CHALMERS
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The Future of Food Waste

LCA of existing and emerging routes to bio-based chemicals

Master's thesis within the Industrial Ecology programme

ERICA CARLSSON

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Photograph of food waste, taken by sporkist, 2006.

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ABSTRACT

There are large quantities of food waste to handle in society. Food waste can be used as a renewable feedstock to produce bio-based platform chemicals such as succinic acid. The goal of this thesis is to compare the environmental impact of three options for food waste management and/or production of bio-based succinic acid. These options were production of biogas from food waste, production of succinic acid from food waste and production of succinic acid from corn. The aim was to evaluate which of these options is the preferred one from an environmental point of view when it comes to managing food waste respectively producing bio-based succinic acid. A cradle-to-gate Life Cycle Assessment (LCA) of the production of biogas and succinic acid from food waste was performed. The results were compared to published LCA results of succinic acid production from corn.

The results show that production of biogas is an environmentally better option for food waste management than production of succinic acid. If food waste or corn is the best feedstock to use for succinic acid production from an environmental point of view depends on the modelling choices. When no impact from food production was included and mass allocation was used, food waste was a better option than corn. If economic or no allocation was used or if the impact from food production was included, corn was a better option than food waste.

Besides the used allocation method and inclusion of emissions from food production, the impact results were also affected by the assumed yield in the recovery process for succinic acid and the enzyme use. The results of this study show that the environmental impacts of producing succinic acid from food waste and corn are in the same range. This can be seen as a motivation to proceed with further environmental investigations.

Key words: Life Cycle Assessment, Food waste, Biogas, Succinic acid, Bio-based, Chemical, Bio-refinery

Framtiden för matavfall
LCA av existerande och kommande vägar till biobaserade kemikalier
Examensarbete inom masterprogrammet *Industriell Ekologi*
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SAMMANFATTNING

Det finns stora mängder matavfall att hantera i samhället. Matavfall kan användas som ett förnyelsebart råmaterial för tillverkning av bio-baserade plattformskemikalier såsom bärnstenssyra. Målet med denna studie är att jämföra miljöpåverkan av tre alternativ för att hantera matavfall och/eller att producera bio-baserad bärnstenssyra. Dessa alternativ var produktion av biogas från matavfall, produktion av bärnstenssyra från matavfall samt produktion av bärnstenssyra från majs. Syftet var att utvärdera vilket av dessa alternativ som är att föredra ur miljösynpunkt för att hantera matavfall respektive producera bio-baserad bärnstenssyra. En livscykelanalys från vagga till grind genomfördes för produktion av biogas och bärnstenssyra från matavfall. Resultaten jämfördes med publicerade resultat från livscykelanalyser för produktion av bärnstenssyra från majs.

Resultaten visade att produktion av biogas är att föredra ur miljösynpunkt för att hantera matavfall jämfört med att producera bärnstenssyra. Om matavfall eller majs är det miljömässigt bästa råmaterialet för att producera bärnstenssyra beror på hur modelleringen utförs. När ingen miljöpåverkan från matproduktion inkluderades samt massallokering användes var matavfall att föredra framför majs. När ekonomisk allokering eller ingen allokering användes eller när miljöpåverkan från matproduktion inkluderades var majs att föredra framför matavfall.

Utöver allokeringsmetod och inkludering av utsläpp från matproduktion så påverkades resultaten av antaget utbyte i uppgraderingsprocessen för bärnstenssyra samt användningen av enzym. Resultaten av denna studie visar att miljöpåverkan av att producera bärnstenssyra från matavfall och majs är inom samma spann. Detta kan ses som en motivering för att fortsätta med vidare utredning av miljöpåverkan.

Nyckelord: Livscykelanalys, Matavfall, Biogas, Bärnstenssyra, Bio-baserad, Kemikalie, Bio-raffinaderi

Contents

Preface

This study was carried out by Erica Carlsson at SP Technical Research Institute of Sweden. It was performed between January and June 2016 as a master thesis at the division of Environmental Systems Analysis, department of Energy and Environment at Chalmers University of Technology. Supervisors at SP Technical Research Institute of Sweden were Birgit Brunklaus and Emma Rex. Examiner and supervisor at Chalmers University of Technology was Matty Janssen.

I would like to thank everyone who has helped me during the conduction of my master thesis. Thanks to my supervisors Birgit, Emma and Matty for your valuable advice and support throughout this project. Thanks to Martin Hedberg, Sune Wännström, Johanna Berlin and Katarina Lorentzon at SP Technical Research Institute of Sweden for your time and helpful input. I also want to thank Robert Lippens, Ragnar Davidsson and Graham Aid at Ragn-Sells for helping out with information and provision of data. Finally, thanks to my family and friends for your support and encouragement.

Göteborg June 2016

Erica Carlsson

Abbreviations

AD	Anaerobic digestion
AP	Acidification Potential
CHP	Combined heat and power plant
CH_4	Methane
CO_2	Carbon dioxide
CO_2-eq	Carbon dioxide equivalents
DC	Direct Crystallization
DDGS	Distiller's Grain with Solubles
d.m.	Dry Mass
E.Coli	Escherichia Coli
EP	Eutrophication Potential
FU	Functional Unit
FW	Food Waste
GAC	Granulated Activated Carbon
GHG	Greenhouse Gas Emissions
GWP	Global Warming Potential
H_2O	Water
HCl	Hydrochloric acid
HTP	Human Toxicity Potential
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
$MgCO_3$	Magnesium Carbonate
MJ-eq	Mega Joule equivalents
NaCl	Sodium chloride
NaOH	Sodium hydroxide
NOx-eq	Nitrogen oxide equivalents
NREU	Non-Renewable Energy Use
REU	Renewable Energy Use
SA	Succinic acid
SO_2-eq	Sulfur dioxide equivalents
w.m.	Wet Mass
1,4-DCB-eq	1,4- Dichlorobenzene equivalents

1 Introduction

There are large quantities of food waste in society. About one third of all food globally produced for human consumption is lost or wasted, which corresponds to around 1.3 billion tonnes each year (FAO, 2011). In year 2012, 1.2 million tonnes of food waste was generated in Sweden, corresponding to an average of 127 kg/person (Naturvårdsverket, 2014). This means that vast amounts of resources are used and greenhouse gas emissions are emitted in vain (FAO, 2011). Due to the growing world population and economy, the amount of food waste is increasing (Uçkun Kıran et al., 2015).

There are several ways to handle food waste. One common industrial use is to produce biogas from the food waste. Another option is to produce bio-based chemicals. The need to find alternatives to fossil fuels in combination with resource depletion and climate change concerns, and waste accumulation has generated attention to using food waste as a renewable feedstock (Pfaltzgraff et al., 2013).

There is currently ongoing research regarding production of valuable bio-based platform chemicals from food waste (Uçkun Kıran et al., 2015). Food waste is rich in nutrients an organics which makes it a potential resource for fermentative production of high value platform chemicals. One such high value platform chemical which can be produced from food waste is succinic acid (SA). SA is a building block or platform chemical which means it can be transformed into many other chemicals (U.S. Department of Energy, 2004). Bio-based succinic acid has received increasing attention due to its large potential market and the opportunity to replace many fossil-derived chemicals (Chimirri et al., 2010) (Uçkun Kıran et al., 2015).

One important aspect to consider is if using food waste to produce chemicals is an environmentally preferable option. This may be analyzed by conducting a life cycle assessment of the process. There are published environmental evaluations of succinic acid produced from corn, sugar cane and corn stover (European Comission, n.d.). To the author's knowledge, there are no commercial processes or published environmental evaluations of using food waste for production of succinic acid yet.

This master thesis has been carried out at SP Technical Research Institute of Sweden in conjunction with the FORMAS project "Introducing high value product formation into the bio-refinery". The goal of this research project is to suggest a possible function and structure of a profitable a bio-refinery process using a mixture of waste to produce both high and low value products such as bio-based platform chemicals or biomaterials (SP Process Development, n.d.). Besides LCA, SP is performing several types of systems analyses until 2018. This thesis can be seen as a pre-study for further environmental analyses within the research project.

1.1 Aim of Master Thesis

Food waste can be seen as both a problem and a resource. This thesis aims to look at food waste from both perspectives. The goal of the thesis is to answer the following research questions:

- From an environmental point of view, what is the best waste management option for processing one tonne of food waste: production of biogas or succinic acid?
- From an environmental point of view, what is the best feedstock option for producing one tonne of bio-based succinic acid: food waste or corn?

To evaluate the possible environmental impacts of producing succinic acid and biogas, respectively, from food waste, a Life Cycle Assessment (LCA) will be conducted. The results will further be compared to published LCA results for production of succinic acid from corn. The results of this thesis may indicate if it is relevant from an environmental point of view to develop a bio-refinery process for production of succinic acid from food waste.

2 Literature Review

2.1 The Bio-refinery Concept

A bio-refinery can be described as a refinery which can transform various kinds of biological feedstocks into a range of different products such as energy, chemicals and materials (Clark et al., 2006). The bio-refinery concept is thus a bio-based equivalent to a conventional petroleum refinery which produces several fuels and products from fossil petroleum (Cherubini and Ulgiati, 2010). To be able to transfer to a bio-based economy, in contrast to the current fossil-based economy, efficient and cost-effective bio-refineries need to be developed (Cok et al., 2014).

The European chemical industries state that industrial biotechnology is a key emerging technology area (Hatti-Kaul et al., 2007). The need for mitigating climate change is a driver for finding and developing new green technologies which can transform waste biomass into bio-based, fuels, chemicals and materials (Sheldon, 2014). Industrial biotechnology, also called white biotechnology, use microorganisms or enzymes to produce chemicals by using the cells' own metabolic pathways (Hatti-Kaul et al., 2007). By modifying metabolic pathways with genetic engineering, organisms can produce several types of platform chemicals.

Due to a growing demand energy, fuels, chemicals and materials in combination with limited resources, industries need to become more resource efficient and find new ways to utilize waste (Lin et al., 2013). By using food waste as a renewable feedstock in bio-refineries, energy and chemicals can be produced while at the same time managing food waste and reducing the dependence of fossil resources (Leung et al., 2012).

To make the chemical industries willing to invest in switching from fossil to renewable feedstocks, there must be an economic incentive (Bozell and Petersen, 2010). By using integrated bio-refineries to co-produce high-value bio-based chemicals, biofuels and energy, both productivity and profitability will be increased (U.S. Department of Energy, 2004). Integrated bio-refineries which co-produce fuels and chemicals could provide a high economic profit and thus attract investment (Bozell and Petersen, 2010).

2.2 Food Waste: Statistics and Valorisation

Food waste (FW) can be defined in a number of ways. According to FAO (2011) food waste can be seen as a food loss i.e. a decrease of food intended for human consumption in the retail or consumption phase. Food waste can also be divided into avoidable and unavoidable food waste. Naturvårdsverket (2014) defines avoidable food waste as food which could have been eaten if treated properly and eaten in time

and unavoidable food waste as waste which is difficult to reduce, e.g. peels and coffee grounds. Thus, avoidable food waste is or was edible while unavoidable food waste is and has never been edible (Papargyropoulou et al., 2014).

Around one third of all food produced for humans, corresponding to 1.3 billion tonnes annually, is lost or wasted in the global food supply chain (FAO, 2011). Food waste generation in Sweden in year 2012 was 1.2 million tonnes which corresponds to an average of 127 kg/person, year (Naturvårdsverket, 2014). These numbers do not include food waste from agriculture, fishing or unavoidable food waste from industries. Households give rise to the largest share of food waste, 771 ktonnes or 81 kg/person and year, out of which 35 % was avoidable waste. In medium and high income countries, most food waste is generated in the consumption phase due to consumer behavior and a wasteful life style (FAO, 2011). Reasons for this includes past expiration dates and inadequate purchase planning.

The food supply chain including production, distribution and consumption gives rise to large environmental impacts such as climate change, eutrophication, acidification, ecotoxicity and loss of biological diversity (Naturvårdsverket, 2014). Production of the annual amount of Swedish food waste corresponds to 2 million tonnes CO_2 which is around 3 % of the total amount of emitted greenhouse gases in Sweden. Reducing the amount of food waste is an important step towards a more sustainable and resource efficient society. This would lead to a reduced environmental impact from the food supply chain or the opportunity to feed more people without increasing the environmental impact.

There are several ways to handle and valorize food waste (Lin et al., 2013). Food waste can be composted, incinerated for energy recovery, landfilled, used as animal feed, used for biogas production through anaerobic digestion or fermented in a bio-refinery to produce chemicals. There are several legislations and directives for how waste should be handled, e.g. the waste hierarchy which aims to find the environmentally best way to handle waste (Papargyropoulou et al., 2014). An increasing number of policies also identify food waste as an important area within waste management.

The food recovery hierarchy, presented in Figure 1, states the preferred order of food waste management actions (US EPA, 2015). Producing chemicals from food waste, which is a type of industrial use, is ranked as the fourth most preferred option. There is no clear distinction of which type of industrial use is the best option between producing platform chemicals or fuel. If one industrial use has larger economic or environmental benefits this might motivate why one industrial use could be a preferred option over another one.

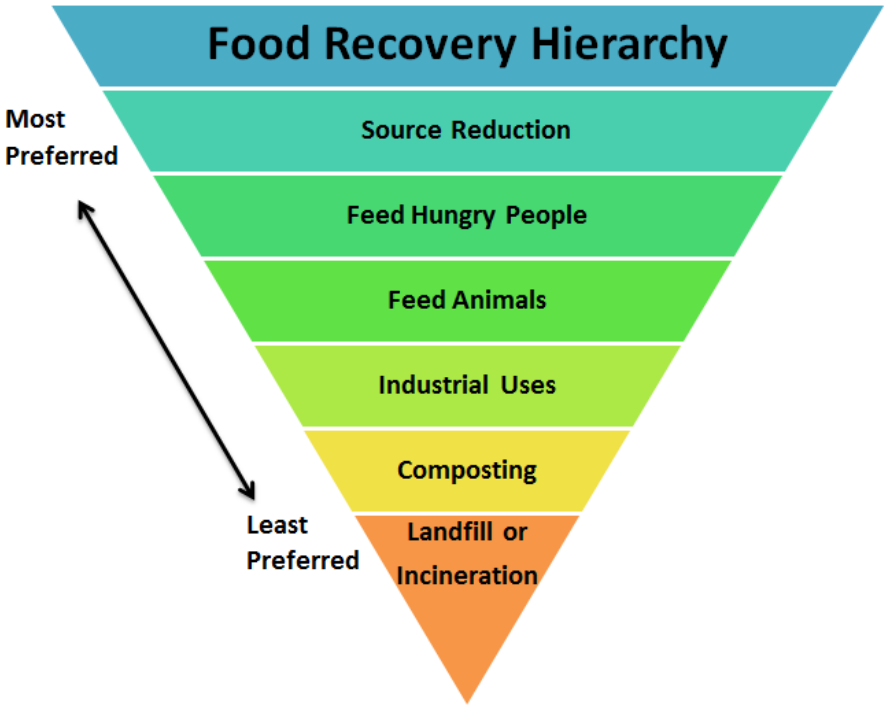

Figure 1: The food recovery hierarchy describing the prioritized order of food waste management, (Adapted from (US EPA, 2015)).

2.3 Bio-refinery Products from Food Waste

There are several products which can be produced through biological industrial processes and bio-refineries, e.g. biogas and platform chemicals.

2.3.1 Biogas

One food waste valorization option is to produce biogas through anaerobic digestion. Anaerobic digestion (AD) is a process where organic matter is degraded and converted into biogas by microorganisms in the absence of oxygen (Li et al., 2011). The final product biogas is a gas containing carbon dioxide and 60-70% methane, which can be upgraded to transportation fuel or used for electricity or heat production (Bondesson et al., 2013). A residue by-product containing high levels of nitrogen is also produced, which can be used as a biofertiliser (Li et al., 2011).

Biogas is currently one of the most produced biofuels in the world, along with bioethanol and biodiesel (Cherubini and Ulgiati, 2010). The energy demand in the transportation sector is increasing and the biofuel market is expected to grow and supply about 10-20 % of the transportation market in 2030. According to a study by Bernstad et al. (2013) the unavoidable food waste from Swedish households could generate around 400 GWh per year.

2.3.2 Platform Chemicals

Another possible food waste valorization option is to produce platform chemicals. A platform chemical is a molecule with several functional groups which can be transformed into a wide range of other molecules (U.S. Department of Energy, 2004). Platform chemicals are thus building blocks from which various kinds of secondary chemicals, intermediates and products can be produced (Uçkun Kıran et al., 2015). Producing platform chemicals usually gives a higher economic profit compared to production of transportation fuel, electricity or using the food waste as animal feed. Bio-based platform chemicals are interesting due to their ability to replace petrochemical building blocks (van Heerden and Nicol, 2013).

2.4 Succinic Acid as a Platform Chemical

One possible platform chemical which can be produced from food waste is Succinic Acid (SA). Succinic acid, shown in Figure 2, is a four-carbon dicarboxylic acid with the molecular formula $C_4H_6O_4$ (Cok et al., 2014). SA is naturally produced in almost all animal, plant and microbial cells as an end-product of the anaerobic metabolism (Song and Lee, 2006). Succinic acid can also be produced through fermentation of carbohydrates by the use of microorganisms (Zeikus et al., 1999).

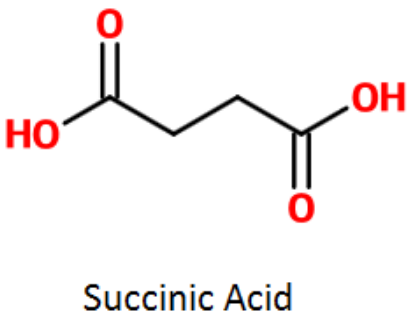

Figure 2: Chemical structure of Succinic Acid.

SA has been listed as one of the top twelve sugar-derived platform chemicals which have the highest potential for deriving bio-based chemicals in integrated bio-refineries (U.S. Department of Energy, 2004). SA is regarded as a highly promising building block chemical due to the many potential chemical derivatives, its economic value and potential market (Chimirri et al., 2010). Succinic acid could be used for producing a wide spectrum of products, such as surfactants, solvents, pigments, detergents, plasticizers, and in pharmaceuticals and food. SA also has a vast potential as a monomer for various types of bio-based polymers (Choi et al., 2015).

The feedstock for producing succinic acid can either be fossil-based or bio-based. Fermentative production of succinic acid has received increasing research attention as a bio-based alternative to the conventional petrochemical production (Leung et al., 2012). Fermentative production uses more gentle operating conditions and thereby requires less energy. It can also utilize more than just one type of feedstock.

Common feedstocks for fermentative SA production are various kinds of sugars, starch and molasses, so called 1st generation feedstocks (Jansen and van Gulik, 2014). Using first generation feedstocks have been subject to criticism since they directly or indirectly compete with food production which is not sustainable in the long term (Sheldon, 2014). Currently, much research instead focuses on the use of 2nd generation feedstocks, i.e. non-edible crops and waste biomass from e.g. forestry and agriculture (Jansen and van Gulik, 2014).

The current annual production of SA is 30 – 35 ktonnes, corresponding to a market value of 225 million US dollars, and the production rate is increasing (Uçkun Kıran et al., 2015). The annual potential market for SA including its derivatives has been estimated to 245 ktonnes (Bozell and Petersen, 2010). The potential market size and available low-cost feedstocks indicates that fermentative bio-based SA will replace fossil based SA in the future (Song and Lee, 2006).

One main benefit of fermentative SA production is that SA can be produced anaerobically and consume one mole CO_2 per mole of SA (Uçkun Kıran et al., 2015). Replacing petrochemicals with intermediates derived from bio-based SA can thus reduce the environmental impact. Production of ethanol, which is another widely produced platform chemical, produces CO_2 as a by-product. (Zeikus et al., 1999). Succinic acid and ethanol fermentation can thus be combined to reduce CO_2 emissions.

2.4.1 Commercial Production of Bio-based Succinic Acid

Several companies have started to develop or already perform production of bio-based succinic acid from renewable feedstocks in an industrial scale (Cok et al., 2014). Examples of such companies are Reverdia, Myriant, BioAmber and BASF-Purac. The starting capacity of the industrial production sites ranges from 10 up to 77 ktonnes per year.

BioAmber have a pilot plant in Pomacle, France which uses recombinant E.Coli to produce sodium or ammonium succinate (López-Garzón and Straathof, 2014). Clarification, electro dialysis, ion exchange and nanofiltration gives 99.5 % pure crystals and a recovery rate of 96%. In 2015 a joint venture between BioAmber and Mitsui completed the construction of the largest succinic acid production plant in the world, located in Sarnia, Canada (BioAmber, n.d.-a). This facility can produce 30 ktonnes SA per year. BioAmber is also planning to construct another plant which will have a capacity of 70 ktonnes of SA. This new plant will be located in North America and is expected to be completed in late 2018.

Reverdia, a joint venture between BASF and Roquette Freres, have built a factory producing 10 ktonnes SA per year in Cassano, Italy (López-Garzón and Straathof, 2014). This facility produces bio-based succinic acid through low pH fermentation using a recombinant strain of *Saccharomyses cerevisiae*. The succinic acid is sold under the name Biosuccinium (Reverdia, 2015). The process "low pH yeast fermentation process with downstream processing by direct crystallization", developed by Reverdia is also called the Reverdia DC process (Cok et al., 2014). This process was used to model Option 3: Corn to SA. A more detailed description of the Reverdia DC process can be found in Chapter 4.

None of the production companies currently use mixed food waste as a feedstock. The company BioAmber however has the ambition to go from the currently used 1st generation feedstock such as corn glucose to 2nd generation feesdtocks in the form of non-food biomass (BioAmber, n.d.-b). The long-term goal is to use industrial waste.

2.5 Succinic Acid from Food Waste

Food waste is composed of 30-60 % starch, 10-40 % lipids and 5-10 % proteins (Lin et al., 2014). Food waste is rich in nutrients an organics which makes it a valuable possible resource for fermentative production of high value platform chemicals (Uçkun Kıran et al., 2015). Food waste can be used as the only raw material for value-added bio-products without adding supplementary nutrients. However some types of food waste may not contain a sufficient amount of nutrients and additional nutrients might need to be added (Lin et al., 2013).

There are several papers evaluating the possibility of producing SA from different types of food waste (Uçkun Kıran et al., 2015). In general, SA fermentation has been performed for one type of pure food wastes and there is a lack of studies regarding SA fermentation from mixed food waste. Leung et al. (2012) developed a bio-refinery concept which uses waste bread as a feedstock for fermentative production of succinic acid. Zhang et al. (2013) analyzed the possibility of using waste cakes and pastries from Starbucks Hong Kong for production of succinic acid. Lam et al (2014) evaluated the economic feasibility of a simulated pilot plant with fermentative SA production from bakery waste, based on the results presented by Leung et al (2012). Their conclusion was that such a process is economically feasible.

Sun et al. (2014) published a study of lab-scale fermentative production of SA from mixed food waste using the microorganisms *A. succinogenes* and recombinant *Escherichia coli* (*E.Coli)*. The food waste was collected from canteens in Hong Kong and consisted of noodles, rice, meat and vegetables. Sun et al. (2014) state that metabolically engineered *E.Coli* was beneficial to use for mixed food waste since it was fast growing and had no formation of by-products, which resulted in an easier recovery process. Sun et al. (2014) concluded that food waste can be used to produce

a nutrient-complete medium and that mixed food waste has a great potential to be used as a renewable feedstock for bio-based chemicals and materials.

