fat*Share_Animal_Fat+HHV_Fish_Waste*Share_Fish_Waste+HHV_Resturant_Waste*Share_Resturant_Waste+HHV_Slaughter_Blood*Share_Slaughter_Blood+HHV_Slaughter_Entrails*Share_Slaughter_Entrails+HHV_Slaughter_Offal*Share_Slaughter_Offal)*OIW*DM_AD	MJ / FU , energy output form incineration of organic industrial waste in terms of HHV (Equation part 1)
HHV_OIW_In	HHV_OIW_In1+(HHV_Diary*Share_Diary+HHV_Fruit_Vegetable*Share_Fruit_Vegetable)*OIW*DM_AD	MJ / FU , energy output form incineration of organic industrial waste in terms of HHV (Equation part 2, final)
HHV_SwSl_In	HHV_SwSl*SwSl*DM_AD	MJ / FU , energy output form incineration of sewage sludge in terms of HHV
HHV_ProductX_In	HHV_ProductX*ProductX*DM_AD	MJ / FU , energy output form incineration of self determined organic waste substrate in terms of HHV
HHV_Incineration_Raw	HHV_Manure_In*(1-MDF)+HHV_Fat_In+HHV_OMW_In+HHV_OIW_In+HHV_SwSl_In+HHV_ProductX_In	MJ / FU , energy output form incineration in terms of HHV (excluded energy loss due to heating of water)
HHV_Incineration	HHV_Incineration_Raw-I_E_Water	MJ / FU , energy output form incineration in terms of LHV (included energy loss due to heating of water)
I_Spreading	(Mass_Manure-(Bg_Mass_Manure_Storage/1000))*MDF	Ton / FU , spreading of manure , incneration scenario
I_Mass	Mass_Pt+I_Water_To	Ton / FU , total mass of organic waste to incineration
I_Water_To	((DM_Balance-(Manure*MDF))/(1-OWc_I))-Mass_Pt	Ton / FU , Mass balance of water based on an optimal water content for incineration (OWc_I)

I_Water_In	Iff(I_Water_To<0, 0, I_Water_To)	Ton / FU , water to watering of organic waste to incineration
I_Water_Out	Iff(-I_Water_To<0, 0, -I_Water_To)	Ton / FU , water form dewatering of organic waste to incineration
I_E_Dewatering	I_Water_Out*E_Dewatering	kWh (e-) / FU , energy requirement for dewatering of organic waste to incineration
I_E_WWT	I_Water_Out*E_WWT	kWh (e-) / FU , energy requirement for waste water treatment of water from dewatering before incineration
I_E_WT	I_Water_In*E_WT	kWh (e-) / FU , energy requirement for water treatment of water to incineration
I_E_Spreading	(Mass_Manure-(Bg_Mass_Manure_Storage/1000))*E_Spreading_Wet	kWh / FU , spreading of manure , incneration scenario
I_E_Water	SHC*(((Mass_Pt)+(I_Water_In-I_Water_Out))*(Temp_Incinerated_Water-FSH))+(HV_W*((Mass_Pt)+(I_Water_In-I_Water_Out)))	MJ lost due to heating of feedstock water in the incinerator
I_E_e	(HHV_Incineration*Energy_Efficiency_e_Incineration)/kWh_to_MJ	kWh (e-) / FU , output elecricity energy from incineration plant per FU
I_E_H	HHV_Incineration*Energy_Efficiency_H_Incineration	MJ (H) / FU , output heat energy from incineration plant per FU
I_E_Total_Efficiency	(I_E_H+(I_E_e*kWh_to_MJ))/E_Feedstock	numeric % efficiency rate of incineration , energy output / feedstock energy incineration
Ash_Manure	((1-VS_Cattle)*Share_Cattle+(1-VS_Pig*Share_Pig))*Manure*DM_AD	ton / FU , ash after incineration of manure per FU
Ash_Fat	(1-VS_Fat)*Fat*DM_AD	ton / FU , ash after incineration of fat per FU
Ash_OMW	(1-VS_OMW)*OMW*DM_AD	ton / FU , ash after incineration of Organic municipal waste per FU