Sun et al. (2014) also presented a comparison of achieved yields of SA from different food waste substrates including e.g. potatoes, orange peel, rapeseed meal, bread, cake, pastry and mixed food waste. The yields ranged between 0.115-1.16 g SA / g glucose, respectively 0.087 – 0.55 g SA / g food waste, wet mass (w.m.). The SA yield of 0.55 g SA/g bread, reported by Leung et al. (2012) is the highest reported yield for any food waste substrate. Mixed food waste in comparison is 0.224 g SA/g food waste, thus around half the yield of bread.

2.6 Previous Environmental Evaluations

Two important drivers for developing bio-refineries are energy security and mitigation of climate change (Cherubini and Ulgiati, 2010). Thereby, impact categories evaluating greenhouse gas emissions (GHG) or energy use are often used in LCA studies. In this section, published environmental impact results of biogas from food waste, succinic acid and emerging technologies are presented.

2.6.1 Environmental Evaluations of Biogas from Food Waste

Only a few published LCAs evaluate the environmental impact of biogas production from food waste through anaerobic digestion (Jin et al., 2015). Bernstad et al. (2012) compared the global warming potential results from 25 different LCA's analyzing different types of food waste treatments including anaerobic digestion. The reported GWP results were from -400 to 400 kg CO_2 per tonne treated food waste and differed much between the studies. These differences were not because of actual differences in environmental impact, but due to how the studies were conducted in terms of methodological choices, system boundary choices and input data. When comparing results from LCA studies, it is therefore important that the system boundaries are identical or at least comparable.

Jin et al. (2015) performed an LCA of an existing biogas system based on food waste located in China. The GWP result was 97 kg CO_2-eq per tonne treated food waste when including pre-treatment, biogas recycling, anaerobic digestion and digestate treatment. Jin et al. (2015) concluded that the total environmental impact was largely affected by properties of the food waste, such as moisture, grease and impurities content as well as biogas production ratio. Jin et al. (2015) also concluded that an increase in biogas production yield can reduce the environmental impact of the biogas system.

In an LCA of biogas production through anaerobic digestion from industry food residues by Poelsch et al. (2012) the climate change impact was -52 kg CO_2-eq/ per tonne processed feedstock. The negative impact is due to the modelling of biogas

substituting fossil fuels and the digestate substituting chemical fertilisers, thus using a consequential LCA approach. In another LCA of biogas production from food waste through anaerobic digestion using data from China, Xu et al. (2015) concluded that in their study, the electricity consumption of the anaerobic digestion resulted in a high environmental impact.

Several studies showed that the transportation of waste is seldom of importance to overall environmental impacts (Bernstad and la Cour Jansen, 2012). Neither is the production, use or disposal of paper bags for collecting the food waste. Plastic collection bags can however contribute significantly to the GWP. The pre-treatment process can have a large contribution to the overall environmental impact. The eco-profile of the used energy affects the result attributed to the energy input. Other assumptions such as characteristics of the food waste including carbon and nutrient content, storage emissions, emissions from pre-treatment and use of biofertiliser and the environmental impact of any substituted goods had a considerable effect on the LCA results.

2.6.2 Environmental Evaluations of Succinic Acid

According to the European Comission (n.d.), published LCA results for bio-based SA are available for the impact categories climate change, land use, primary energy and non-renewable energy. The LCA's are evaluated for the feedstocks corn, sugar cane or corn stover and are either of cradle-to-gate or cradle-to-grave type. Today, bio-based SA can save 0 to 55% of the NREU from cradle-to-factory gate compared to fossil SA (BREW project, 2006). In the future, bio-based SA could save between 30 to 60 %. According to the BREW project (2006), the use phase for bio-based chemicals in most cases does not give rise to emissions. The impact of the use phase can therefore be considered as negligible.

There are two published cradle-to-gate LCA's of the Reverdia DC process producing biosuccinum ((Cok et al. (2014):(Smidt et al. (2015)). Both evaluate the non-renewable energy use (NREU) and greenhouse gas emissions (GHG) due to the production of 1 kg of bio-based SA (≥99.5 wt-% pure) from corn. A contribution analysis describing how much different sub-processes contribute the total GHG impact is also presented in both studies.

Based on studies performed by Cok et al. (2014) and the BREW project (2006) the climate change impact for SA production from corn in a cradle-to-gate LCA is between 0,3-3,1 kg CO_2-eq per kg bio-based SA. The Non-renewable energy use for the same LCA is between 28,0 – 66.5 MJ per kg bio-based SA. In the study by Cok et al. (2014), 1 kg of bio-based SA corresponded to 32.7 MJ using the impact assessment method cumulative energy demand. The GWP results is 0.88 kg CO_2-eq per kg bio-based SA if the carbon uptake from corn cultivation is included and using the impact

method IPCC 2007 GWP 100a. Of the total CO_2 impact, 49 % was attributed to dextrose from corn, out of which 76 % was attributed to corn production. The main hot-spots of the Reverdia DC process were utilities, direct field emissions and the drying of corn in the dextrose production.

Smidt et al. (2015) published a study which builds on the previous article by Cok et al. (2014). In the study by Smidt et al. (2015), the greenhouse gas emissions (GHG) of 1 kg of bio-based SA (≥99.5 wt-% pure) is 2,34 kg CO_2-eq/ kg bio-based SA if excluding the carbon uptake in the corn cultivation. The total environmental impact is also presented in a number of different impact categories as weighted results using the ReCiPe method. When using the Eco impact assessment Single score, the highest impacts were found to be Climate change human health, Particulate matter formation, Agricultural land occupation and Fossil depletion.

An LCA was also performed for bio-based succinic acid produced by the company BioAmber (BioAmber, n.d.-b). BioAmber claim that their bio-based succinic acid reduces the greenhouse gas emissions by 100 % and the energy consumption by 60 % compared to petroleum-based succinic acid. To the author's knowledge, there is currently no published study evaluating the environmental impacts of fermentative production of succinic acid from food waste. This thesis can thus provide new insight in this area.

3 LCA Methodology

3.1 The LCA Procedure

Life Cycle Assessment (LCA) is a method for assessing the environmental impacts of products and services through its whole life cycle (Baumann and Tillman, 2004). The LCA procedure assess the potential environmental impacts of the associated resources and emissions of a product or service through its whole life cycle including raw material extraction, manufacturing, use and waste disposal. LCA is a standardized procedure described in the international standard series ISO 14040-14043. The results of an LCA are commonly used as a basis for decision-making or to find possibilities for improvement.

The LCA procedure is an iterative process which includes goal and scope definition, inventory analysis, impact assessment and interpretation as described in Figure 3. The goal and scope describes what product should be studied and for which reason (Baumann and Tillman, 2004). The inventory analysis (LCI) is about building the model as formulated in the goal and scope and includes construction of a flow model, data collection and calculations of resources and emissions in relation to the functional unit. The impact assessment (LCIA) involves evaluating the environmental impacts based on the results from the data collection in the inventory analysis. This is done through classification, i.e. sorting resources and emissions after the type of environmental impact they give rise to and classification, i.e. calculating the relative contribution for each resource and emission. Using different weighting procedures are optional and can be used to aggregate the results to one single index. The final step of an LCA is interpretation of the results.

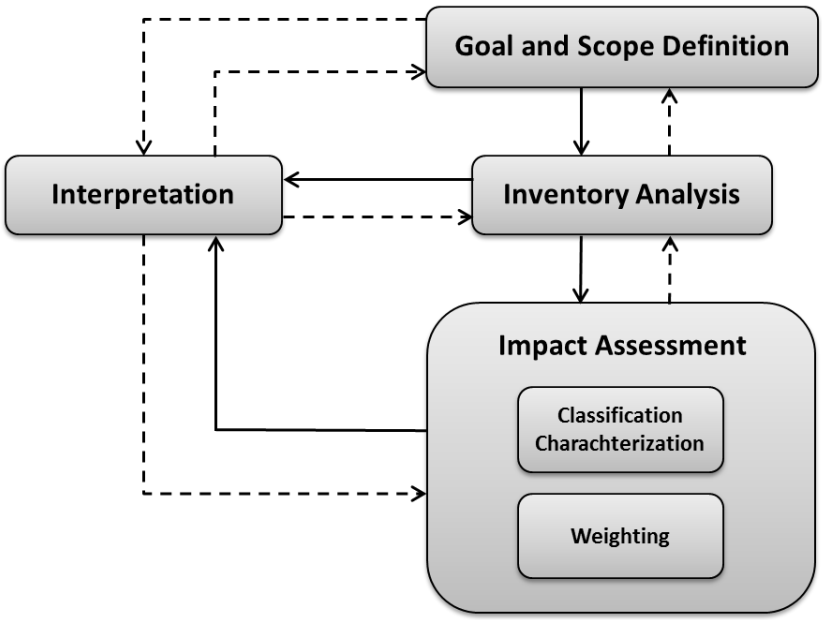

Figure 3: The LCA procedure. Arrows show the procedural order while the dashed arrows show iteration possibilities. (Adapted from (Baumann and Tillman, 2004))

3.2 Goal and Scope Definition

The goal and scope definition is the first step when conducting an LCA (Baumann and Tillman, 2004). The goal definition involves defining the purpose of the study and who the result intends to be communicated to. The scope definition describes the modelling aspects such as the context of the study. It also describes how the modelling will be performed including e.g. the functional unit, system boundaries, level of detail, data requirements and what environmental impacts will be considered.

3.2.1 Goal Definition

This master thesis will model and evaluate the environmental impacts of a bio-refinery process producing succinic acid from food waste using life cycle assessment (LCA). The results will be used as input to further research in the research project "Introducing high value product formation into the bio-refinery". The results of this thesis may show if it is relevant from an environmental point of view to develop a processes for production of succinic acids from food waste, and if so, what main areas to focus on to reduce the environmental impact of such a process.

Food waste can be seen as both a problem and a resource. This thesis aims to look at food waste from both perspectives. The goal of this master thesis is to answer the following research questions:

- From an environmental point of view, what is the best waste management option for processing one tonne of food waste: production of biogas or succinic acid?
- From an environmental point of view, what is the best feedstock option for producing one tonne of bio-based succinic acid: food waste or corn?

This will be fulfilled by evaluating and comparing the environmental impacts of the following three options for food waste management and/or production of bio-based succinic acid:

- Option 1: Food waste to biogas
- Option 2: Food waste to succinic acid
- Option 3: Corn to succinic acid

The aim of the thesis is further to:

- Identify the steps in the life cycle which gives the largest contribution to the total environmental impact through a dominance analysis.
- Perform a sensitivity analysis to evaluate changes in the most critical data.

LCAs are performed for Option 1: FW to biogas and Option 2: FW to SA using the software openLCA. The modelling of Option 1: FW to biogas uses data from Ragn-Sells existing biogas facility in Heljestorp, while Option 2: FW to SA is mainly based on literature data. The environmental impacts of Option 3: Corn to SA is evaluated based on available LCA results of the Reverdia Direct Crystallization (DC) process

published by Cok et al. (2014) and Smidt et al. (2015). The Reverdia DC process was chosen to represent Option 3 since published LCA results were available and since this process had the lowest environmental impact out of the three processes for fermentative SA production evaluated by Cok et al. (2014).

3.2.2 Scope Definition

3.2.2.1 Functional Unit

The functional unit (FU) is a unit which describes the function of the system (Baumann and Tillman, 2004). In this case, the function of the modelled bio-refinery serves two main functions: to take care of food waste and to produce valuable chemicals. To be able to answer the research questions, two functional units will be used: 1 tonne of processed food waste, dry mass (d.m.) and 1 tonne of produced succinic acid crystals, (≥99.5 wt-% pure), respectively. By using two functional units, Option 1: FW to biogas and Option 2: FW to SA can be compared in terms of managing food waste, while Option 2: FW to SA and Option 3: Corn to SA can be compared for producing succinic acid.

3.2.2.2 Impact Categories and Impact Assessment Choices

By using impact categories, the inputs and outputs in the form of resources and emissions can be translated to what type of environmental impact they might give rise to (Baumann and Tillman, 2004). In this study, six different impact categories will be used. The impact categories Global Warming Potential (GWP), Acidification Potential (AP), Eutrophication Potential (EP) and Human Toxicity Potential (HTP) will be assessed using the impact assessment method CML 2001. Non-Renewable Energy Use (NREU) and Renewable Energy Use (REU) from the impact assessment method Cumulative Energy Demand will also be used. These categories were chosen since they are commonly used in LCA studies and are environmental concerns. GWP and NREU were especially important to be able to compare the LCA results with published LCA studies of Option 3: Corn to SA published by Cok et al. (2014) and Smidt et al. (2015).

The reason for using CML 2001 was that it gives results in many different impact categories. Cummulative Energy Demand was chosen because this method was used in the LCA's of the Reverdia DC process by Cok et al. (2014) and Smidt et al. (2015). These studies also used the method IPCC 2007 100a to evaluate Greenhouse Gas Emissions (GHG), which gives similar results as CML 2001, and the GWP results can thereby be compared. Weighting will not be applied. The results of the different impact categories will be presented as kg equivalents (kg-eq) for each impact category in the form of graphs and tables.

3.2.2.3 System Boundaries and Type of LCA

The system boundaries determines what parts of the system under study are included in the LCA (Baumann and Tillman, 2004). The system boundaries of this LCA include all upstream processes for extraction and production of used energy and resources and the production of biogas or SA until it leaves the factory gate. The study is thereby a cradle-to-gate LCA, and will be attributional (i.e. accounting type). The reason is that it would be difficult to do a consequential or cradle-to-grave type LCA since SA can be used in a wide range of applications and it is unknown what the succinic acid is intended to replace. Moreover, the BREW project (2006) claims that the impact of the use phase for bio-based chemicals can be considered as negligible. According to Heimersson et al. (2014), a cradle-to-gate LCA is sufficient for systems which produce the same product and function. The end of life process for bio-based SA could thus be considered the same regardless if it is produced from food waste or corn and the waste management is therefore reasonable to exclude.

The geographical boundaries of the foreground system of Option 1: FW to biogas will be focused on Sweden since it is modelled as the Heljestorp biogas facility located in Sweden. The foreground system of Option 2: FW to SA will also be modelled as geographically located in Sweden. The background systems for the biogas facility and the food waste bio-refinery will use data for Swedish electricity mix. The rest of the background systems will use average European data, mainly from Ecoinvent 3.1. Option 3: Corn to SA is located in Italy and use corn cultivated in Europe. The time horizon will focus on the present and near future, using as recent data as possible. Mass allocation was used for processes producing multiple products follow the recommendations of the ISO standard.

3.2.2.4 Flowcharts of the analysed systems

Simplified flowcharts giving a general overview of the three options is presented in Figure 4. Further information about the three modelled options, additional flowcharts and an extensive technical description of each option can be found in Appendix A. Detailed flowcharts of the three options can be found in Appendix B.

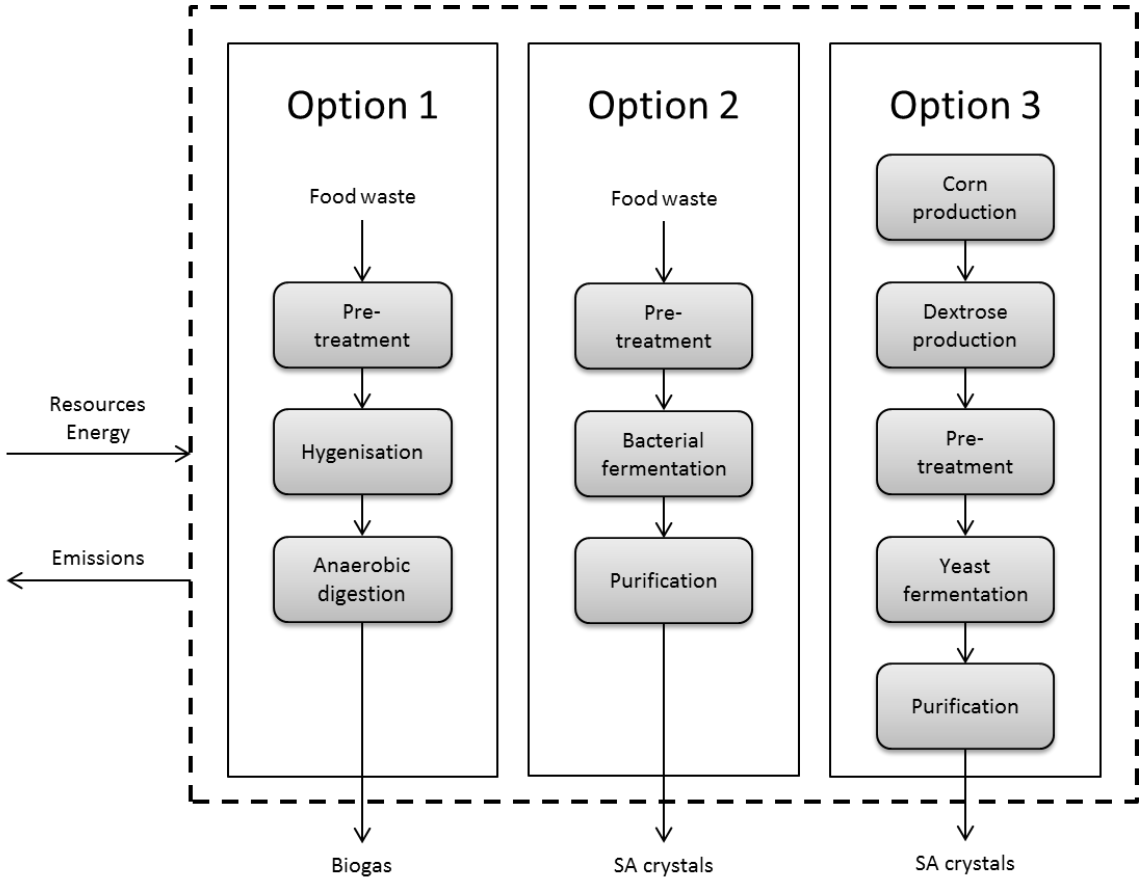

Figure 4: Simplified flowcharts of Option 1: FW to biogas, Option 2: FW to SA and Option 3:Corn to SA.

Option 1: Food waste to biogas

The modelling of Option 1: FW to biogas is based on Rang-Sells sorting and anaerobic digestion facility located in Heljestorp. Mixed waste is transported to the biogas facility where it enters the pre-treatment process (Ragn-Sells, n.d.) (Personal communication Ragn-Sells, 2016). The mixed waste is sorted, unwanted materials are removed and a combustible waste fraction is separated from the organic waste. The organic fraction is then grinded, hygenised with steam in the hygenisation process and mixed with water to a slurry. In the anaerobic digestion, a biogas constituting of 60 % CH_4 and 40 % CO_2 is produced from the slurry as well as a solid and liquid biofertiliser. A share of the produced biogas is used internally in a boiler to produce steam and hot water, while another share is combusted in a torch. The biogas is sold and upgraded to vehicle gas while the biofertiliser is sold to farmers. A solid residue is also obtained which is used to cover the nearby landfill. A more detailed description can be found in Appendix A.

Option 2: Food waste to Succinic Acid

The LCA modelling of Option 2: FW to SA is mainly based on literature data of a simulated pilot plant using waste bread for fermentative production of SA, published by Lam et al (2014). In the pre-treatment, food waste is grinded and mixed with water to a slurry (Lam et al., 2014). The slurry is then hydrolysed with enzymes to release

nutrients and subsequently centrifuged to remove undigested solids from the hydrolysate. E.Coli bacteria is cultivated and added to a bio-reactor together with the food waste hydrolysate to conduct the bacterial fermentation. CO_2, Sodium hydroxide (NaOH) and magnesuim carbonate ($MgCO_3$) are added to the fermentation reactor.

In some cases, food waste does not contain enough nutrients (Lin et al., 2013). Therefore an input of Dried Distiller's Grain with Solubles (DDGS) is modelled to supply additional nitrogen. To obtain SA crystals, the SA produced in the bacterial fermentation must be purified from the fermentation broth (Lam et al., 2014). The broth containing SA is centrifuged to remove biomass which can be sold as fish feed. Impurities are removed in a granulated activated carbon (GAC) column. By-product organic acids are removed in an ion exchange column by adding hydrochloric acid (HCl) and sodium hydroxide (NaCl) brine solution. Water is evaporated in a flash which makes the solution supersaturated and SA crystals form. Finally the SA crystals are dried in a tray drier. A more detailed description can be found in Appendix A. Additional theory of the SA fermentation process can be found in Appendix C.

Option 3: Corn to Succinic Acid

Option 3: Corn to SA is based on the description of the Reverdia DC process found in the LCA studies published by Cok et al. (2014) and Smidt et al. (2015). An LCA of Option 3 was not performed in this study due to lack of available data. The Reverdia DC process uses dextrose from corn to produce SA in a low pH yeast fermentation process. At a low pH the SA is in its' acid form rather than the succinate salt form which makes the recovery process easier. (Jansen and van Gulik, 2014). After the yeast fermentation, SA crystals are obtained in an upgrading process. A more detailed description can be found in Appendix A.

3.2.2.5 Data Quality Requirements

A literature study was performed to gather data and information for the modelling. The most recent data available was used. The main sources for the data collection in this study are:

- Direct contact with the waste management company Ragn-Sells.
- Literature data of a simulated pilot plant using bread for fermentative production of SA, published by Lam et al. (2014).
- Literature data of lab-scale fermentative SA production from mixed food waste, published by Sun et al. (2014).
- LCA results of the Reverdia DC process located in Italy which produces SA from corn, published by Cok et al. (2014) and Smidt et al. (2015).
- Average data from the database Ecoinvent 3.1, attributional version.

Data for Option 1: FW to biogas was based on site-specific data from Ragn-Sells sorting and anaerobic digestion facility, located in Heljestorp. The modelling of Option 2: FW to SA was based on combined data from a simulated pilot plant model

using bread and lab-scale yields of SA production from mixed food waste. Data for producing Dried Distiller's Grain with Solubles (DDGS), used to provide additional nitrogen in the bio-refinery was obtained from site-specific data from the Agroetanol process by Lantmännen. Due to lack of available data, Option 3: Corn to SA was not modelled in this thesis. Instead, LCA results of the Reverdia DC process published by Cok et al. (2014) and Smidt et al. (2015) were used as a basis for comparison to the LCA results of Option 2: FW to SA obtained in this study. The Ecoinvent database version 3.1 was used to obtain average data for the background processes. European average values were used except for the electricity where Swedish electricity mix was used.

3.2.2.6 Assumptions and Limitations

Assumptions of this study:

- The food waste is assumed to have no environmental impact. The reason for this is that the purpose of producing food is to produce food for eating, not to produce food waste. The environmental impact of food production and consumption is therefore considered to be completely allocated to the food which is eaten.
- The environmental impact of corn cultivation and dextrose production from corn is included in Option 3: Corn to SA since the corn is grown for the specific purpose of producing succinic acid. Thereby the environmental impact of corn and dextrose should be included in the published LCA results to get a fair comparison of the three options.
- The food waste is assumed to have a dry mass (d.m.) content of 30% (Bernstad Saraiva Schott et al., 2013) (Zhang et al., 2007). Thereby 1 tonne FW, wet mass (w.m.) corresponds to 0,3 kg food waste, dry mass (d.m.)
- The CO_2-emissions from combustion of biogas and landfill gas in Option 1: FW to biogas is assumed to be biogenic.
- The modelling of Option 2: FW to SA is mainly based on data for a pilot plant using waste bread for fermentative SA production, published by Lam et al. (2014). These data are assumed to be the same when using food waste instead of waste bread except for the yield of SA in the fermentation and upgrading processes and the host organism.
- E.Coli is assumed to be used as host organism in the food waste bio-refinery instead of A. Succinogenes. This is because the highest realized SA yield reported in literature has been achieved using metabolically engineered E.Coli (van Heerden and Nicol, 2013).
- The fermentation yield in the modelled food waste bio-refinery in Option 2: FW to SA is assumed to be 0.224 g SA / g food waste (w.m.) as stated by Sun et al. (2014).
- The yield of SA crystals in the upgrading process of Option 2: FW to SA is unknown, but assumed to be 60 %.

- An input of DDGS is assumed to be added in the bacterial fermentation in Option 2: FW to SA to supply extra nitrogen. When calculating the required amount of DDGS, the high value 1 kg E.Coli bacteria (d.m.) is assumed to be required per 1 kg SA to have a worst case scenario. For calculation details, see Appendix B.
- The cultivation of E.Coli bacteria is modelled as the process "Fermentation of whey" from Jungbluth et al. (2007) where the produced yeast paste is assumed to be propagated E.Coli.
- 0.1 g E.Coli bacteria (d.m.) is assumed to be required to produce 1 kg SA in the bacterial fermentation process in Option 2: FW to SA. This is however an uncertain value.
- The required amount of CO_2 in the fermentation of Option 2: FW to SA is assumed to follow the theoretical optimal reaction for transforming glucose to SA, presented by Heerden et al. (2013), see Appendix C, Section C.3. All supplied CO_2 is assumed to be converted to SA.
- The CO_2 used in the fermentation of Option 2: FW to SA is modelled as liquid CO_2, i.e. purchased CO_2 produced off-site.
- The steam and electricity used in Option 2: FW to SA is assumed to be divided in equal shares to the processes Pre-treatment, Bacretial fermentation and SA upgrading.
- The water used in Option 2: FW to SA is assumed to be divided between the processes Pre-treatment and SA upgrading. The same amount of water as food waste is assumed to be used in the Pre-treatment while the remaining share of water is assumed to be used in the SA upgrading process.

Limitations of this study:

- The environmental impact of the use phase and end of life is not included since the LCA is of cradle-to-gate type. Thereby no impacts from waste management, treatment or use of produced products, by-products or waste is included.
- The transport of waste to the biogas facility respectively the food waste bio-refinery is not included since it is seldom of importance to overall environmental impacts from a biogas facility (Bernstad and la Cour Jansen, 2012).
- No emissions from the biogas facility are accounted for other than CO_2-eq from combustion of internally used landfill gas and biogas due to lack of data.
- This study does not consider any economic or social impacts stemming from the three options.
- Impacts of capital goods are not included.
- Maintenance or resources used for maintenance of the biogas facility respectively the food waste bio-refinery is not included.

3.2.2.7 Sensitivity Analysis

A sensitivity analysis, i.e. changing the values of input parameters to see how it affects the results, was performed to identify the most critical parameters for the total result (Baumann and Tillman, 2004). Since Option 1: FW to biogas and Option 3: Corn to SA are based on site-specific data from existing processes while Option 2: FW to SA includes more uncertain data, it was most interesting to change parameter values for Option 2: FW to SA.

The base cases, i.e. the original model setups for the biogas facility and food waste bio-refinery are:

- **Base case Option 1: FW to biogas:** CO_2 emissions from combustion of landfill gas and biogas modelled as biogenic CO_2, no impact associated to food waste, mass allocation in the processes Pre-treatment and Anaerobic digestion.
- **Base case Option 2: FW to SA:** Include the DDGS input and use liquid CO_2 in the bacterial fermentation process, 60% yield in the SA upgrading process, normal enzyme and E.Coli amount, no impact associated to food waste, mass allocation in the processes DDGS production, E.Coli propagation and SA upgrading.

In the sensitivity analysis, different input parameters or assumptions are changed compared to the base case. These changes will be describes as different model changes. The model changes which will be evaluated in the sensitivity analysis for Option 1: FW to biogas are:

- **Biogenic CO_2 modelled as fossil CO_2:** Model biogenic CO_2 emissions in from the boiler and torch as fossil CO_2. This is done to make the impact of biogenic CO_2 visible.
- **Including CO_2 from food production:** Include the CO_2 impact of food production as an impact associated to the food waste. This is modelled as fossil CO_2 in the Pre-treatment process. This way a share of the impact from producing the food is allocated to the food waste.
- **No allocation in Anaerobic Digestion:** Do not use any allocation between the produced biogas and other co-produced by-products in the process Anaerobic digestion. This is done to see the total impact of the biogas facility.

The sensitivity analysis of Option 2: FW to SA will be evaluated for both functional units. The sensitivity analysis is divided into four sub-groups. The sub-groups of model changes which will be evaluated in the sensitivity analysis for Option 2: FW to SA are:

Modelling of CO_2

- **Including CO_2 from food production:** Include the CO_2 impact of food production as an impact associated to the food waste. This is modelled as fossil CO_2 in the Pre-treatment process. This way a share of the impact from producing the food is allocated to the food waste.
- **Biogenic CO_2 in fermentation:** Model the CO_2 input in the Bacterial fermentation process as biogenic CO_2 instead of liquid CO_2. This can be interpreted as combining SA production with another process and using biogenic CO_2 from an adjacent facility.
- **Fossil CO_2 uptake in fermentation**: Model the CO_2 input in the Bacterial fermentation process as fossil CO_2 instead of liquid CO_2. This can be interpreted as using fossil CO_2 from an adjacent facility, thus mitigating climate change.

Yield

- **30 % yield in upgrading process:** Decrease the upgrading yield of the process SA upgrading from 60 % to 30 %. This is done since the yield of the SA upgrading process in unknown and the yield has been reported to be of large importance for the environmental impact (Janssen et al., 2016).
- **90 % yield in upgrading process:** Increase the upgrading yield of the process SA upgrading from 60 % to 90 %. This is done since the yield of the SA upgrading process in unknown and the yield has been reported to be of large importance for the environmental impact (Janssen et al., 2016).

Process input changes

- **Tenfold enzyme increase**: Increase the enzyme use 10 times. This is done since enzyme production has previously been reported to have a large effect on the environmental impact (Janssen et al., 2016).
- **Excluding DDGS input:** Remove the DDGS input into the Bacterial fermentation process. Thereby the mixed food waste is assumed to contain enough nitrogen to supply the SA fermentation. This is done since Sun et al. (2014) concluded that food waste can be used to produce a nutrient-complete medium.
- **Thousandfold increase of E.Coli:** Assume 0.1 kg E.Coli bacteria (d.m.) is required to produce 1 kg SA in the bacterial fermentation process. This is done since the E.Coli requirement is a highly uncertain value. By using this large increase of 0.1 kg instead of the low value of 0.1 g in the original modeling, results similar to a best and worst case scenario is obtained.

Allocation

- **Economic allocation in SA upgrading:** Use economic allocation instead of mass allocation between the outputs SA crystals and biomass in the process SA upgrading. This is done since economic allocation could be more reasonable to use if the purpose of the food waste bio-refinery is to produce SA for economic profit.
- **No allocation in SA upgrading:** Do not use any allocation between the produced SA crystals and biomass in the process SA upgrading. This is done to see the total impact of the food waste bio-refinery.

4 Results

This chapter presents the results of the Life Cycle Inventory analysis, the Life Cycle Impact Assessment and the dominance analysis.

4.1 Life Cycle Inventory Results

Table 1 presents a selected choice of LCI results. It presents the main emissions contributing the most to the evaluated impact categories together with some inputs, and outputs for Option 1: FW to biogas and Option 2: FW to SA.

As can be seen in Table 1, there are quite different inputs and outputs when comparing the two options. Option 2: FW to SA has much higher emissions, non-renewable energy use, renewable energy use and water consumption compared to Option 1: FW to biogas, even if both options process 1 tonne of FW, d.m. Higher emissions and resource consumption leads to a larger environmental impact. The value for input of food waste and electricity consumption is the only thing which is similar between option 1: FW to biogas and Option 2: FW to SA.

One large difference between Option 1 and Option 2 is the NREU and REU impact. For Option 1, both the REU and NREU originate almost completely from the consumed electricity. For Option 2, the NREU impact mostly stems from the production of steam and NaCl while the REU impact mostly stems from production of NaCl, electricity and enzymes. Even though electricity is one of the most contributing factors to the REU impact for Option 2, many other flows contributes as well. Thus the electricity accounts for a smaller share of the total impact in Option 2 than in Option 1. Option 1 only have inputs of mixed food waste, landfill gas, fuel oil, electricity and water, while Option 2 have many more inputs and also uses slightly more electricity. Therefore, these additional inputs for Option 2 results a higher NREU and REU impact, especially due to the steam, NaCl brine and enzymes.

As can be seen in Table 1, the inputs, outputs and emissions are the same for both functional units for option Option 2: FW to SA, but the values are 2.23 times higher when producing 1 tonne of SA crystals compared to processing 1 tonne FW, d.m. This is due to how the LCA model was constructed in openLCA.

Table 1: The main LCI results, inputs and outputs for processing 1 tonne of food waste, d.m. respectively production of 1 tonne SA crystals.

LCI Results	Option 1: FW to biogas	Option 2: FW to SA	Option 2: FW to SA	Unit
	1 tonne processed FW, d.m.	1 tonne processed FW, d.m.	1 tonne produced SA crystals	
Inputs				
Mixed food waste	3.3	3.3	7.4	tonne
Landfill gas	364	n/a	n/a	kWh
Fuel oil	1.7	n/a	n/a	kWh
Electricity	258	274	611	kWh
Water	2.6	32	71	tonne
Enzymes	n/a	1.8	4.1	kg
DDGS	n/a	1.1	2.4	tonnes
Steam	n/a	11.0	24.5	tonnes
Propagated E.Coli	n/a	0.07	0.17	kg
$MgCO_3$	n/a	74.2	166	kg
NaOH	n/a	53.4	119	kg
CO_2	n/a	139	311	kg
HCl	n/a	21.4	47.7	kg
NaCl brine	n/a	4.1	9.1	tonnes
NREU	2.1	4066	9075	MJ-eq
REU	0.95	308	687	MJ-eq
Outputs				
Biogas (sold share)	1195	n/a	n/a	kWh
Liquid biofertiliser	3.24	n/a	n/a	tonnes
Solid biofertiliser	0.01	n/a	n/a	tonnes
Solid residue	0.13	n/a	n/a	tonnes
SA crystals	n/a	0.4	1.0	tonnes
Biomass	n/a	2.9	6.4	tonnes
Emissions to air				
Nitrogen oxides	0.057	597	1333	g
Ammonia	0.005	75	166	g
Sulfur dioxide	0.060	608	1357	g
Carbon dioxide, fossil	0.019	194	434	kg
Carbon dioxide, biogenic	0.024	9.7	21.6	kg
Methane, fossil	0.041	998	2228	g
Methane, biogenic	0.011	11	25	g
Chromium VI	0.003	20	46	mg
Arsenic	0.019	128	285	mg
PAH	0.007	113	253	mg

4.2 Life Cycle Impact Assessment Results

Each subchapter presents the LCIA result for Option 1: FW to biogas and Option 2: FW to SA in mass equivalents for one impact category. For Option 3: Corn to SA, results are only available for Global Warming Potential and Non-Renewable Energy Use. Each sub-chapter also includes the results of the dominance analysis.

4.2.1 Global Warming Potential

The results of the global warming potential are presented in Figure 5-7.

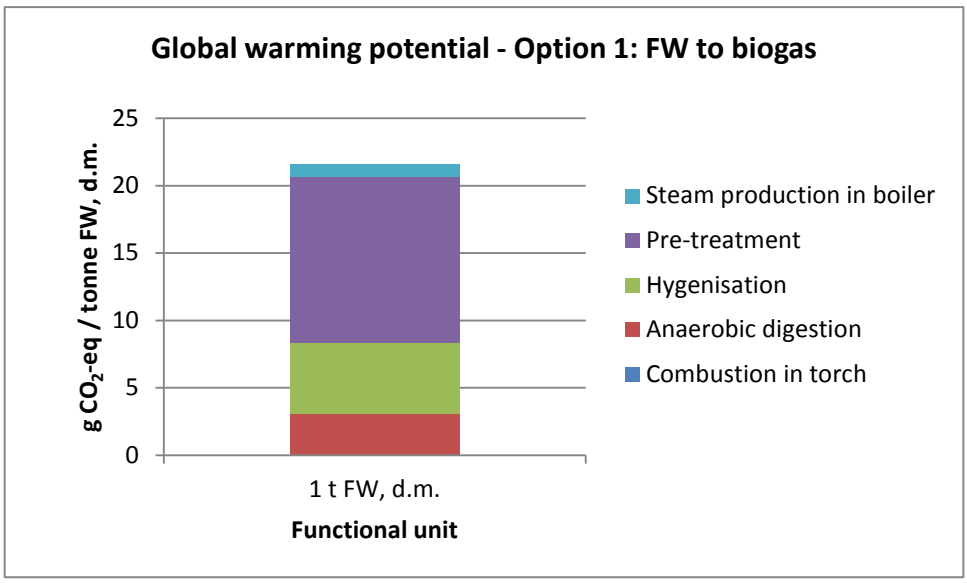

Figure 5: Global warming potential of Option 1: FW to biogas.

For Option 1: FW to biogas, processing 1 tonne of food waste d.m. gives rise to 22 g CO_2-eq. The largest contributing flow is electricity in all processes except Steam production in boiler. Since the Pre-treatment uses the most electricity, it contributes most to the impact, 57 %. The impact from Steam production in boiler is mainly due to combustion of fuel oil.

The fact that the electricity used in Option 1: FW to biogas accounts for most of the environmental impact is in line with the results from Xu et al. (2015) and Heimerson et al. (2014). Xu et al. (2015) conclude that the electricity consumption resulted in a high environmental impact for biogas production from food waste through anaerobic digestion. Heimerson et al. (2014) concludes that the GWP results strongly depend on used electricity mix. The impact of the electricity does however depend on what type of electricity mix is used.

The global warming impact is very low compared to other LCA studies of biogas production from food waste. Jin et al. (2015) reported a GWP result of 97 kg CO_2-eq / tonne treated food waste. Bernstad et al. (2012) presented summarized results of several LCA studies which reported -400 to 400 kg CO_2 per tonne food waste. The

results of these studies largely depended on the modelling choices. One reason for why the environmental impact of Option 1: FW to biogas is so low could be that few processes and emissions are modelled and the impact almost only stems from electricity. In addition, the mass allocation used in the Anaerobic digestion process leads to that only around 4 % of the impact from the biogas facility is allocated to the biogas. It might be more fair to use no allocation between the biogas and biofertiliser produced in the biogas facility. Moreover, this study is a cradle-to-gate LCA and does not use any system expansion of e.g. substituted products.

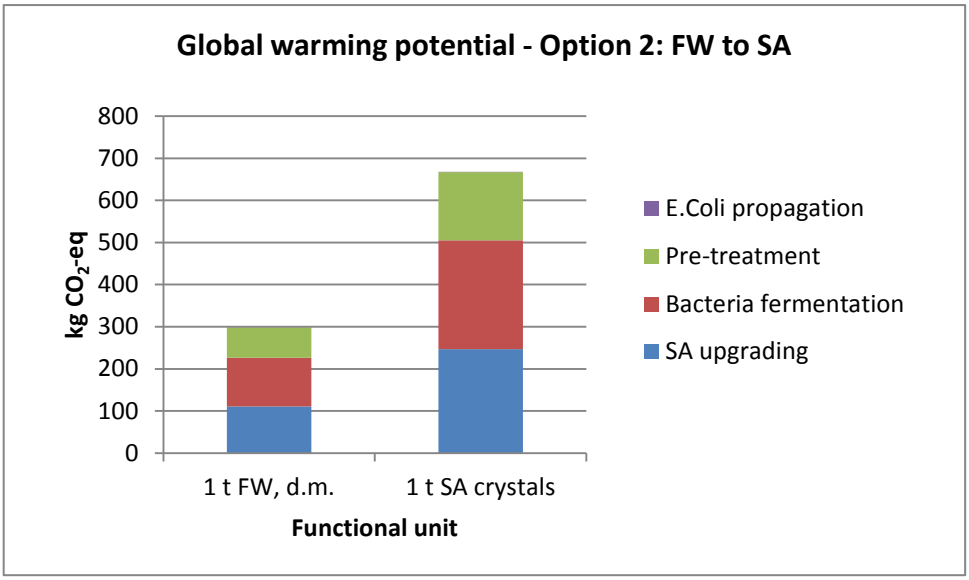

Figure 6: Global warming potential of Option 2: FW to SA.

For Option 2: FW to SA, the global warming potential is 300 kg CO_2-eq for processing 1 tonne of food waste d.m. and 670 kg CO_2-eq for production of 1 tonne SA crystals, see Figure 6. For both functional units, Bacterial fermentation is the process which has the largest contribution to the total impact, 39 %. This is mainly due to the DDGS input from the background process Ethanol production. The SA upgrading also contributes much, mainly due to the production of sodium chloride brine solution. The impact from the Pre-treatment is mostly due to steam production.

When producing 1 tonne of SA crystals in Option: 2 FW to SA, the impact is 2.23 times higher compared to processing 1 tonne of food waste, d.m.. This is valid for all evaluated impact categories. This is because only one model of Option 2: FW to SA was constructed in openLCA. The same model and same reference flow is used for both functional units. This also leads to that the result of the contribution analysis, i.e. the relation between how much each process contributes to the total impact, are identical for both functional units.

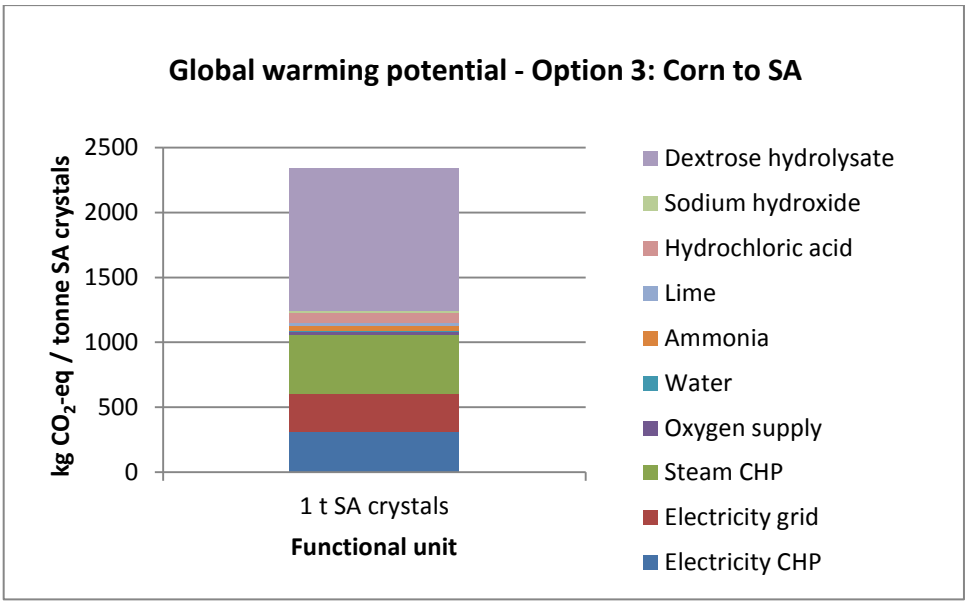

Figure 7: Global warming potential of Option 3: Corn to SA. The results are based on Smidt et al. (2015).

For Option 3: Corn to SA, the global warming potential is 2.34 tonnes CO_2-eq for production of 1 tonne SA crystals, see Figure 7. The results are excluding carbon uptake by corn when it is growing to get a more fair comparison since the carbon uptake from the food waste is not included in the model for Option 2: FW to SA. The climate change impact for SA production from corn in an cradle-to-gate LCA has been evaluated to between 0.3-3.1 kg CO_2-eq/ kg bio-based SA (European Comission, n.d.). The GWP results of this study for Option 2: FW to SA are in line with these results.

The dextrose hydrolysate (which is the production of dextrose from corn including corn cultivation) accounts for the largest share of the total GWP impact, 47 % (Cok et al., 2014) (Smidt et al., 2015). The major part of the other half of the impact comes from electricity and steam from the CHP and electricity from the grid. This is in line with what has been found in other studies. According to Hatti-Kaul et al., (2007) cultivation of corn can have a major contribution to the overall impact of a bio-based product since corn requires much energy, fertiliser and pesticides. The Wet milling of corn, i.e. the extraction of gluten, oil, starch and sugar from corn is energy-intensive.

In Figure 8 and 9, the total GWP impact is presented for the functional unit 1 tonne of processed food waste, d.m. and 1 tonne of produced SA crystals, respectively. These are the same values as presented in Figure 5-7 but rearranged to be able to compare the results for each functional unit.

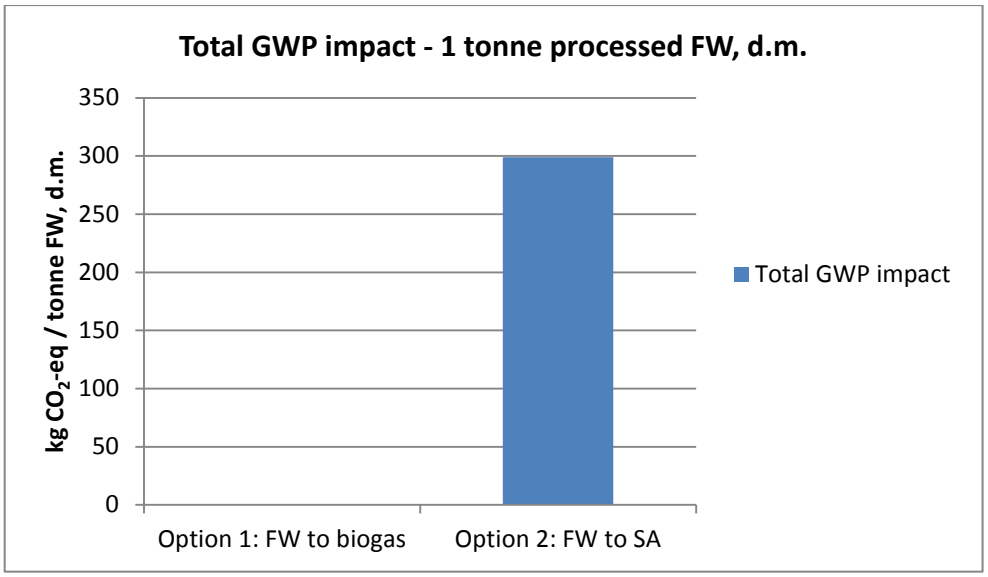

Figure 8: Total global warming potential for the functional unit 1 tonne of processed food waste, d.m.

As can be seen in Figure 8, the GWP impact for Option 1: FW to biogas is much lower than for Option 2: FW to SA, so small it is not even visible in the graph. Thus when treating food waste, production of biogas is an environmentally better option than production of succinic acid. The reason for the low impact from biogas production is that the impact for Option 1 almost completely is attributed to the electricity while Option 2 uses slightly more electricity and have many more inputs contributing to the impact than Option 1, especially steam, DDGS and NaCl brine solution.