Ash_OIW1	((1-VS_Animal_Fat)*Share_Animal_Fat+(1-VS_Fish_Waste)*Share_Fish_Waste+(1-VS_Resturant_Waste)*Share_Resturant_Waste+(1-VS_Slaughter_Blood)*Share_Slaughter_Blood+(1-VS_Slaughter_Entrails)*Share_Slaughter_Entrails+(1-VS_Slaughter_Offal)*Share_Slaughter_Offal)*OIW*DM_AD	ton / FU , ash after incineration of organic industrial waste (Euation part 1)
Ash_OIW	Ash_OIW1+((1-VS_Diary)*Share_Diary+(1-VS_Fruit_Vegetable)*Share_Fruit_Vegetable)*OIW*Share_Fly_Ash*DM_AD	ton / FU , ash after incineration of organic industiral waste (Equation part 2, finished) per FU
Ash_SwSl	(1-VS_SwSl)*SwSl*DM_AD	ton / FU , ash after incineration sewage sludge per FU
Ash_ProductX	(1-VS_ProductX)*ProductX*DM_AD	ton / FU , ash after incineration
Ash_Incineration	Ash_Manure+Ash_Fat+Ash_OMW+Ash_OIW+Ash_SwSl+Ash_ProductX	ton / FU , ash after incineration
Fly_Ash	Ash_Incineration*Share_Fly_Ash	ton / FU , ash that becomes fly ash
Bottom_Ash	Ash_Incineration*Share_Bottom_Ash	ton / FU , ash that becomes bottom ash
T_Manure	(Mass_Manure*km_Manure*(1-MDF))	MJ / FU , tansported manure
T_Fat	Mass_Fat*km_Fat	MJ / FU , tansported frying fat
T_OMW	km_OMW*Mass_OMW_Inorganic	MJ / FU , tansported organic municipal waste
T_OIW	km_OIW*Mass_OIW_Inorganic	MJ / FU , tansported organic industrial waste
T_SwSl	km_SwSl*Mass_SwSl	MJ / FU , tansported sewage sludge
T_ProductX	Mass_ProductX*km_ProductX	MJ / FU , tansported self defined organic waste

T_Feedstock	T_Manure+T_Fat+T_OMW+T_OIW+T_SwSl+T_ProductX	tkm / FU , transport requiered for the transport of all the feedstock organic wastes per FU
E_T_Feedstock	(T_Feedstock-T_OMW)*E_Transport_EUR5+(T_OMW*E_Transport_Municipal_Collection)	MJ / FU , energy requiered for the transport of all the feedstock organic wastes per FU
T_Dg_Wet	km_Dg*Dg_Raw*Digestate_Use_Wet*(1-Incineration)	tkm / FU , transport of untreated bioresidual
T_Dg_Dry	km_Dg*Dg_Dry*Digestate_Use_Dry*(1-Incineration)	tkm / FU , transport of dewatered bioresidual
T_Dg_Separated	km_Dg*Dg_raw*Digestate_Use_Separated*(1-Incineration)	tkm / FU , transport of both the dewatered bioresidual and the water part of dewatered bioredidual
T_Compost	km_Compost*Dg_Compost*Digestate_Use_Compost*(1-Incineration)	tkm / FU , transport of composted bioresidual
T_Bm	(km_Bm_LBG*(Bm_Mass/1000+((Bm_Mass/1000)*11.5)))*(Biomethane_Use_Bus+Biomethane_Use_Gasoline_Car+Biomethane_Use_Diesel_Car)*(1-Incineration)	tkm / FU , sum of bioresidual transport to fulfill the FU. The biomethane mass is devided by 1000 to go from kg to ton, plus 11.5 ton emty gas tanks per ton biomethane transported, Jørgensen. R (2015)
T_LBG	(LBG_Mass*km_Bm_LBG)/1000*(LBG_Use_Bus+LBG_Use_Gasoline_Car+LBG_Use_Diesel_Car)	tkm / FU , sum of Liquid biogas (LBG) transport to fulfill the FU
E_Biofuel	(T_BM+T_LBG)*E_Transport_EUR5	tkm / FU , sum of bioresidual transport to fulfill the FU
E_Bioresidual	(T_Dg_Wet+T_Dg_Dry+T_Dg_Separated+T_Compost)*E_Transport_EUR5	MJ / FU , sum of bioresidual transport to fulfill the FU
T_End_Product	T_Dg_Wet+T_Dg_Dry+T_Dg_Separated+T_Compost+T_Bm+T_LBG	tkm / FU , mass * distanse = sum of all transport of end-products to fulfill the FU
E_T_End_Product	T_End_Product*E_Transport_EUR5	MJ / FU , sum of all transport of end-products to fulfill the FU

T_Tot	T_Feedstock+T_End_Product	tkm / FU , sum of all transport to fulfill the FU
E_T_Tot	E_T_Feedstock+E_T_End_Product	MJ / FU , sum of all energy requiered by transport to fulfill the FU
T_Fly_Ash	km_Fly_ash*Fly_Ash	tkm / FU , transport of fly ash
T_Bottom_Ash	km_Bottom_ash+Bottom_Ash	tkm / FU , transport of bottom ash
E_T_Ash	(T_Fly_Ash+T_Bottom_Ash)*E_Transport_EUR5	MJ / FU , sum of bottom and fly ash transport