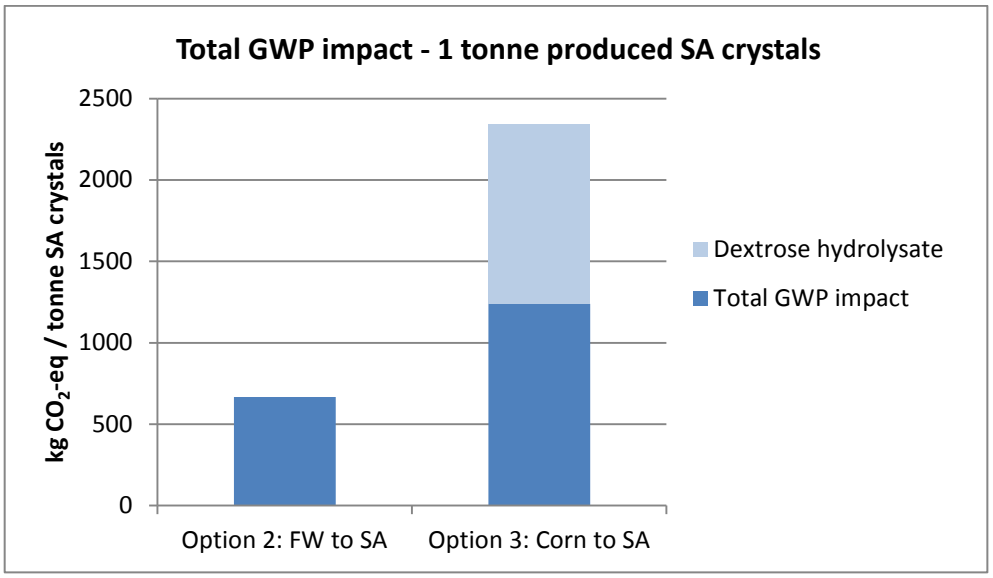

Figure 9: Total global warming potential for the functional unit 1 tonne of produced SA crystals. The results for Option 3: Corn to SA. are based on Smidt et al. (2015). For Option 3: Corn to SA, the share of the total GWP impact attributed to the dextrose hydrolysate is shown.

As can be seen in Figure 9, the GWP impact for Option 3: Corn to SA is higher than the GWP impact for Option 2: FW to SA. Note that the dextrose production should be included in the total impact for Option 3: Corn to SA in Figure 9. For production of SA, food waste is thereby an environmentally better option than corn. However, in this comparison, the food waste is assumed to have no environmental impact. If the impact of dextrose hydrolysate is excluded, food waste is still an environmentally better option than corn.

4.2.2 Acidification Potential

The results of the acidification potential are presented in Figure 10 and 11.

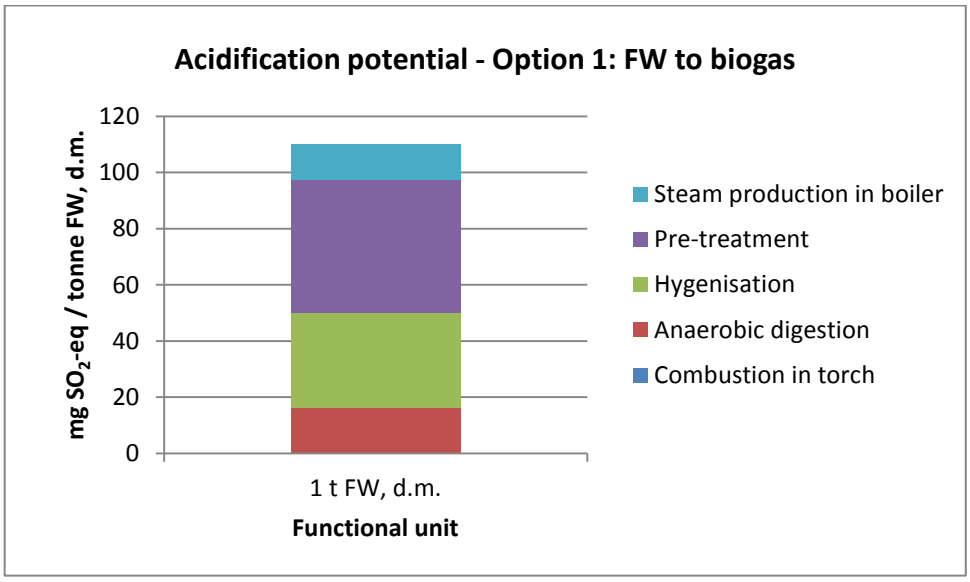

Figure 10: Acidification potential of Option 1: FW to biogas.

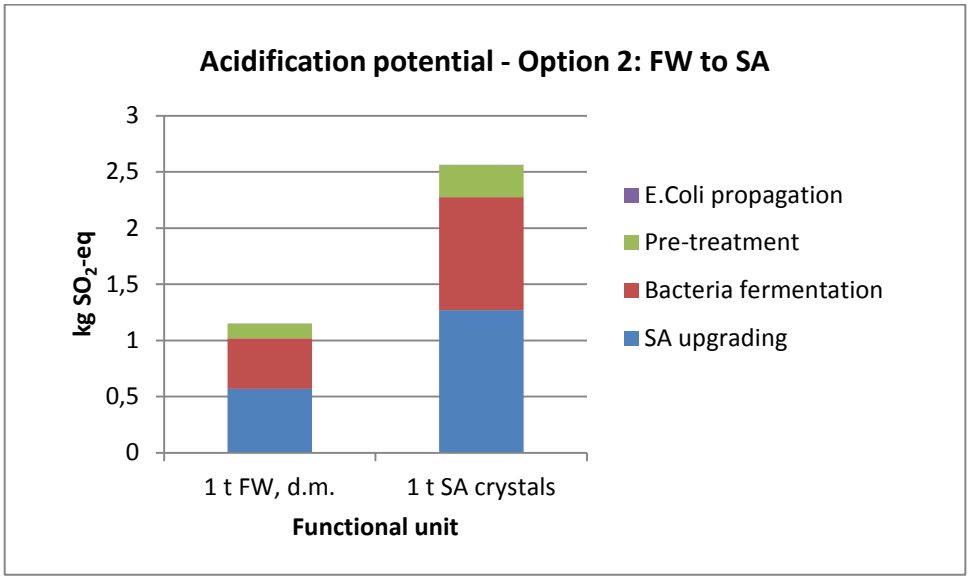

Figure 11: Acidification potential of Option 2: FW to SA.

For processing 1 tonne of food waste d.m., the acidification potential is 110 mg SO_2-eq for Option 1: FW to biogas and 1.15 kg SO_2-eq for Option 2: FW to SA. When treating food waste, production of biogas is thus an environmentally better option than production of succinic acid. All of the other impact categories show the same trend which means that biogas is a better food waste management option than to produce SA in all impact categories. For production of 1 tonne SA crystals, the acidification potential is 2.56 kg SO_2-eq for Option 2: FW to SA.

Just as for GWP, the largest contributing flow for Option 1: FW to biogas is electricity in all processes except Steam production in boiler. Since the Pre-treatment is the process which uses the most electricity, it contributes to the largest AP impact, 43 %. The impact from Steam production in boiler is mainly due to combustion of fuel oil. This trend of the impact stemming mostly from electricity and pre-treatment process contributing most to the impact due to highest electricity consumption is valid for all impact categories. For Option 2: FW to SA, SA upgrading account for 50 % of the total impact, mainly due to the production of sodium chloride brine solution. The 39% from Bacterial fermentation is largely because of the DDGS input and the impact from the Pre-treatment is mostly due to steam production.

4.2.3 Eutrophication Potential

The results of the eutrophication potential are presented in Figure 12 and 13.

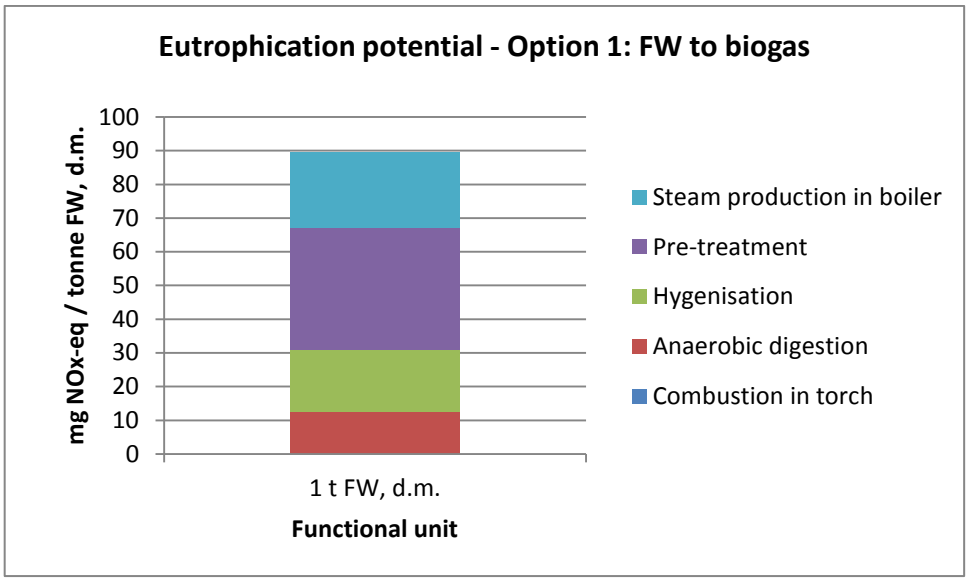

Figure 12: Eutrophication potential of Option 1: FW to biogas.

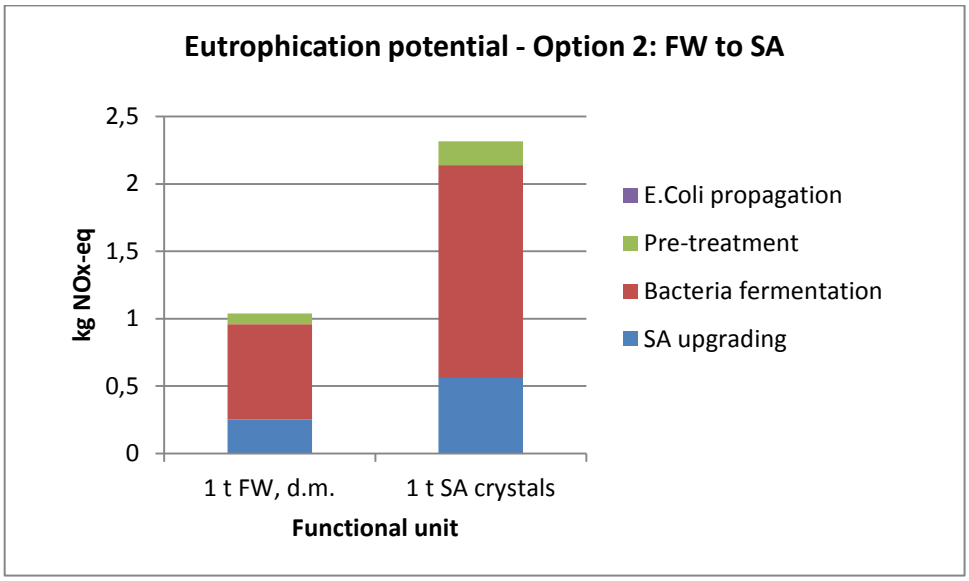

Figure 13: Eutrophication potential of Option 2: FW to SA.

For processing 1 tonne of food waste d.m., the eutrophication potential is 90 mg NOx-eq for Option 1: FW to biogas and 1.04 kg NOx-eq for Option 2: FW to SA. When treating food waste, production of biogas is an environmentally better option than production of succinic acid. For production of 1 tonne SA crystals, the eutrophication potential is 2.31 kg NOx-eq for Option 2: FW to SA.

Again, the largest contributing flow for Option 1: FW to biogas is electricity in all processes except Steam production in boiler. Since the Pre-treatment is the process which uses the most electricity, it contributes to the largest impact, 63 %. The impact from Steam production in boiler is again due to combustion of fuel oil. For Option 2: FW to SA, Bacterial fermentation accounts for 68 % of the total EP impact due to the DDGS input. The impact of SA upgrading is again mainly due to the production of sodium chloride brine solution and the impact from the Pre-treatment is mostly due to steam production.

4.2.4 Human Toxicity Potential

The results of the human toxicity potential are presented in Figure 14 and 15.

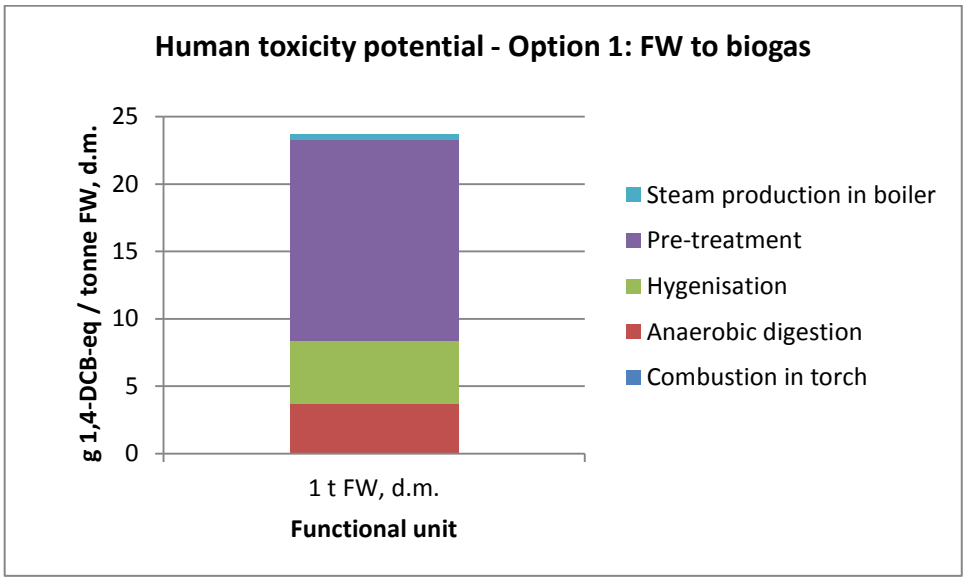

Figure 14: Human toxicity potential of Option 1: FW to biogas.

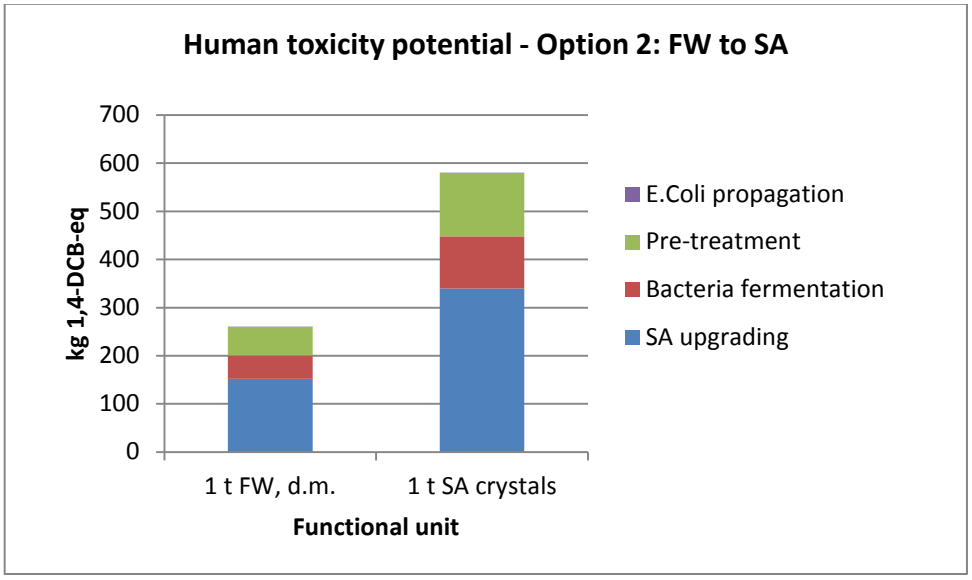

Figure 15: Human toxicity potential of Option 2: FW to SA.

For processing 1 tonne of food waste d.m., the human toxicity potential is 24 g 1,4-DCB-eq for Option 1: FW to biogas and 260 kg 1,4-DCB-eq for Option 2: FW to SA. When treating food waste, production of biogas is an environmentally better option than production of succinic acid. For production of 1 tonne SA crystals, the human toxicity potential is 580 kg 1,4-DCB-eq for Option 2: FW to SA.

The largest contributing flow for Option 1: FW to biogas is again electricity in all processes except Steam production in boiler. Since the Pre-treatment is the process which uses the most electricity, it contributes to the largest impact, 63 %. The impact from Steam production in boiler is due to combustion of fuel oil. For Option 2: FW to SA, SA upgrading accounts for 58 % of the total HTP impact. This is due to the production of sodium chloride brine solution. The 23 % from Pre-treatment is mainly due to enzymes from the background process Glucoamylase production. The 19% from bacterial fermentation is most due to liquid carbon dioxide and steam production.

4.2.5 Non-Renewable Energy Use

The results of the Non-renewable energy use are presented in Figure 16 and 17.

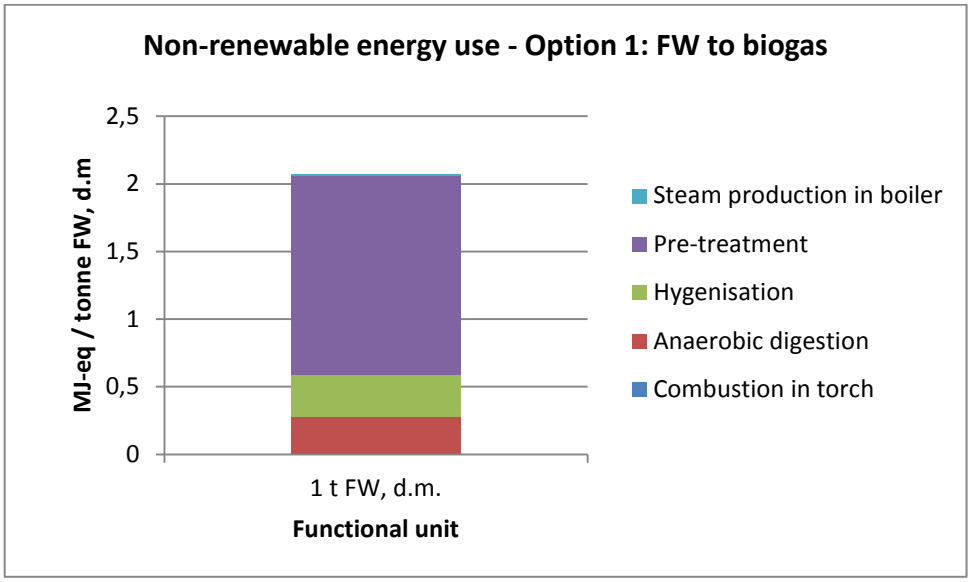

Figure 16: Non-renewable energy use of Option 1: FW to biogas.

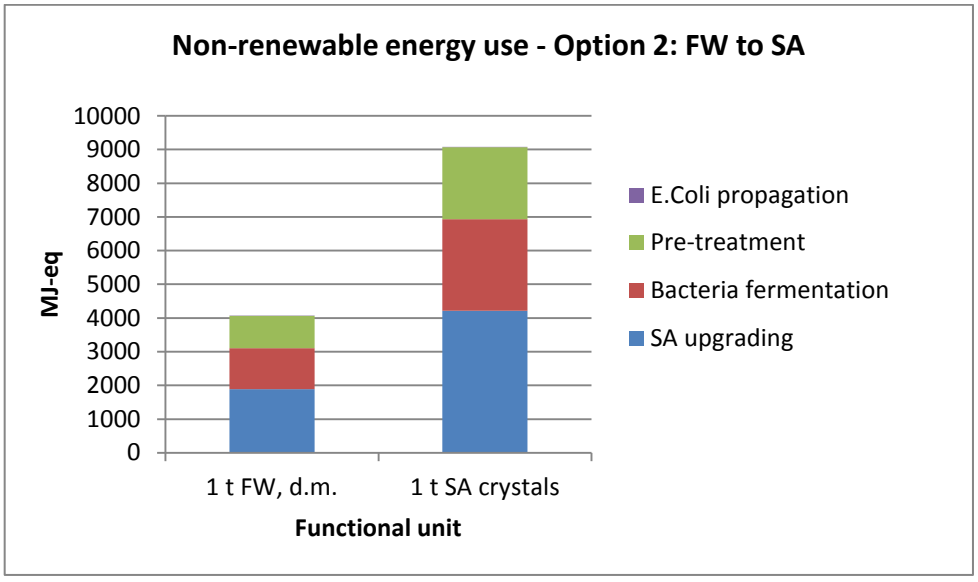

Figure 17: Non-renewable energy use of Option 2: FW to SA.

For processing 1 tonne of food waste d.m., the non-renewable energy use is 2.1 MJ-eq for Option 1: FW to biogas and 4 070 MJ-eq for Option 2: FW to SA. When treating food waste, production of biogas is an environmentally better option than production of succinic acid. Based on previous studies, the Non-renewable energy use for SA production from corn is between 28.0 – 66.5 MJ / kg bio-based SA (European Comission, n.d.). The NREU result of this study for Option 2: FW to SA is lower than these results.

There is large difference in NREU between Option 1 and Option 2. For Option 1: FW to biogas, 71% of the NREU impact origins from the pre-treatment. For Option 2: FW to SA, the largest impact is from the SA upgrading. The NREU impact for Option 1 is almost completely due to electricity while for Option 2 most of the impact originates from production of steam and NaCl. The production of these thereby has a much larger impact compared to the electricity.

In Figure 18 the total NREU impact is presented for the functional unit 1 tonne of produced SA crystals. These are the same values as presented in Figure 17 used again to be able to compare the results to Option 3: Corn to SA. For production of 1 tonne SA crystals for Option 2: FW to SA, the Non-renewable energy use is 9080 MJ-eq or 9 GJ- eq. Cok et al. (2014) report that the NREU results for 1 kg SA crystals in the Reverdia DC process is 32,7 MJ/kg SA, i.e. 32 700 MJ/tonne SA. Thus for production of 1 tonne SA crystals, the Non-renewable energy use is lower for Option 2: FW to SA than for Option 3: Corn to SA, see Figure 18. Thereby, when producing succinic acid, using food waste is and environmentally better option than using corn.

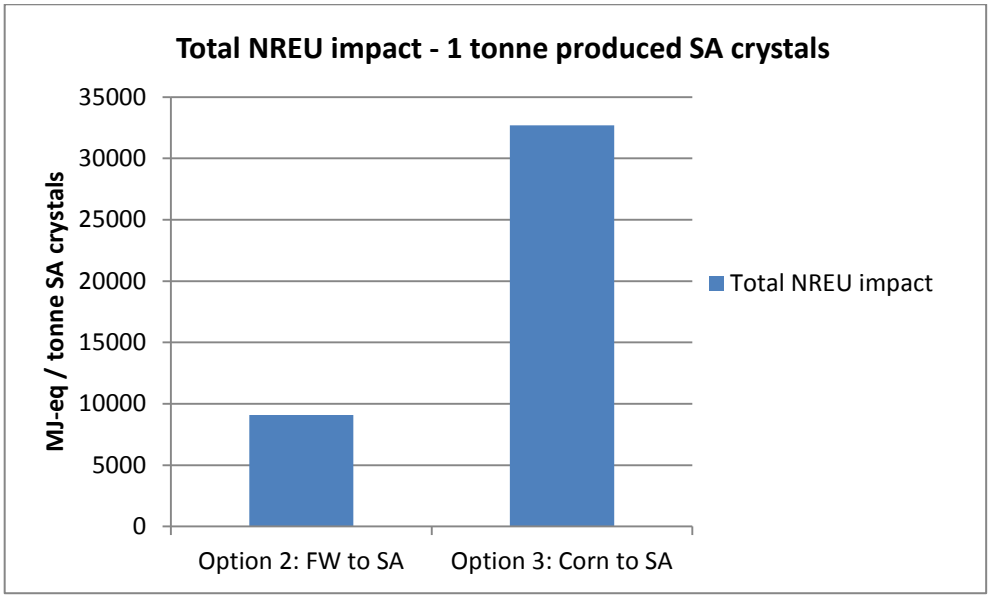

Figure 18: Total Non-renewable Energy Use for the functional unit 1 tonne of produced SA crystals. The results for Option 3: Corn to SA are based on Cok et al. (2014).

4.2.6 Renewable Energy Use

The results of the renewable energy use are presented in Figure 19 and 20.

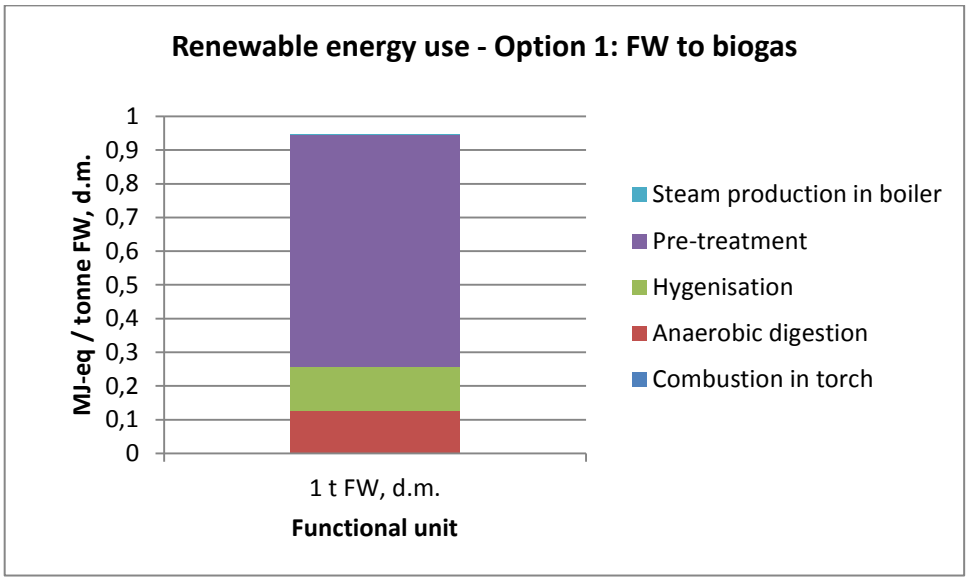

Figure 19: Renewable energy use of Option 1: FW to biogas.

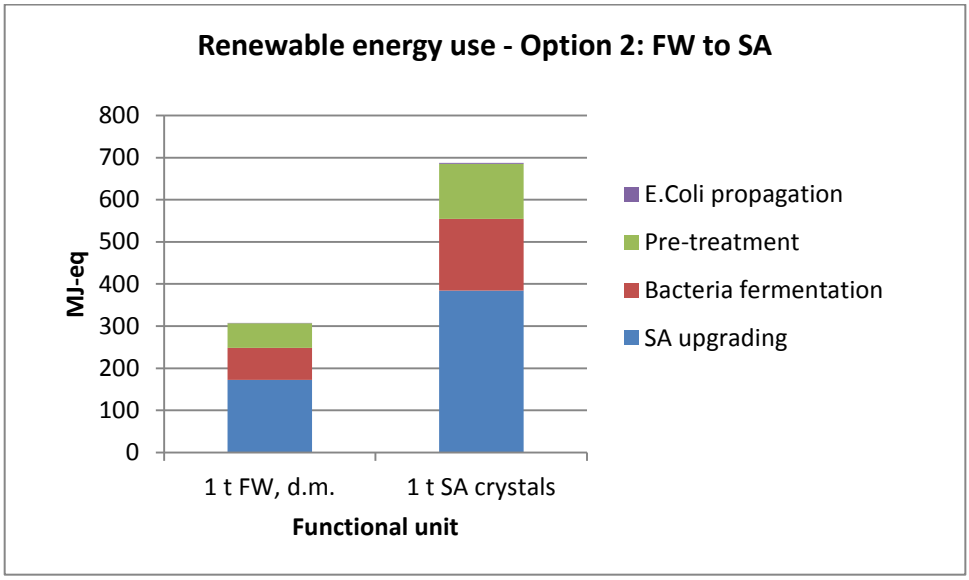

Figure 20: Renewable energy use of Option 2: FW to SA.

For processing 1 tonne of food waste d.m., the renewable energy use is 0.95 MJ-eq for Option 1: FW to biogas and 310 MJ-eq for Option 2: FW to SA. When treating food waste, production of biogas is an environmentally better option than production of succinic acid. For production of 1 tonne SA crystals, the renewable energy use is 690 MJ-eq for Option 2: FW to SA.

There is large difference in REU between Option 1 and Option 2. For Option 1: FW to biogas, 73% of the REU impact origins from the pre-treatment. For Option 2: FW to SA, the largest impact of 56 % is from the SA upgrading. For Option 1, the REU impact is almost completely due to electricity. The REU impact for Option 2 mostly stems from production of NaCl, electricity and enzymes. Even though electricity is a

one of the most contributing factors to the REU impact for option 2, many other flows contributes as well and the electricity stands for a smaller share of the total impact in Option 2 than in Option 1. These other flows, mainly NaCl and enzymes thus accounts for the larger impact for Option 2.

4.2.7 Summary of results

To summarise, when comparing Option 1: FW to biogas and 2 for the functional unit 1 tonne of processed food waste, Option 1: FW to biogas clearly gives much lower results in every impact category. Based on the results from this study, production of biogas is thereby a better option to handle food waste than to produce succinic acid from an environmental point of view. When comparing Option 2: FW to SA and Option 3: Corn to SA for the functional unit 1 tonne produced SA crystals, Option 2: FW to SA gives lower environmental impact for both GWP and NREU. Based on the results of this study, food waste is an environmentally better option than corn as a feedstock for production of bio-based SA.

The process Combustion in torch gives zero impact in all impact categories. This is because the only emission from the torch is biogenic CO_2 from combustion of biogas. Biogenic CO_2 does not give rise to any impact, thereof no contribution. If the biogenic CO_2 is modelled as fossil CO_2, the impact becomes visible. Modeling the biogenic CO_2 emissions in the Boiler and Torch as fossil CO_2 is tested as a model change in the sensitivity analysis, see Chapter 6. The E.Coli propagation process is shown to give zero impact in all impact categories in Option 2: FW to SA. The impact is not zero but so small that is negligible compared to the impact of the other processes.

The transport of waste to the biogas facility respectively the food waste bio-refinery was excluded since it according to Bernstad and la Cour Jansen (2012) seldom is of importance to overall environmental impacts from a biogas facility. However Lundie and Peters (2005) found that the transportation of organic waste for composting accounted for most of the impact when composting food waste. The impact of the biogas production in this study turned out to be very low, so including the transportation of waste to the biogas facility would likely increase the impact results.

It is essential to remember that this analysis is a cradle-to-gate LCA, thereby only the production phase is included. The environmental impact of the use phase and end of life is not included in the results. Several waste streams and by-products are produced in the modelled options, e.g. combustible waste and biofertiliser in Option 1: FW to biogas, respectively biomass and different types of waste in Option 2: FW to SA and Option 3: Corn to SA. Their use or end of life treatment is not included in this study.

It is also important to remember that the results likely could be very different if system-expansion is used. The results would differ if biogas and biofertilisers would replace other products or if succinic acid from food waste would replace production of fossil based succinic acid. The results would likely also be very different if a full cradle-to-grave LCA was performed. For biogas, the environmental impact of final use of biogas in vehicles is about 4 times higher than combustion of biogas in the torch and boiler combined when using the weighting method Eco Indicator 99 (Ljungkvist, 2008). On the other hand, according to the BREW project (2006), the use phase impact of succinic acid can be considered as negligible.

5 Sensitivity Analysis

In this chapter the results of the sensitivity analysis are presented. A description of the different base cases and evaluated model changes can be found in Chapter 3, Section 3.3.2.7.

5.1 Option 1: FW to Biogas

The results of the sensitivity analysis for Option 1: FW to biogas is presented are Table *2.* The results are presented in two units: mass equivalents in each impact category and the impact change in percent compared to the base case.

Table 2: Sensitivity analysis of Option 1: FW to biogas for the functional unit 1 tonne of processed food waste, d.m.

Impact category	Unit	Base case	Model Change		
			Biogenic CO_2 modelled as fossil CO_2	Including CO_2 from food production	No allocation in Anaerobic Digestion
GWP	kg CO_2-Eq	0.022	37.5	8.38	12.2
	%	n/a	174 000	38 700	56 400
AP	g SO_2-Eq	0.11	0.11	0.11	61
	%	n/a	0	0	56 400
EP	g NOx-Eq	0.09	0.09	0.09	51
	%	n/a	0	0	56 400
HTP	kg 1.4-DCB-Eq	0.024	0.024	0.024	13
	%	n/a	0	0	56 400
NREU	MJ-Eq	2.07	2.07	2.07	1 170
	%	n/a	0	0	56 400
REU	MJ-Eq	0.95	0.95	0.95	540
	%	n/a	0	0	56 400

Modelling the biogenic CO_2 emitted from combustion of landfill gas and biogas in the boiler and the torch as a fossil CO_2 to make the biogenic CO_2 visible gives a massive increase in the GWP impact, almost 180 000 %. This means that much biogenic CO_2 is emitted from the biogas facility. This GWP result is still much lower than for Option 2: FW to SA. Including the CO_2 impact from food production also gives a higher GWP impact, but not as large as modelling the biogenic CO_2 as fossil CO_2. If no allocation is used in the anaerobic digestion process, the environmental impact in all impact categories is increased by around 56 000 %. Thereby, both the CO_2 impact from food production and the allocation method are important parameters for the total result.

All these model changes increase the environmental impact to a large extent. However, even if all three model changes were combined, the environmental impact is still lower for Option 1: FW to biogas compared to Option 2: FW to SA except for REU. This means that producing biogas is still an environmentally better option to treat food waste compared to producing SA except for REU if no allocation is used in the Anaerobic Digestion process.

5.2 Option 2: FW to SA

The results of the sensitivity analysis for Option 2: FW to SA are divided into four sub-groups and are presented in Table 3-6. The results are available in two units: mass equivalents in each impact category and the impact change in percent compared to the base case. The sensitivity analysis of Option 2 gave the same results in terms of impact change in percent for both functional units except when changing the yield in the upgrading process. Therefore, only results for the functional unit 1 tonne of produces SA crystals are presented.

5.2.1 Modelling of CO_2

The results of the sensitivity analysis including model chances regarding the modelling of CO_2 are presented in Table 3.

Table 3: Sensitivity analysis of Option 2: FW to SA regarding modelling of CO_2 for the functional unit 1 tonne of processed food waste, d.m.

Impact category	Unit	Base case	Model Change		
			Including CO_2 from food production	Biogenic CO_2 in fermentation	Fossil CO_2 uptake in fermentation
GWP	kg CO_2-Eq	667	2 730	634	592
	%	n/a	310	-5	-11
AP	kg SO_2-Eq	2.56	2.56	2.47	2.47
	%	n/a	0	-4	-4
EP	kg NOx-Eq	2.31	2.31	2.27	2.27
	%	n/a	0	-2	-2
HTP	kg 1.4-DCB-Eq	580	580	533	533
	%	n/a	0	-8	-8
NREU	MJ-Eq	9 080	9 080	8 660	8 660
	%	n/a	0	-5	-5
REU	MJ-Eq	687	687	659	659
	%	n/a	0	-4	-4

Modelling biogenic CO_2 or a fossil CO_2 uptake in the Bacterial fermentation process instead of liquid CO_2 reduce the impact for all impact categories by up to 11 %. How the CO_2 input in is modelled does not have a large effect on the total impact in any impact category. Including CO_2 from food production increase the GWP impact by around 300 % and is thereby an important parameter for the GWP results. One important result is that when including CO_2 from food production, the GWP impact of Option 2: FW to SA exceeds the GWP impact of Option 3: Corn to SA of 2 340 kg CO_2- eq/ tonne SA crystals (Smidt et al., 2015). Thereby, if including CO_2 from food production, corn is an environmentally better feedstock for production of succinic acid than food waste.

If data regarding CO_2 input to the fermentation from Lam et al. (2014) is used, the GWP impact results become unreasonably high. To get more reasonable results, the CO_2 input was instead calculated using a mass balance based on theoretical optimal reaction for transforming glucose to SA, presented by van Heerden et al. (2013), see Appendix C, Section C.3. All supplied CO_2 was also assumed to be converted to SA, so in reality the input of CO_2 would likely be higher.

To achieve an industrial scale production of bio-based SA, carbon dioxide for the fermentation needs to be concentrated and supplied from a low-cost source (Jansen and van Gulik, 2014). This could be achieved by using off-gas from an ethanol or fermentation plant. It could thereby be a good idea to construct an SA fermentation plant in connection to an existing ethanol plant or other plant producing CO_2, for example the Hejlestorp biogas facility. Wu et al. (2011) have coupled SA production with ethanol fermentation using E.Coli and recycled the CO_2 from the ethanol fermentation in the SA production. Thus SA and ethanol can be co-produced while reducing CO_2 emissions from ethanol production. The result of this study also show that by using either fossil or biogenic CO_2 from another source, the GWP impact can be reduced compared to using liquid CO_2. Using a renewable feedstock and fixating CO_2 during fermentation thus gives bio-based SA an environmental advantage compared to fossil-based SA (Chimirri et al., 2010).

5.2.2 Yield

The results of the sensitivity analysis when changing the yield in the upgrading process are presented in Table 4.

Table 4: Sensitivity analysis of Option 2: FW to SA regarding yield for the functional unit 1 tonne of processed food waste, d.m.

Impact category	Unit	Base case	Model Change	
			30 % yield in upgrading process	90 % yield in upgrading process
GWP	kg CO_2-Eq	667	1 330	445
	%	n/a	100	-33
AP	kg SO_2-Eq	2.56	5.12	1.71
	%	n/a	100	-33
EP	kg NOx-Eq	2.31	4.63	1.54
	%	n/a	100	-33
HTP	kg 1.4-DCB-Eq	580	1 160	387
	%	n/a	100	-33
NREU	MJ-Eq	9 080	18 200	6 050
	%	n/a	100	-33
REU	MJ-Eq	687	1 370	458
	%	n/a	100	-33

Changing the yield in the SA upgrading process affects all categories by – 33% when increasing the yield to 90 % or by +100 % when decreasing the yield to 30%. Thereby the yield in the upgrading process is an important parameter for the result for all impact categories. That the yield is of high importance for the total environmental impact and that a low yield results in higher impact is in line with the results presented by Janssen et al. (2016).

Changing the yield was the only model change which gave different results between the two functional units. For the functional unit 1 tonne of processed food waste, d.m., changing the yield in the SA upgrading process did not affect the total impact. This is reasonable since 1 tonne of food waste is still processed regardless of the yield. The output of SA crystals does however depend on the yield, thereof the change in impact for the functional unit 1 tonne of produced SA crystals.

The upgrading yield of the Reverdia DC process used for Option 3: Corn to SA is not publically known. The yield in the SA upgrading process in Option 2 is unknown but assumed to be 60 %. For the pilot plant producing SA from waste bread described by Lam et al. (2014) which the modelling of Option 2 is based on, 25 388 kg SA crystals was produced from 312 tonnes of bread. This means that only 8% of the mass is converted from bread to SA crystals. At the same time the lab scale yield of SA in the fermentation process is 0.55 g SA/g bread and 0.224 g SA/g food waste (Sun et al.,

2014). If 0.55 g SA/g bread is produced in the fermentation process in the bread pilot plant, the upgrading yield for bread must be around 15 % to obtain 25 388 kg SA crystals. There is thus reason to think that the yield of the upgrading process in Option 2: FW to SA could be lower than 30%, which was the lowest upgrading yield modelled in this study. If that is the case, the environmental impact of producing 1 tonne SA crystals from food waste would be even higher.

5.2.3 Process input changes

The results of the sensitivity analysis for changing different inputs are presented in Table 5.

Table 5: Sensitivity analysis of Option 2: FW to SA regarding process input changes for the functional unit 1 tonne of processed food waste, d.m.

Impact category	Unit	Base case	Model Change		
			Tenfold enzyme increase	Excluding DDGS input	Thousandfold increase of E.Coli
GWP	kg CO_2-Eq	667	1 260	561	852
	%	n/a	89	-16	28
AP	kg SO_2-Eq	2.56	2.8	2.03	4
	%	n/a	9	-21	56
EP	kg NOx-Eq	2.31	2.74	0.97	5.1
	%	n/a	19	-58	122
HTP	kg 1.4-DCB-Eq	580	1 320	579	636
	%	n/a	128	0	10
NREU	MJ-Eq	9 080	11 300	9 080	10400
	%	n/a	24	0	14
REU	MJ-Eq	687	975	687	2790
	%	n/a	42	0	306

Increasing the enzyme use ten times increase the impact in all categories in varying extent. The effect is largest for HTP with an increase of around 130 %, and second largest for GWP with an increase of 90%. The enzyme use is thereby an important parameter for the GWP and HTP. Excluding the DDGS input in the Bacterial fermentation process reduce the impact for GWP, AP and EP. The largest effect is for EP where the impact is reduced by 60 %. Increasing the requirement of E.Coli by a 1000 times increase the impact in all categories, but most significantly for REU by 300 % and EP by 120 %. Even though an increase of 300% can be considered high, this is for a 1000 times increase of the E.Coli requirement. This means the E.Coli requirement does not have a large effect on the impact results.

The DDGS is added to supply extra nitrogen. However, both Sun et al. (2014) and Kiran et al. (2014) have concluded that food waste can be used as the only raw material for value-added bio-products without adding supplementary nutrients. If that is the case, an input of DDGS might not be needed and the environmental impact of Option 2: FW to SA would be lower. Besides nitrogen DDGS also contain carbon and other nutrients. This carbon could possibly be used for the fermentative SA production and thus lead to a lower requirement of food waste to supply carbon. This has not been considered in the modelling, instead the food waste is assumed to contain enough carbon to supply the SA production. For further work it would be relevant to include a mass balance of carbon to evaluate how much SA could be produced based on the combined carbon and nitrogen content of food waste and DDGS.

5.2.4 Allocation

The results of the sensitivity analysis for different allocation methods are presented in Table 5.

Table 6: Sensitivity analysis of Option 2: FW to SA regarding allocation for the functional unit 1 tonne of processed food waste, d.m.

Impact category	Unit	Base case	Model Change	
			Economic allocation in SA upgrading	No allocation in SA upgrading
GWP	kg CO_2-Eq	667	3 760	4 970
	%	n/a	463	644
AP	kg SO_2-Eq	2.56	14.4	19.1
	%	n/a	463	644
EP	kg NOx-Eq	2.31	13	17.2
	%	n/a	463	644
HTP	kg 1.4-DCB-Eq	580	3 270	4 320
	%	n/a	463	644
NREU	MJ-Eq	9 080	51 100	67 500
	%	n/a	463	644
REU	MJ-Eq	687	3 870	5 110
	%	n/a	463	644

Changing the allocation method in the SA upgrading process from mass allocation as used in the base case to either economic allocation or no allocation has the largest effect on the results out of all tested model changes. Economic allocation increase the result in all impact categories increase by 460 % while no allocation increase the result for all impact categories by 640 %. The allocation method is thereby a critical parameter for the total results. This is in line with the conclusion by Heimersson et al. (2014) that the choice of allocation method to a large extent decides the LCA results.

One important result is that when using economic allocation or no allocation in the SA upgrading process, the GWP and NREU impacts exceeds the impacts for Option 3: Corn to SA of 2 340 kg CO_2-eq/tonne SA and 32 700 MJ/tonne SA (Smidt et al., 2015, Cok et al., 2014). Thus when using another allocation method than mass allocation, corn is an environmentally better feedstock option for production of succinic acid than food waste. The highest results for bio-based SA reported by European Comission (n.d.) were 3.1 kg CO_2-eq / kg SA for GWP and 66.5 MJ / kg bio-based SA for NREU. When using economic or no allocation, the results of Option 2: FW to SA exceeds the GWP results of the European Comission (n.d.) while the NREU results are similar.

In the LCA studies of the Reverdia DC process by Cok et al. (2014) and Smidt et al. (2015) mass allocation was used except for the CHP plant where system expansion was used. At the same time Cok et al. (2014) claim that no allocation is required for the process of producing SA from dextrose. This likely means that mass allocation was used for the corn to dextrose process but no allocation was used for the dextrose to succinic acid process. The LCA of the Reverdia DC process thus seems to use no allocation and includes the environmental impact of corn production. If that is the case, it would be most fair to compare the result of the Option 3: Corn to SA with the results of SA production from food waste when no allocation is used and the CO_2 impact of food production is included. In this case, the environmental impact of producing succinic acid from food waste is higher than the impact from Reverdia DC process using corn for both GWP and NREU. Thereby corn would again be an environmentally better feedstock to use than food waste.

It is problematic when allocation has such a large effect to the overall impact. Which allocation method is most relevant to use is subjective, although the ISO standard state the prioritized order of firstly avoiding allocation, then allocation based on physical relationships and lastly allocation based on other relationships (Baumann and Tillman, 2004). It is important to use the same allocation method when comparing results to get a fair comparison.

To summarize, the result of this sensitivity analysis show that whether food waste or corn is the best option for production of 1 tonne of SA crystals from an environmental point of view depends on the modeling choices. Food waste is an environmentally better feedstock option compared to corn only if CO_2 from food production is excluded and if mass allocation is used in the SA upgrading process. If CO_2 from food production is included or if economic allocation or no allocation is used in the SA upgrading process, corn is an environmentally better feedstock than food waste.

6 Discussion

This Chapter discusses limitations of the study and LCA model, availability of food waste in Sweden and food waste as a feedstock. A discussion regarding economic barriers for SA production, integrated bio-refineries and suggestions for further work are also found in this chapter.

6.1 Limitations of the study

The main issues with the modelling in this study the limited availability of data. There is a large uncertainty in the technical layout of the process in Option 2: FW to SA. Moreover, there is no available large scale process using food waste for production of succinic acid which makes the process difficult to model. This has led to uncertainties and incomplete modelling as well as the need for assumptions.

The industrial processes for fermentative production of bio-based SA are recently implemented. Details of how these processes work are company secrets, making it difficult to get access to relevant data. According to Heimersson et al (2014) it can difficult to perform a relevant LCA for emerging technologies with low experience of full-scale implementation due to lacking data or unknown application areas. Such an LCA can however still be valuable for guiding technology development and finding areas which can be environmentally improved.

In 2007, Hatti-Kaul et al. (2007) stated that there were a rather limited number of LCA studies addressing the environmental aspects of bioprocesses and bio-based chemicals. In the few studies available, it was reported that using a renewable feedstock is not always the best option from an environmental point of view. Although using industrial biotechnology for producing chemicals can result in a cleaner production process, the benefits of a cleaner production must be related to the overall energy and material requirements for a product. It is thus important to evaluate the environmental impact for the products entire life cycle.

6.2 Limitations of the LCA model

As mentioned previously, only one model of Option 2: FW to SA was constructed in the software openLCA. The reference flow in the product system was set to 1 tonne of produced SA crystals. 1 tonne of food waste produces 0.448 tonne SA crystals when a yield of 60% is assumed in the SA upgrading process. To get result for the functional unit 1 tonne of processed food waste, the reference flow was set to 0.448 tonnes SA crystals. This is the reason for why the impact results of Option 2: FW to SA differs by a factor of 2.23 between the two used functional units. Using the same reference flow in this way in the openLCA model might not be correct. In retrospect, it could have been better to build two separate models for Option 2: FW to SA in openLCA, using two different reference flows.

6.3 Availability of Food Waste in Sweden

Even if production of SA from food waste is technically possible, there must be enough food waste available to supply the production. According to the modelling in this study, 1 tonne of food waste, d.m. gives 0.448 tonnes SA crystals. The existing industrial production plants for bio-based SA have an annual production capacity of 10 ktonnes or more (Cok et al., 2014). Thus the minimum capacity for an economically feasible production is reasonably 10 ktonnes per year. Based on this study, around 22.3 ktonnes food waste, d.m. is required to produce 10 ktonnes SA crystals. If the food waste has a dry mass of 30 %, this corresponds to 74.3 ktonnes food waste, w.m.. Similar numbers are presented by Lin et al. (2013) which claim that 27 ktonnes of waste bread is required for production of 15 ktonnes SA, assuming an overall yield of 0.55 g SA per g bread. The annual amount of food waste in Sweden is 1.2 million tonnes (Naturvårdsverket, 2014). Thereby there is more than enough food waste in Sweden to supply an annual production of 10 ktonnes SA crystals.

6.4 Food Waste as a Feedstock

It is important to remember that the focus of food waste management should follow the food recovery hierarchy by the US EPA (2015). Thus it is most important to firstly minimize surplus food and food waste and secondly to use the surplus food to feed humans and animals. Avoiding food waste is around ten times more efficient than treating the produced food waste biologically, but at the same time it is important to take care of the existing food waste in a resource efficient way (Naturvårdsverket, 2014). In contrast to many other waste streams, food degrades over time (Papargyropoulou et al., 2014). Thus edible food can become inedible due to decomposition. Time is therefore an important aspect to consider when discussing food and food waste since the properties of food waste as a result of time affects the position in the food waste hierarchy.

Although food waste has great potential as a renewable feedstock for bio-based chemicals and materials, there are also barriers to overcome. Challenges with using food waste are varying chemical composition, variations in available volumes and high moisture content which requires a robust process (Lin et al., 2013). Laws and regulations may also limit the usage of food waste for some applications. Contamination of food waste might be a problem which can make the down-stream purification of SA more difficult (Uçkun Kıran et al., 2015).

6.5 Economic Barriers for SA production

To achieve a production of bio-based SA in industrial scale, the process must be economically feasible. Biological production of SA is currently not economically competitive compared to fossil based production (Cheng et al., 2012). Reasons for this are expensive feedstocks, low concentration of SA after fermentation, formation of low value by-products and a complicated recovery process The downstream processing usually accounts for 30-40 % of the total production costs (López-Garzón and Straathof, 2014).

Several things need to be resolved to achieve an economically feasible process. The SA yield from microorganisms need to be increased through metabolical engineering and they must be able to produce SA from a number of different sugars (Cheng et al., 2012) (Chimirri et al., 2010). The costs of the recovery and purification process must be decreased. Further research on organisms will enable new fermentation reactor designs for increased productivity and thereby increase the economic viability of a bio-based SA production (van Heerden and Nicol, 2013).

6.6 Integrated Bio-refineries

The cost and efficiency of fermentative SA production could be improved by using an integrated bio-refinery to co-produce various products (Uçkun Kıran et al., 2015). Such a bio-refinery would be expensive since the technology is immature and requires further development. This might however be compensated by environmental benefits, low feedstock and waste disposal costs. Only a few valorization techniques have been tested in pilot scale. There is a need for more optimization and scale-up studies to evaluate if using food waste for production of bio-based chemicals is feasible.

Industrial biotechnology has potential to co-produce bio-based chemicals and energy and add economic value to waste biomass (Hatti-Kaul et al., 2007). To realize the transfer from a petrochemical to a bio-based production, there is also a need to develop a market for biofuels, which subsequently is driven by policies promoting bio-based products. To drive the development and increase investments, policy instruments promoting the use of renewable feedstocks should be implemented.

6.7 Further Work

The LCA results evaluating early bio-refinery technologies might later become irrelevant due to new technology development (Bozell and Petersen, 2010). An LCA for emerging technologies as presented in this study can however still be valuable for guiding technology development and finding areas which can be environmentally improved (Heimersson et al., 2014). This study could be seen as an early stage screening LCA which can be used as input to further research.

The main issues within the project which needs to be addressed in further work are:

- **Data availability:** Go through the data collection for Option 2: FW to SA to find more extensive data since the technology setup is unsure. It is especially important to evaluate what yield in the upgrading of SA is technically possible since it is unknown and has a large effect on the result. Also assumptions regarding the input of DDGS and CO_2 could be reconsidered.
- **Upscaling:** Upscaling from lab scale to commercial scale is often difficult (van Heerden and Nicol, 2013). It is thus important to evaluate how parameters and data will change if going from lab scale to pilot scale or industrial scale and to evaluate what would be technically possible in larger scale.
- **Combined openLCA model:** It would be better to construct two separate models for the modelling of producing succinic acid from food waste. In further work, create one model for each functional unit and use separate reference flows.

7 Conclusion

For both production of biogas and SA from food waste, the environmental impact was largely affected by the allocation approach and the inclusion of the CO_2 impact from food production. The impact of the SA production process was also affected by the yield in the upgrading process and the enzyme use.

The results show that from an environmental point of view, production of biogas is a better option to treat food waste compared to production of succinic acid. Whether food waste or corn is the environmentally best feedstock option for production of bio-based succinic acid depends on the modeling choices. If CO_2 from food production is excluded and if mass allocation is used in the SA upgrading process, food waste is an environmentally better option than corn. If CO_2 from food production is included or if economic allocation or no allocation is used in the SA upgrading process, corn is an environmentally better feedstock than corn.

Food waste has a great potential as a renewable feedstock for producing chemicals. There is more than enough food waste in Sweden to supply a pilot plant for succinic acid production. However, the main focus of food waste should be to reduce it as much as possible. To improve the cost and efficiency of fermentative SA production the productivity and profitability must be improved. This could be achieved by using integrated bio-refineries. Further optimization and up-scaling studies are required to evaluate if using food waste for production of bio-based chemicals is feasible.

Modelling the fermentative succinic acid production from food waste was difficult due to limited availability of data and uncertain process layout. Only one model was constructed for this process using the same reference flow for both evaluated functional units. This might not give completely correct modelling results. For further work, it is important to collect better and more extensive data and evaluate how the data could be affected by upscaling.

The results of this study show that the environmental impacts of producing succinic acid from food waste and corn are in the same range. This can be seen as a motivation to proceed with further environmental investigations.

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Appendix A – Technical system descriptions

Appendix A includes further information about the three modelled options and a detailed technical description of each option. Simplified flowcharts of the modelled options are presented and each of the modelled processes is explained. For more detailed flowcharts, see Appendix B.

A.1 Option 1: Food Waste to Biogas

The LCA modelling of Option 1: FW to biogas is based on Rang-Sells sorting and anaerobic digestion facility located in Heljestorp. Information and data about the process were kindly provided by Ragnar Davidsson, Graham Aid and Robert Lippens at Ragn-Sells.

A.1.1 Heljestorp Biogas Facility

The waste management company Ragn-Sells operates a sorting and anaerobic digestion facility, located in Heljestorp. At the Heljestorp biogas facility, organic household waste is converted to biogas, solid and liquid biofertiliser (Ragn-Sells, n.d.). The produced biogas is sold to the nearby town Trollhättan where it is upgraded to vehicle gas. The co-produced biofertiliser is sold to farmers (Personal communication Ragn-Sells, 2016). A process map of Heljestorp biogas facility is presented in Figure A.1.

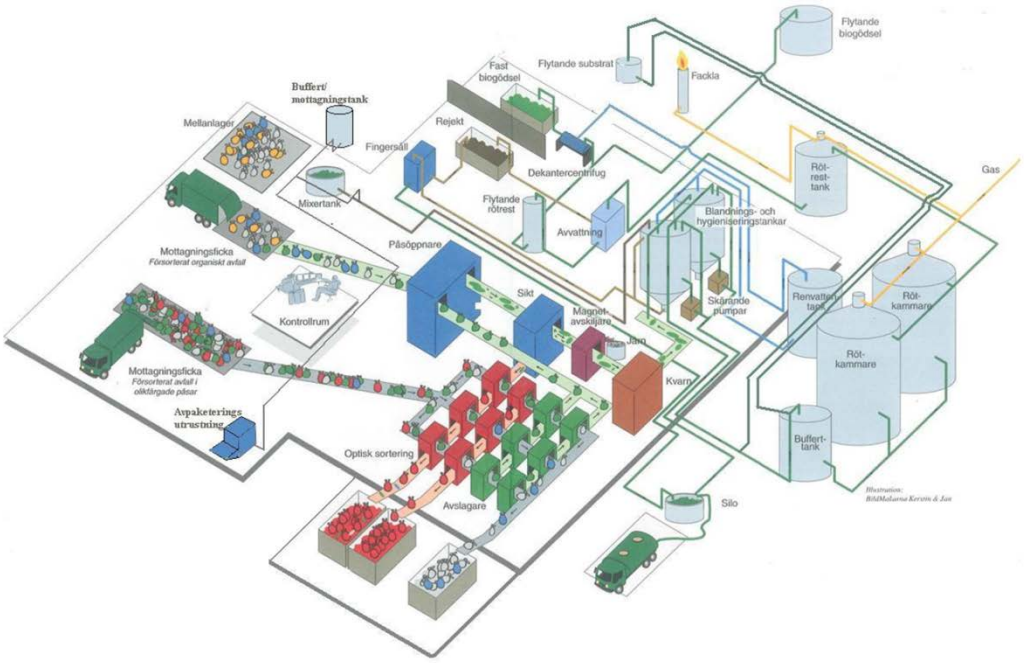

Figure A.1: Process map of the Heljestorp biogas facility. (Source Ragn-Sells, provided by Graham Aid)

A.1.2 Flowchart Option 1: FW to Biogas

The LCA flowchart for this process is presented in Figure A.2. A more detailed flowchart of the foreground system can be found in Appendix B, Figure B.1.

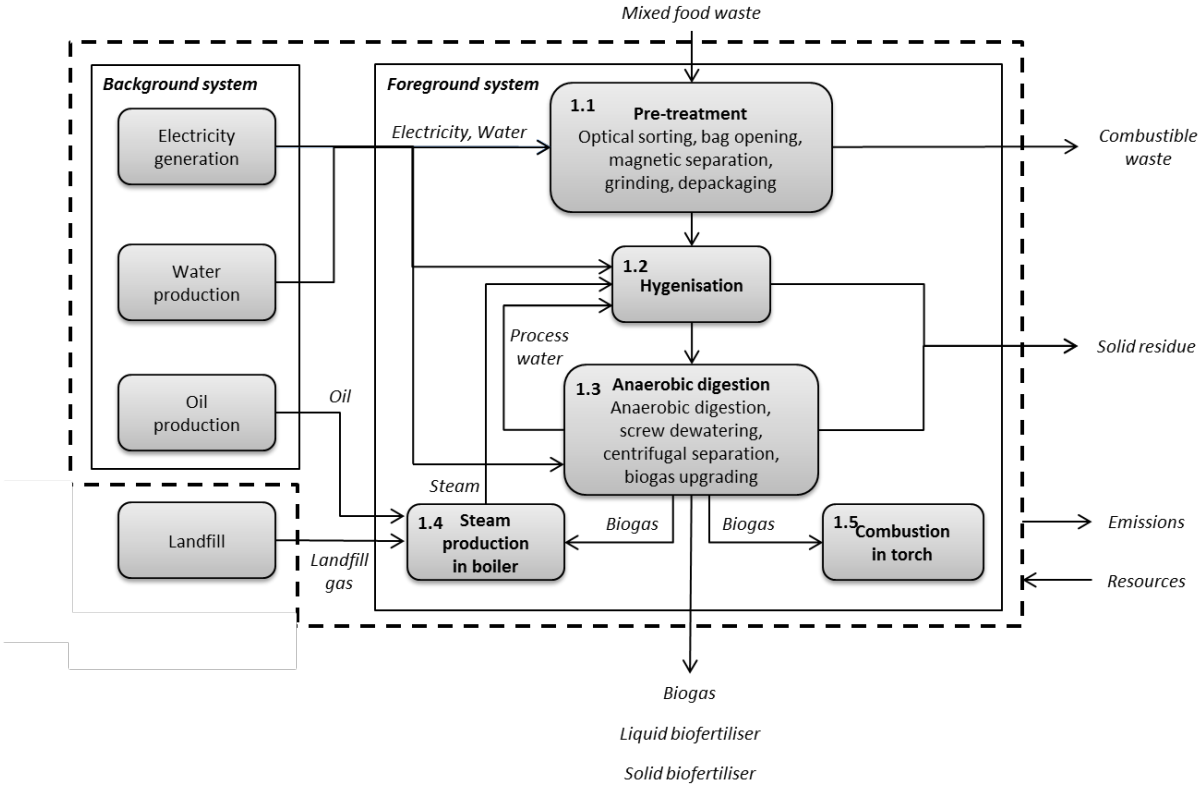

Figure A.2: Simplified flowchart of Option 1: FW to biogas. The dashed lines indicates the system boundary while the solid lines indicate the foreground respectively background system. (Ragn-Sells, n.d.) (Personal communication Ragn-Sells, 2016)

A.1.3 Technical System Description Option 1: FW to Biogas

Households from the municipalities surrounding Heljestorp sort their organic waste into green bags and combustible waste into red bags (Ragn-Sells, n.d.). Ragn-Sells collect the waste and transport it to the biogas production facility in Heljestorp. The transport of waste is not included in this LCA since it is seldom of importance to overall environmental impacts from a biogas facility (Bernstad and la Cour Jansen, 2012). Once the waste has been transported to Heljestorp, the waste enters the Pre-treatment process.

A.1.3.1 Pre-treatment

In the Pre-treatment process an optical sorting machine separates the green bags from the red bags (Ragn-Sells, n.d.). The red bags containing combustible waste are sent to other locations for heat and energy recovery. The green bags are opened and the organic content is sieved to so that the plastic can be separated and collected for heat and energy recovery. The organic waste goes through a magnetic separator, is grinded and transferred to the mixing and hygenisation tanks in the Hygenisation process. The

inputs to the pre-treatment process are electricity, process water and the outputs are combustible waste and grinded food waste.

Apart from household waste the facility also handles some industrial food waste. Packaged waste is opened in a depackaging station where materials are removed (Personal communication Ragn-Sells, 2016). The organic content is transferred into a buffer tank, leading directly into the mixing and hygenisation tanks. Other industrial food wastes such as fat separators are collected in a silo, leading into the mixing and hygenisation tanks.

A.1.3.2 Hygenisation

The hygenisation process involves diluting the grinded and unpackaged food waste with water to a slurry with 8-10 % dry mass content (Ragn-Sells, n.d.) (Personal communication Ragn-Sells, 2016). The slurry is hygenised with steam which is produced in the on-site boiler. The slurry is mixed and circulated by pumps while heavy and light fractions of residue materials are separated from the top and bottom of the tanks and collected as a solid residue. The slurry is then transported to a buffer tank. The inputs to the Hygenisation process are electricity, process water, grinded food waste, depackaged food waste, steam from the boiler and recirculated process water from the anaerobic digestion. The outputs are a hygenised liquid slurry and a solid residue.

A.1.3.3 Anaerobic Digestion

In the Anaerobic digestion process, the slurry from the hygenisation process is transferred to the anaerobic digestion tanks where it is digested for 18 days to produce biogas containing 60 % CH_4 and 40 % CO_2 (Ragn-Sells, n.d.) (Personal communication Ragn-Sells, 2016). The residue material from the anaerobic digestion tanks is further digested in an anaerobic digestion residue tank to produce more biogas. Thereafter, the remaining material is dewatered in a screw dewaterer to obtain a liquid fraction and a solid residue. The solid residue is separated and used to cover the nearby landfill (Personal communication Ragn-Sells, 2016). The liquid fraction is a liquid biofertiliser which is directly transferred to a biofertiliser tank for storage. The liquid fraction can also be separated in a centrifuge to obtain a solid biofertiliser and process water which is reused in the mixing and hygenisation tanks.

Within the Anaerobic digestion process, the produced biogas is dried and the pressure is increased before it is either used on-site or transported to Trollhättan where it is upgraded to vehicle gas (Personal communication Ragn-Sells, 2016). The upgrading to vehicle gas in Trollhättan is not included in the LCA. The inputs to the Anaerobic digestion process is electricity, process water and the food waste slurry. The outputs are biogas, solid biofertiliser, liquid biofertiliser, a solid residue and recirculated process water.

A.1.3.4 Steam Production in Boiler

The onsite boiler is used for producing steam for the hygenisation process as well as to produce hot water for the process and for heating the facilities (Personal communication Ragn-Sells, 2016). The boiler combusts a share of the produced biogas and landfill gas containing 50 % CH_4 and 50% CO_2 which is generated in the nearby landfill. In some cases the boiler also uses heating oil for energy. The inputs to the Steam production in boiler process are thus biogas, landfill gas and heating oil while the outputs are steam, hot water and CO_2 emissions.

A.1.3.5 Combustion in Torch

A share of the biogas is also combusted in a torch located at the facility. This process thus has biogas as the input, and CO_2 emissions as the output.

A.1.3.6 Background Processes

The background processes used in Option1 : FW to biogas are Electricity generation, Water production, Oil production. Data for these processes were obtained from the Ecoinvent database version 3.1, see details in Appendix C and Appendix E.

A.2 Option 2: Food Waste to Succinic Acid

A.2.1 Flowchart Option 2: FW to SA

The LCA flowchart for Option 2: FW to SA is presented in Figure A.3. A more detailed flowchart of the foreground system can be found in Appendix B, Figure B.2. A detailed description of each modelled process is described in the coming sections.

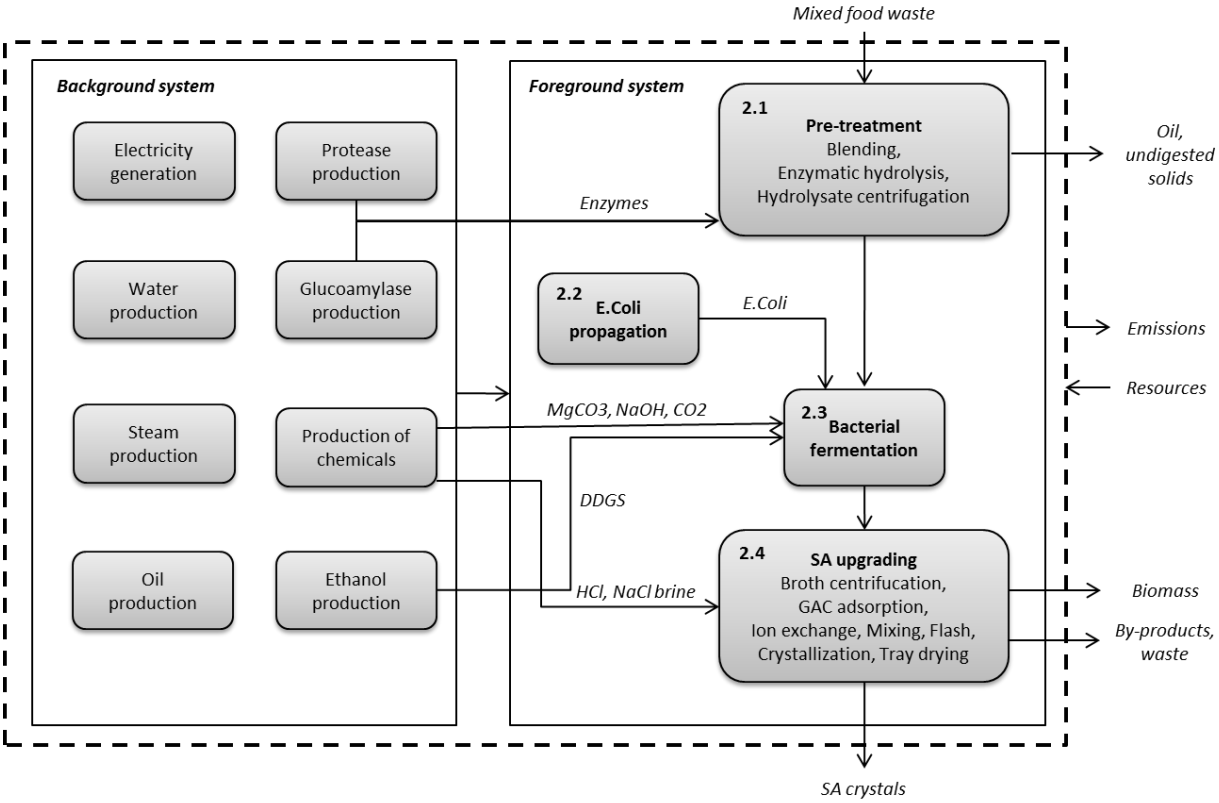

Figure A.3: Simplified flowchart of Option 2: FW to SA. The dashed lines indicates the system boundary while the solid lines indicate the foreground respectively background system. (Modified after Lam et al (2014))

A.2.2 Technical System Description Option 2: FW to SA

The LCA modelling of Option 2: FW to SA is mainly based on literature data of a simulated pilot plant using waste bread for fermentative production of SA, published by Lam et al (2014). These data is assumed to be the same when using food waste instead of bread except the yield of SA in the fermentation and upgrading processes and E.Coli is assumed to be used instead of A. Succinogenes. The yield in the fermentation process is obtained from literature data of lab-scale fermentative SA production from mixed food waste, published by Sun et al. (2014).

A.2.2.1 Pre-treatment

In the Pre-treatment process, food waste is grinded into small pieces and blended with water to form a liquid slurry (Lam et al., 2014). The liquid slurry is mixed with the

commercial enzymes glucoamylase and protease in a bioreactor where enzymatic hydrolysis is conducted for 24 hours and 55 °C. The glucoamylase is used to transform e.g. starch to glucose while protease is used to release amino acids. The slurry is then centrifuged for 15 minutes at 7000 rpm remove oil and undigested solids to obtain a food waste hydrolysate. The inputs to the Pre-treatment process are food waste, electricity, steam, water, glucoamylase and protease. The outputs are a food waste slurry, oil and undigested solids. The oil and undigested solids are not modelled in the LCA since no information regarding the mass or size of these flows are available in Lam et al (2014).

A.2.2.3 E.Coli Propagation

In parallel to the Pre-treatment process, the bacteria E.Coli cultivated in the process E.Coli propagation. This process is modelled using data from a process of fermentation of whey, published in Jungbluth et al. (2007). During fermentation of whey, ethanol, protein concentrate and a yeast paste is produced. The yeast paste is assumed to represent the propagated E.Coli. The E.Coli propagation process has many inputs and outputs, see details in Appendix C.

A.2.2.4 Bacterial Fermentation

The propagated E.Coli and the food waste slurry are transferred to another fermenter where the Bacterial fermentation process takes place (Lam et al., 2014). CO_2 is added continuously to maintain the fermentation. Sodium hydroxide (NaOH) and magnesuim carbonate ($MgCO_3$) and is added to control the pH. The required amount of CO_2 is assumed to follow the theoretical optimal reaction for transforming glucose to SA, presented by Heerden et al. (2013). All supplied CO_2 is assumed to be converted to SA. For details regarding calculations, see Appendix B. 0.1 g propagated E.Coli is assumed to be required to produce 1 kg of SA in the Bacterial fermentation process. The bacterial fermentation is operated at 37 °C for 44 hours and produce a broth containing SA (Lam et al., 2014). The yield in the fermentation is assumed to be 0.224 kg SA per kg food waste (w.m.) according to lab-scale data of fermentative SA production from mixed food waste, published by Sun et al. (2014).

In some cases, food waste does not contain enough nutrients and nutrients such as nitrogen might need to be added (Lin et al., 2013). Therefore an additional input of Dried Distiller's Grain with Solubles (DDGS) is modelled in this LCA. DDGS is a dried by-product from ethanol production which is rich in nitrogen and commonly used for animal feed (Bernesson and Strid, 2011). The DDGS input is assumed to be the product Agrodrank 90 from the company Lantmännen. Agrodrank 90 contains 90 % dry mass out of which 5,7 % is nitrogen. Data regarding the environmental impact of DDGS is obtained from Bernesson and Strid (2011). For details regarding calculations of the nitrogen from DDGS needed to supply the SA fermentation, see Appendix B. The inputs to the Bacterial fermentation process are the food waste

slurry, propagated E.Coli (modelled as yeast paste), $MgCO_3$, NaOH, CO_2, DDGS, electricity and steam. The outputs are a broth containing SA and an exhaust gas. The exhaust gas is not modelled in the LCA since no information regarding the size or content of this flow is available in Lam et al. (2014).

A.2.2.5 SA Upgrading

To obtain SA crystals, the SA needs to be purified from the fermentation broth (Lam et al., 2014). The broth containing SA is first centrifuged to remove biomass which can be sold as fish feed. The broth is processed through a granulated activated carbon (GAC) column where impurities are removed by adsorption on the carbon. The broth is then processed through an ion exchange column to remove by-product organic acids. This is done by keeping the pH in the broth above the pKa of the by-product organic acids but below the pKa for SA. Hydrochloric acid (HCl) and a sodium hydroxide (NaCl) brine solution are assumed to be used in the ion exchange process.

To concentrate the SA in the liquid stream, a flash is used to evaporate more than 97 % of the water (Lam et al., 2014). The flash is kept at a pressure of 1,2 atm and a temperature of 105 °C which results in an SA stream with 44 wt-% water content. In the subsequent crystallization step the stream is cooled to 4 °C to make the solution supersaturated and to form SA crystals. The remaining solution is recirculated and mixed with the stream entering the flash. As a last process step, the SA crystals are dried in a tray drier to obtain dry SA crystals. The end product is anhydrous SA crystals of more than 99 % purity.

The yield of SA crystals in the upgrading process is unknown and therefore a yield of 60 % is assumed. The inputs to the SA upgrading process are the broth containing SA, HCl, NaCl brine solution, electricity, water and steam. The outputs are SA crystals, biomass and other waste streams. The other waste streams are not modelled in the LCA since no information regarding the size or content of these flows are available in Lam et al. (2014).

A.2.2.6 Background Processes

The background processes used in Option 2: FW to SA are Electricity generation, Water production, Steam production, Oil production, Protease production, Glucoamylase production, Ethanol production and Production of chemicals. Data for the production of the enzymes protease and glucoamylase was obtained from Nielsen et al. (2006). Data for the ethanol production generating DDGS as a by-product was taken from the Agroetanol process by the company Lantmännen, obtained in Bernesson and Strid (2011). Data for the other processes were obtained from the Ecoinvent database version 3.1, see details in Appendix E and Appendix G.

A.3 Option 3: Corn to Succinic Acid

A.3.1 Flowchart Option 3: Corn to SA

The LCA flowchart for this process is presented in Figure A.4. The process is based on the Reverdia DC process as described in Cok et al. (2013). A more detailed flowchart of the foreground system can be found in Appendix B, Figure B.3. A detailed description of the process is described in the coming sections.

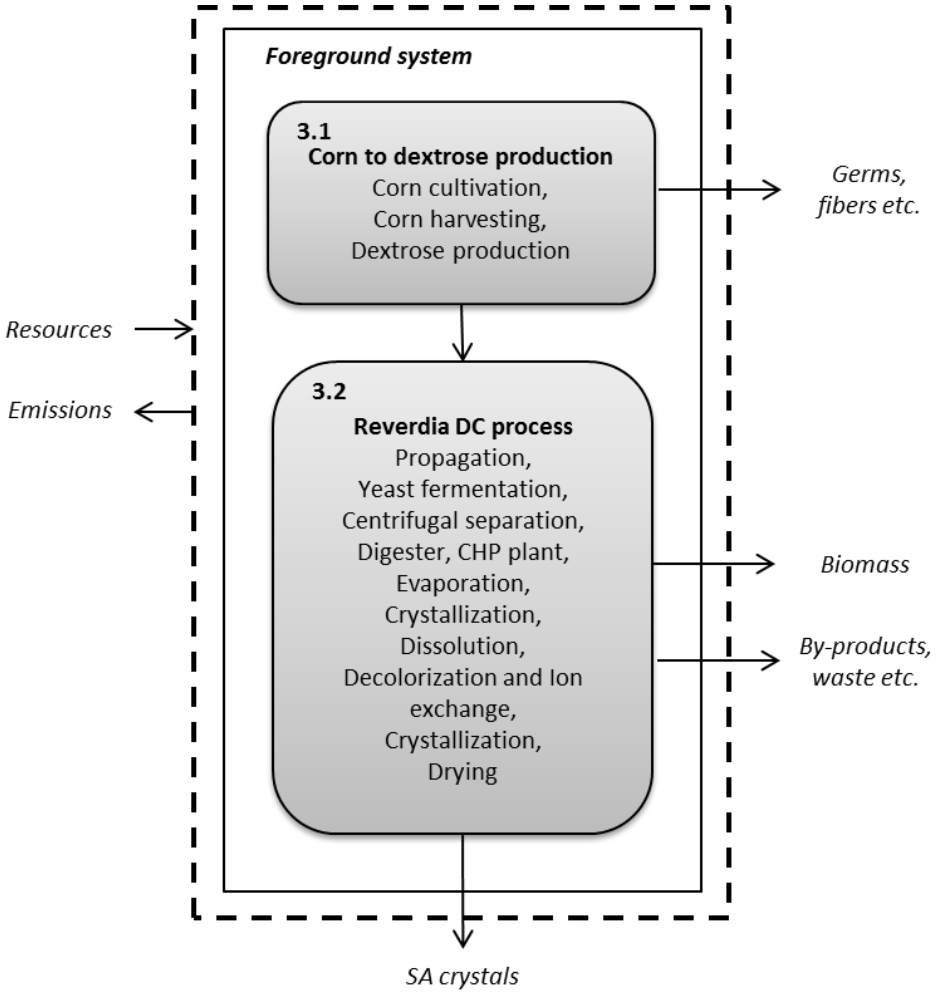

Figure A.4: Simplified flowchart of the foreground system of Option 3: Corn to SA. The dashed lines indicate the system boundary while the solid line indicates the foreground system. (Modified after Cok et al. (2014) and Smidt et al. (2015)).

A.3.2 Technical System Description Option 3: Corn to SA

A.3.2.1 Corn to Dextrose Production

Corn is cultivated, harvested and used as a feedstock for dextrose production. The dextrose is produced in a corn wet mill located in Europe, operating on European energy mix (excluding Switzerland) (Smidt et al., 2015). To obtain the sugar dextrose, the corn must be dried and hydrolyzed in the corn wet mill. The corn wet milling process includes handling of corn, steeping, separation of gluten, fibers and germs and finally starch washing (Tsiropoulos et al., 2013). The obtained starch is converted to glucose, also called dextrose, by addition of water. The yield is 1.11 kg glucose per kg starch since water is incorporated. The corn wet milling process co-produce several products except dextrose and the glucose corresponds to about 68 % of the used corn. Allocation is therefore important. The dextrose produced from corn is used as the raw material for the fermentation (Smidt et al., 2015). The dextrose is assumed to be produced at the same site as the succinic acid (Cok et al., 2014).

A.3.2.2 Reverdia DC Process

The Reverdia DC process use dextrose from corn and other inputs to produce succinic acid crystals (≥99.5 wt-% pure) (Cok et al., 2014).Yeast microorganisms are propagated on dextrose together with nutrients and ammonia (NH3). The yeast is added to a fermenter where it is mixed with CO_2, enzymes and more dextrose. The yeast produces succinic acid through fermentation of the dextrose and absorption of CO_2. After the fermentation, the solution is centrifuged to separate biomass from the fermentation broth containing SA. Hydrochloric acid (HCl) is used to lower the pH in the process liquor.

In the evaporation process, water is evaporated from the fermentation broth containing the SA by using of heat from mechanical vapor recompression (Cok et al., 2014). In the subsequent crystallization process, mother liquor containing 7 wt-% dissolved SA is removed to obtain SA crystals. In the following process steps Dissolution, Decolorization and Ion exchange, Crystallization and Drying, the succinic acid is purified to obtain crystals of high-grade bio-based succinic acid (≥99,5 wt-% pure).

The biomass separated in the centrifugal separation and the mother liquor obtained in the crystallization process is transferred to an on-site digester and converted to biogas (Cok et al., 2014).The biogas is used in the on-site combined heat and power plant (CHP) to replace a share of the used natural gas or coal used for steam production. The CHP co-produce both steam and electricity for the plant. The CHP produces all necessary steam and a share of the required electricity.

Appendix B – Detailed flowcharts

Appendix B contains detailed flowcharts of the foreground system of the three options evaluated in this study.

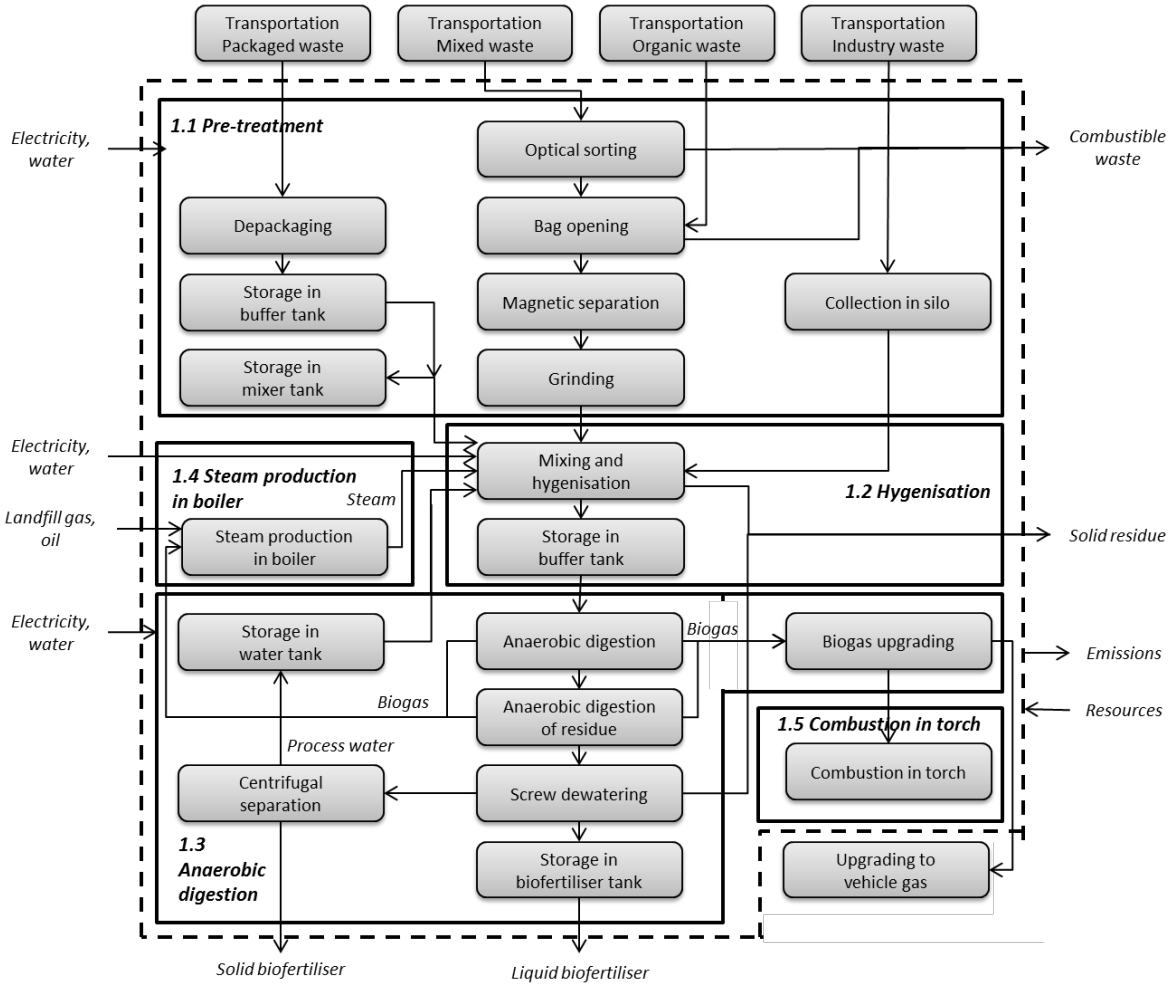

Figure B.1: Detailed flowchart of the foreground system of Option 1: FW to biogas. Modified after (Ragn-Sells, n.d.) and (Personal communication Ragn-Sells, 2016)

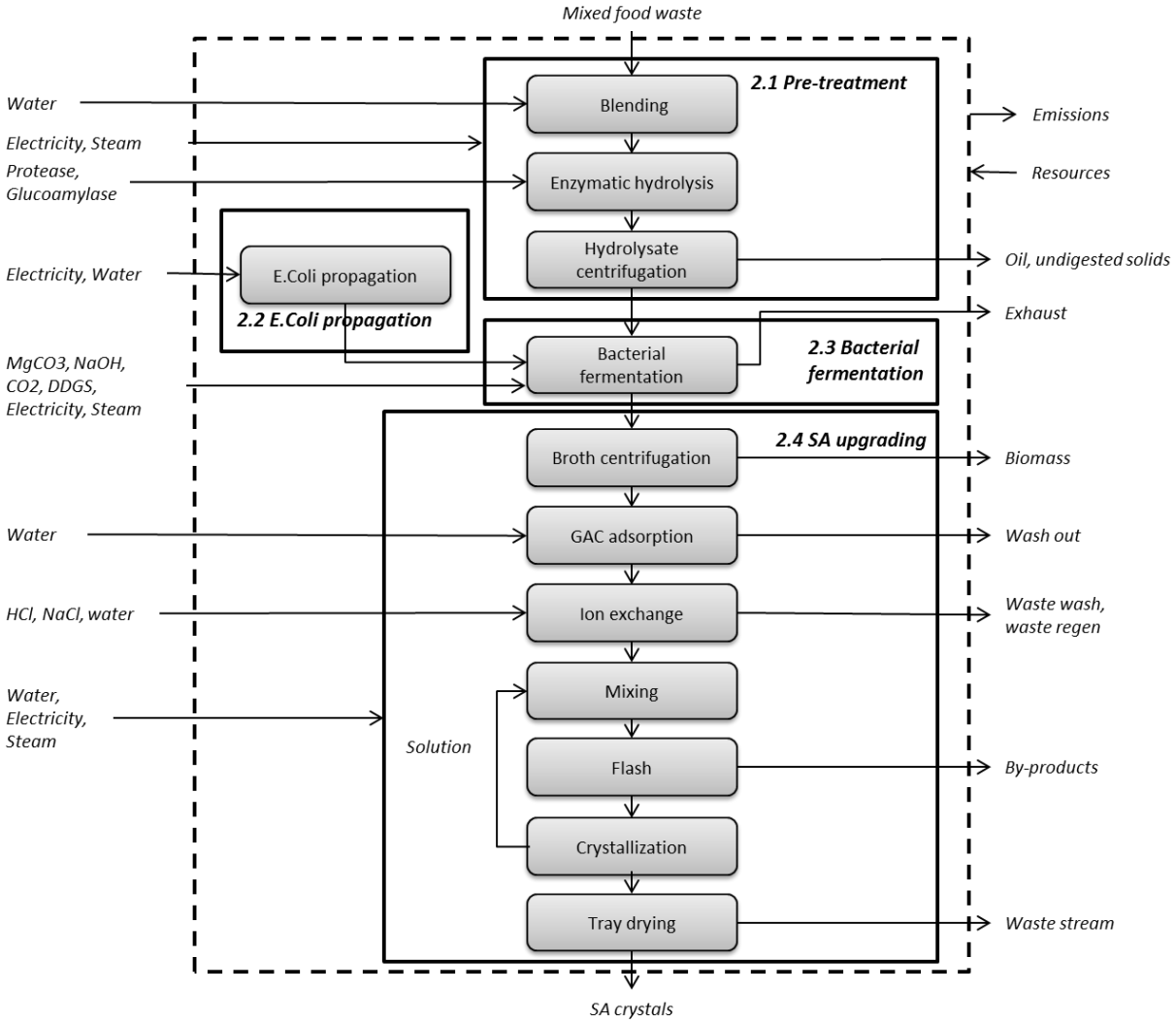

Figure B.2: Detailed flowchart of the foreground system of Option 2: FW to SA. . (Modified after Lam et al (2014))

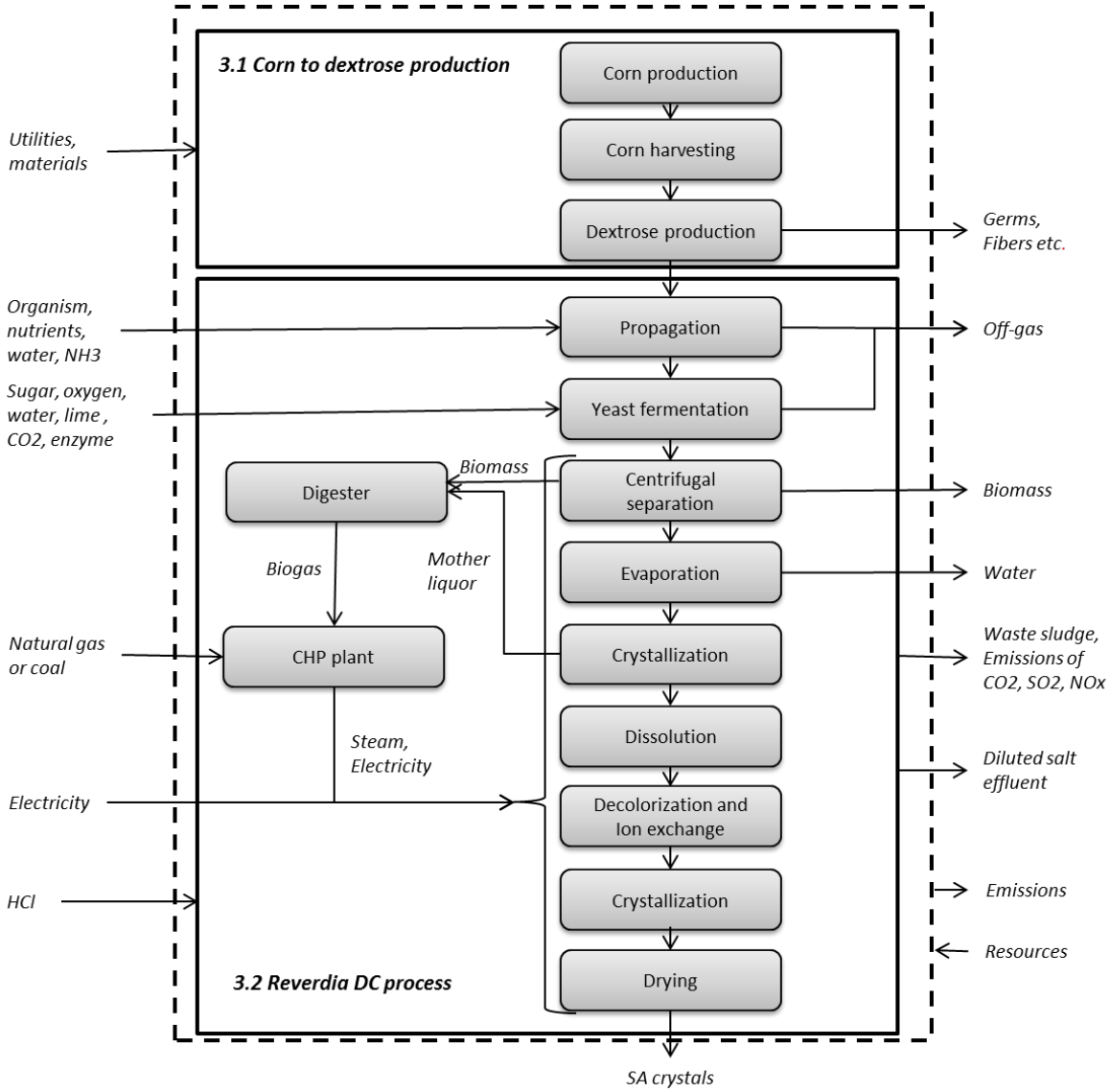

Figure B.3: Detailed flowchart of the foreground system of Option 3: Corn to SA. (Modified after Cok et al. (2014) and Smidt et al. (2015)).

Appendix C – The SA fermentation process

Appendix C presents additional theory regarding the SA fermentation process.
The process for fermentative production of SA generally constitutes of feedstock pre-treatment, seed cultivation, fermentation and at last recovery, purification and concentration, also called down-streaming or upgrading (Song and Lee, 2006). In the coming sections, the process steps in the SA fermentation process are described as well as some benefits with the Reverdia DC process.

C.1 Feedstock Pre-treatment

The first step in fermentative production of SA from food waste is to pre-treat the food waste. Microorganisms require nutrients and materials such as amino acids, sugar monomers, and fatty acids from the food waste to produce SA in the fermentation process (Lin et al., 2014). Before the fermentation, large molecules in food waste must therefore be degraded through hydrolysis. Hydrolysis can be performed by adding alkali or acids, sometimes at high temperatures (Uçkun Kıran et al., 2015).

Enzymatic hydrolysis can also be used in which enzymes breaks down large starch and protein molecules into usable sugars and amino acids (Leung et al., 2012). Such enzymes can either be purchased as commercial enzymes or be produced through solid-state fermentation using fungi such as *Aspergillus awamori* or *Aspergillus oryzae*. Enzymatic hydrolysis has the benefits of mild reaction conditions, no use of hazardous chemicals and a low risk of producing substances which inhibits the fermentation. In parallel, seed cultivation or seed fermentation is performed to cultivate the succinic acid producing microorganisms needed in the fermentation process (Lam et al., 2014).

C.2 Host Organisms

There are several microorganisms such as bacteria and fungi which can be used for fermentive SA production. Some commonly used bacteria include
Actinobacillus succinogenes, *Mannheimia succiniciproducens*, *Anaerobiospirillum succiniciproducens* and several strains of metabolically engineered *Escherichia coli* (*E. Coli*) (van Heerden and Nicol, 2013). *Saccharomyces cerevisiae*, commonly called yeast, have also been used. Which host organism is used for the fermentation to a large extent determines the structure of the production process (Jansen and van Gulik, 2014).

E.Coli can produce SA through six different metabolical pathways. The main product from fermentation with E.Coli is ethanol, lactic, formic and acetic acid and the SA yield low (Song and Lee, 2006). Nevertheless, E.Coli has several advantages. Through metabolical engineering, cell growth can be inhibited and the production of

by-products can be suppressed. The genome of E.Coli is well-known and can therefore easily be genetically modified (Jansen and van Gulik, 2014). Less formation of by-products makes the purification steps easier (Sun et al., 2014). Another benefit with E. Coli is that it can produce SA under both aerobic and anaerobic conditions (Song and Lee, 2006). One drawback is that E.Coli requires neutral pH and titrants must therefore be added to maintain a neutral pH in the fermentation which results on a more complicated recovery process (Jansen and van Gulik, 2014).

C.3 Succinic Acid Fermentation

In the fermentation step, the sugar glucose (also called dextrose) released during the hydrolysis can be converted to succinic acid. The theoretical optimal reaction for transforming glucose to SA is:

$$7\ Glucose + 6\ CO_2 \rightarrow 12\ SA + 6\ H_2O$$

This reaction does not consider any biomass growth and gives the maximum theoretical yield of SA from glucose 1.12 g SA/g glucose (van Heerden and Nicol, 2013). The highest realized SA yield reported in literature is over 1 g SA per g of glucose, which is close to the maximum theoretical yield. This yield has been achieved using metabolically engineered E.Coli.

Succinic acid is a weak acid and has a pKa value of 4.16 and 5.61 (López-Garzón and Straathof, 2014). During the fermentation, the SA will dissociate and be present in the form of an ionized succinate salt instead of the carboxylic acid form (Song and Lee, 2006). The pH in the fermentation medium, called fermentation broth will thus decrease while SA is produced (López-Garzón and Straathof, 2014). A low pH can inhibit the microorganisms to produce more SA. To handle this, a microorganism which tolerates low pH can be used. The pH can also be controlled by adding chemicals to neutralize the carboxylic acid or the SA can be removed from (Song and Lee, 2006). SA can also be produced at a low pH. When the pH is two units below the pKa value, 99% of the carboxylic acid is in its undissociated form (López-Garzón and Straathof, 2014).

The yield of SA can be kept high by controlling the concentration of nutrients.
Microorganisms requires carbon and various types of nutrients to grow (Chen and Strevett, 2003). During the fermentation, cell growth might interfere with SA production (López-Garzón and Straathof, 2014). The production of biomass can however be controlled by limiting the amount of nutrients such as nitrogen or phosphorus. By providing an excess amount of carbon and a limited amount of nitrogen, cell growth will be reduced thorough nitrogen limitation and more of the desired product will be produced (Chen and Strevett, 2003).

C.4 Recovery of Succinic Acid

Once the SA has been produced in the fermentation it must be recovered and purified through upgrading, also called downstream processing. The downstream processing for succinic acid and other carboxylic acids usually starts with removal of residue cells and particles such as remaining sugars, salts and by-products. The product is then removed from the bulk aqueous solution (López-Garzón and Straathof, 2014). If required, the ionized succinate salt can be converted back to the carboxylic acid form through ion exchange by addition of H^+ (U.S. Department of Energy, 2004).

The carboxylic acid is further concentrated by removing impurities and solvents to the desired product (López-Garzón and Straathof, 2014). By removing water in combination with cooling, a super supersaturated solution if created which can crystallize. The highest reported yield in the crystallization step is only 73 % which mean around a fourth of the produced SA is lost in the crystallization. The process structure and economics of the recovery process depends of what acids and bases are used in the recovery process.

C.5 Benefits of the Reverdia DC process

The Reverdia DC process has several advantages. The process use metabolically engineered yeast to produce succinic acid by fermentation at pH 3 (Sheldon, 2014). One large benefit with using yeast is that it can produce SA at low pH and the pH does not need to be adjusted leading to a low requirement of acids and bases (Choi et al., 2015) (López-Garzón and Straathof, 2014). At pH 3, around 6 % of the SA on molar basis is present at mono-dissociated succinate (Jansen and van Gulik, 2014). Since most SA is present as succinic acid and not the succinate salt, the recovery process is easier in the form of less process operations, thus reducing investment costs and yield losses.

Appendix D – Calculations

Appendix D presents the calculations of the data collected in the Life Cycle Inventory.

D.1 Calculations Option 1: FW to biogas

All data for Option 1: FW to biogas is based on data from Ragn-Sells biogas facility in Heljestorp, measured during year 2015. The data was kindly provided by Ragn-Sells.

D.1.1 Division of electricity and process water

Around 80 % of the process water goes to the hygenisation process (Personal communication Ragn-Sells, 2016). Assume the remaining 20 % is divided equally between the processes Pre-treatment and Anaerobic digestion.

17 915 tonnes *0.8 = 14 332 tonnes is used in Hygenisation.
17 915 tonnes *0.1 = 1 791.5 tonnes is used in Pre-treatment respectively Anaerobic digestion.

Around 80 % of the electricity goes to the Pre-treatment process (Personal communication Ragn-Sells, 2016). Assume the remaining 20 % is divided equally between the processes Hygienisation and Anaerobic digestion.

1 762 723 kWh * 0.8 = 1 410 178.4 kWh is used in Pre-treatment.
1 762 723 kWh * 0.1 = 176 272.3 kWh is used in Hygenisation respectively Anaerobic digestion.

D.1.2 Calculations of CO_2 emissions from Torch and Boiler

CO_2-eq from combusted landfill gas

7.8 GWh landfill gas from Heljestorp landfill corresponds to 12 700 tonnes CO_2-eq. (Ragn-Sells, 2007).

Calculate amount of kg CO_2-eq / kWh landfill gas:

12 700 000 kg CO_2-eq / 7 800 000 kWh = 1.6282051 kg CO_2-eq / kWh landfill gas.
Landfill gas is around 50 % CH_4, 50 % CO_2. (Personal communication Ragn-Sells, 2016).

When combusted, the CH_4 share is converted to CO_2 according to the chemical reaction:

$CH_4 + 2\ O_2 \rightarrow CO_2 + 2\ H_2O$

Thus one CH_4 is converted to one CO_2 when combusted.
1 kg CH_4 equals 23 kg CO_2-eq.

When 1 kWh landfill gas is combusted, the emitted CO_2-eq becomes:

1.6282051 kg CO_2-eq / kWh landfill gas*(0.5 + 0.5/23) = 0.8494983 kg CO_2-eq.
Thus 1 kWh combusted landfill gas gives rise to 0.8494983 kg CO_2-eq.

CO_2-eq from combusted biogas

10.3 GWh of produced biogas from Heljestorp biogas facility corresponds to 16 900 tonnes CO_2-eq. (Ragn-Sells, 2007).

Calculate amount of kg CO_2-eq / kWh produced biogas:

16 900 000 kg CO_2-eq. / 10 300 000 kWh = 1.6407767 kg CO_2-eq / kWh biogas.

Biogas is around 60 % CH_4, 40 % CO_2. (Personal communication Ragn-Sells, 2016).

Calculating the same way as above, when 1 kWh biogas is combusted, the emitted CO_2-eq becomes:

1.6407767 kg CO_2-eq / kWh biogas * (0.4 + 0.6/23) = 0.6991136

Thus 1 kWh combusted biogas gives rise to 0.6991136kg CO_2-eq.

CO_2 Emissions from the boiler

During year 2015, the boiler used 2 487 095 kWh landfill gas and 2 613 349.6 kWh biogas.

The annual CO_2-eq emissions from the boiler becomes:

2 487 095 kWh landfill gas * 0.8494983kg CO_2-eq / kWh landfill gas + 2 613 349.6 kWh biogas * 0.6991136 kg CO_2-eq / kWh biogas = 3 939 811.16 kg CO_2-eq = 3 939.81116 tonnes CO_2-eq.

CO_2 Emissions from the torch

During year 2015, 356 168.9949 kWh biogas was combusted in the torch.

The annual CO_2-eq emissions from the torch becomes:

356 168.9949 kWh biogas *0.6991136kg CO_2-eq / kWh biogas = 249 002.57 kg CO_2-eq = 249.00257 tonnes CO_2-eq.

D.1.3 Calculations of CO_2 emissions from food waste

900 000 tonnes unavoidable household food waste gives rise to 1 860 000 tonnes CO_2-eq. (Bernesson and Strid, 2011).

1 tonne food waste (w.m.) thus gives rise to:

1 860 000 tonnes CO_2-eq /900 000 tonnes food waste = 2.0666667 tonnes CO_2-eq/tonne food waste

D.1.4 Allocation

Mass allocation was used in the process Pre-treatment. 68.5 % of the environmental impact was allocated to the flow “food waste slurry” and 31.5 % was allocated to the flow “Combustible waste”.

Mass allocation was also used in the process Anaerobic digestion. The environmental impact was allocated between the products “Biogas, sold to Trollhättan”, “Liquid biofertiliser” and “Solid biofertiliser”. 4.2 % of the impact was allocated to the biogas, 95.4 % to the liquid biofertiliser and 0.4 % to the solid biofertiliser.

D.1.5 Rescale to data to functional unit

The yearly amount food waste used for biogas production is 22 768.101 tonnes.

All data from Ragn-Sells is available in yearly amounts. To get input and output data per 1 tonne food waste, all data were divided by 22768.101 tonnes.

Examples of total solid content of food waste is 33.3 % Total solids (Bernstad Saraiva Schott et al., 2013) or 30.9 % Total solids (Zhang et al., 2007). Based on these data, the dry mass (d.m.) of food waste is assumed to be 30 %, i.e. 1 tonne of food waste (w.m.) corresponds to 0.3 tonnes of food waste (d.m.)

To get the life cycle inventory data provided from Ragn-Sells scaled per the functional unit 1 tonne of processed food waste, (d.m.), all data were also multiplied by 100/30.

This gives the data listed in tables C.1-C.5 in Appendix C.

D.2 Calculations Option 2: FW to SA

The data used for Option 2:FW to SA ismainly a combination of data from a lab scale study of succinic acid from mixed food waste using E.Coli by Sun et al. (2014) and an economic evaluation of pilot plant producing succinic acid from waste bread by Lam et al. (2014).

D.2.1 Yield correction

The yield must be adapted when using food waste in the pilot plant instead of bread.

The pilot plant process 312 tonnes bread (w.m.) per year (Lam et al., 2014). Assume the same amount of food waste is used as bread waste, i.e. 312 tonnes bread (w.m.) per year.

The yield of fermentative production of SA from mixed food waste is 0.224 kg SA/kg food waste, (w.m.) (Sun et al., 2014). The yield of SA in the Bacterial fermentation process is thus assumed to be 0.224 kg SA/kg food waste (w.m.).

Assume the yield of the recovery process in SA upgrading is 60 %. Thus 60 % of the produced mass of SA in the fermentation is obtained as SA crystals. The yield for the whole process becomes:

0.224 kg SA/kg food waste (w.m.) * 0.6 = 0.1344 kg SA/kg food waste (w.m.)

Assume dry mass of food waste is the same as for Option 1, i.e 30 %.
Recalculated to a basis of dry food waste instead, the yield for the whole process becomes:

0.1344 kg SA/kg food waste (w.m)* 100/30 = 0.448

Thus 0.448 kg SA is produced per kg food waste (d.m.).

D.2.2 Sensitivity analysis of upgrading yield

As a part of the sensitivity analysis, the yield of the recovery process in SA upgrading is assumed to be 30 % respectively 90 % instead of 60 %. Following the same calculations as above, the yield for the whole process becomes:

For 30 %: 0.224 kg SA/kg food waste (w.m.)* 0.3* 100/30 = 0.224 kg SA/kg food waste (d.m.)

For 90 %: 0.224 kg SA/kg food waste (w.m.)* 0.9* 100/30 = 0.672 kg SA/kg food waste (d.m.)

D.2.3 E.Coli propagation

Due to lack of data for cultivation for E.Coli cultivation, the E.Coli was modelled as yeast instead. To model the propagation of yeast (E.Coli) data from Jungbluth et al. (2007). Unit process fermentation of whey was used. In this process ethanol is produced and yeast paste and a protein concentrate is produced as a by-products.

In the bacterial fermentation process, 0.1 g E.Coli cells (d.m.) is assumed to be required to produce 1 kg SA. This is however an unsure value. As a part of the sensitivity analysis, this assumption is changed to 0.1 kg E.Coli bacteria per 1 kg SA instead.

The yield of SA from food waste in the Bacretial fermentation is 0.224 kg SA/kg FW (w.m.) (Sun et al., 2014). The required amount of E.Coli (yeast) per kg Food waste (w.m.) becomes:

0.224 kg SA/kg FW (w.m) * 0.1 g E.Coli (yeast)/kg SA = 0.0224 g E.Coli (yeast) / kg FW, w.m.

Thus 0.0224 kg E.Coli (yeast) is needed per tonne FW (w.m.) when 0.1 g E.Coli cells (d.m.) per 1 kg SA is assumed.

For the sensitivity analysis:

0.224 kg SA/kg FW (w.m) * 0.1 kg E.Coli (yeast)/kg SA = 0.0224 kg E.Coli (yeast) / kg FW, w.m.

Thus 22.4 kg E.Coli (yeast) is needed per tonne FW (w.m.) when 0.1 kg E.Coli cells (d.m.) per 1 kg SA is assumed

D.2.4 CO_2 input in bacterial fermentation

In the Bacterial fermentation process, 0.224 tonne SA is produced per tonne FW (w.m) (Sun et al., 2014).

Calculate the mass of CO_2 required per tonne of food waste (w.m.):

The molar mass of SA is 118.09 g/mole, and the molar mass of CO_2 is 44.01 g/mole.
224 kg SA /118.09 kg/kmole = 1.8969 kmoles SA

The theoretical optimal reaction for transforming glucose to SA is (van Heerden and Nicol, 2013):
$7\ Glucose + 6\ CO_2 \rightarrow 12\ SA + 6\ H_2O$

All supplied CO_2 is assumed to be converted to SA.

1.8969 kmoles SA *6/12 = 0.9484 kmoles CO_2

0.9484 kmoles CO_2 * 44.01 kg/kmole = 41.74 kg CO_2

Thus 224 kg SA requires 41.74 kg CO_2.

41.74 kg CO_2 is needed per tonne FW (w.m.).

D.2.5 DDGS input to supply nitrogen for SA production

Calculate the required amount of wet DDGS to supply E.Coli with the nitrogen required for E.Coli growth.

1 kg FW (w.m.) is assumed to be 30% d.m. Dry food waste contains 3.16 % N (Zhang et al., 2007).

1 kg FW (w.m.) * 0.30 * 0.0316 = 0.00948 kg N / kg FW (w.m.)

Thus 1 kg wet food waste contains 0.00948 kg N, 1 kg of dry FW contains 0.0316 kg N.

The dry weight of E.Coli is assumed to constitute of 14% nitrogen (Chen and Strevett, 2003).

1kg E.Coli (d.m) * 0.14 = 0.14 kg N / kg E.Coli (d.m.)

In this case, assume 1 kg E.Coli (d.m.) is needed to produce 1 kg of SA. This is a very high value and thereby this is a worst case scenario.

Thus 1 kg SA requires 1 kg E.Coli cells (d.m.) which requires 1 *0.14 = 0.14 kg N per kg SA

The SA yield in the bacterial fermentation process is 0.224 kg SA/kg FW (w.m.)

Recalculated to kg FW (d.m.):

0.224 kg SA/kg FW (w.m.) / 0.3 kg FW (d.m.)/kg FW (w.m.) = 0.7467 kg SA/kg FW (d.m.)

Assume E.Coli is propagated on nitrogen from DDGS during the bacterial fermentation. Assume all nitrogen in the DDGS is available for biomass production for E.Coli, although there likely is a yield coefficient which would be reasonable to

include. In that case, the required amount of DDGS to supply nitrogen for E.Coli growth is:
N supplied by DDGS = N needed for E.Coli to produce SA - N content in FW

0.14 kg N / kg E.Coli (d.m.)* 1 kg E.Coli (d.m.)/kg SA * 0.7467 kg SA/kg FW (d.m.) - 0.0316 kg N / kg FW (d.m.) = 0.07293 kg N from DDGS/ kg FW (d.m.)

Thus 0.07293 kg N/kg FW, d.m. needs to be supplied by DDGS.

N content in DDGS is 0.075 kg N/kg TS DDGS (Bernesson and Strid, 2011).
The DDGS product Agrodrank 90 is 90 % dry mass (Bernesson and Strid, 2011).

DDGS that needs to be added to supply E.Coli with N is:

0.07293 kg N/kg FW (d.m.) / (0.075 kg N/kg DDGS (d.m.) * 0.90 kg DDGS (d.m.)/kg DDGS (w.m.)) = 1.0805 kg DDGS (w.m.)/kg FW, d.m.

Thus 1,0805 tonnes wet Agrodrank 90 needs to be supplied per tonne dry FW, to supply E.Coli. with nitrogen.

D.2.6 Calculations of impact from DDGS production process

1 ha gives 1892 kg DDGS and 1748 kg etanol (Bernesson and Strid, 2011).

For 1.0805 tonne DDGS requirement, the area needed is 1.0805 *10 000 / 1892 = 5.711 m2

The amount of ethanol co-produced from 5.711 m2 is 1.0805 * 1748/1892 = 0.998 tonne ethanol

Data for emissions from ethanol production incl. wheat production in g/ha can be found in (Bernesson and Strid, 2011). Recalculate emission data in g/ha to kg/tonne FW (d.m.):

If 1.0805 kg wet DDGS is needed per kg FW (d.m.), and 5.711 m2 is needed to produce 1.0805 kg DDGS, then X g emissions/ha*5.711 (m2/kg FW (d.m.)) /10 000 (m2/ha) = kg X emissions / ton FW (d.m.). Using this calculation approach, the emissions from ethanol production becomes:

Ethanol production (including wheat production)		
Outputs:	g/ha	kg /tonne FW (d.m.)
DDGS	n/a	529.4734
Ethanol	n/a	1.143513
CO_2	927 135	0.362891
CO	2 002.35	0.350709
HC	635.44	3.914637
CH_4	614.11	0.888877
NOx	6 854.73	0.912041
SOx	1 556.47	2.291138
NH_3	1 597.03	0.054139
N_2O	4 011.9	2.28 E-05
HCl	94.8	0.180263
PAH	0.04	5.48667
Particles	315.65	529.4734
PO_4^{3-}	9 607.44	1.143513

D.2.7 Division of electricity, steam and process water

Electricity

Electricity used in the process during one year is 25 631 kWh (Lam et al., 2014). It is unknown where in the process. Assume the steam is used in Pre-treatment, Bacterial fermentation and SA upgrading, divide into 3 equal shares:

25 631 kWh / 3 = 8543.67 kWh in each process.

Steam

Steam used in the process during one year: 1029 metric tonnes (Lam et al., 2014). It is unknown where in the process. Assume the steam is used in Pre-treatment, Bacterial fermentation and SA upgrading, divide into 3 equal shares:

1029 / 3 = 343 metric tonnes in each process.

Process water

Process water used in the process during one year: 2974 metric tonnes (Lam et al., 2014). It is unknown where in the process. Assume water is used in the Pre-treatment, same mass as incoming food waste, i.e. 312 metric tonnes per year. Assume remaining share of process water is used in SA upgrading:

2974 – 312 = 2662 tonnes

D.2.8 Allocation

Mass allocation was used in the process Ethanol production. 48.0 % of the environmental impact was allocated to the flow “Ethanol” and 52.0 % was allocated to the flow “DDGS”.

Mass allocation was used in the process E.Coli propagation. 12.4 % of the environmental impact was allocated to the flow “ethanol”, 73.4 % to the flow “protein concentrate” and 14.2 % to the flow “yeast paste” which is assumed to be propagated E.Coli.

Both mass allocation and economic allocation was used for the process SA upgrading. The environmental impact was allocated between the products “Biomass” and “SA crystals”.

When using mass allocation, 13.4 % of the environmental impact was allocated to the flow “SA crystals” and 86.6 % was allocated to the flow “biomass”.

For the economic allocation, the following data from Lam et al. (2014) was used:
Economic value SA crystals: 9 US$ / kg
Economic value Biomass: 0.45 US$ / kg

When using economic allocation, 75.6 % of the environmental impact was allocated to the flow “SA crystals” and 24.4 % was allocated to the flow “biomass”.

D.2.9 Rescaling data between functional units

Option 2: FW to SA is evaluated for two different functional units: 1 tonne of processed food waste (d.m.) and 1 tonne of produced succinic acid crystals. All data in this calculation appendix is calculated for the functional unit 1 tonne food waste (d.m.) Thus all data must be rescaled to 1 tonne of produced succinic acid crystals.

How to rescale the data to 1 tonne of produced SA crystals depends on if yield of the recovery process in SA upgrading is assumed to be 30%, 60% or 90%.

Based on previous calculations:
For 30 %: 0.224 kg SA is produced per kg food waste (d.m).
For 60 %: 0.448 kg SA is produced per kg food waste (d.m.).
For 90 %: 0.672 kg SA is produced per kg food waste (d.m.).

To rescale the LCI data from 1 tonne FW (d.m.) to 1 tonne SA crystals, all data must be divided by either 0.224, 0.448 or 0.672 respectively.

Appendix E – Data tables of modelled processes

Appendix E presents the processes modelled in openLCA.

E.1 Option 1: FW to biogas

Table E.1: Modelled data for the process Pre-treatment.

Pre-treatment (1.1)					
Inputs	**Data reference**	**Used flow in openLCA**	**Data for 1 tonne of processed food waste (dry mass)**	**Unit**	**Comment**
Total waste input	(Ragn-Sells, 2015)	Total waste input (dummy flow)	4.87	tonne	
Electricity	(Ragn-Sells, 2015) (Personal communication Ragn-Sells, 2016)	electricity, medium voltage - SE	206.46	kWh	
Water	(Ragn-Sells, 2015)	tap water - Europe without Swizerland	0.26	tonne	
Outputs					
Food waste slurry	(Ragn-Sells, 2015)	Food waste slurry (dummy flow)	3.33	tonne	
Combustible waste	(Ragn-Sells, 2015)	Combustible waste (dummy flow)	1.53	tonne	
CO_2-eq from food production	(SIK, 2008)	carbon dioxide, fossil - air, unspecified	6.89	tonne	Include this flow in the sensitivity analysis

Table E.2: Modelled data for the process Hygenisation.

Hygenisation (1.2)					
Inputs	**Data reference**	**Used flow in openLCA**	**Data for 1 tonne of processed food waste (dry mass)**	**Unit**	**Comment**
Food waste slurry	(Ragn-Sells, 2015)	Food waste slurry (dummy flow)	3.33	tonne	
Steam from boiler	(Personal communication Ragn-Sells, 2016)	Steam from boiler (dummy flow)	1.0	tonne	Set to 1 since no data available on amount of steam
Electricity	(Ragn-Sells, 2015)	electricity, medium voltage - SE	25.81	kWh	
Water	(Ragn-Sells, 2015) (Personal communication Ragn-Sells, 2016)	tap water - Europe without Swizerland	2.10	tonne	
Recirculated process water	(Ragn-Sells, 2015)	Recirculated process water (dummy flow)	0.72	m3	
Outputs					
Digestate	n/a	Digestate (dummy flow)	6.43	tonne	Assume to be mass of food waste slurry and tap water. Just a flow to link processes in the model.

Table E.3: Modelled data for the process Anaerobic digestion.

Anaerobic digestion (1.3)					
Inputs	**Data reference**	**Used flow in openLCA**	**Data for 1 tonne of processed food waste (dry mass)**	**Unit**	**Comment**
Digestate	n/a	Digestate (dummy flow)	6.43	tonne	Assume to be mass of food waste slurry and tap water. Just a flow to link processes in the model.
Electricity	(Ragn-Sells, 2015)	electricity, medium voltage - SE	25.81	kWh	
Water	(Ragn-Sells, 2015)	tap water - Europe without Swizerland	0.26	tonne	
Outputs					
Biogas (sold to Trollhättan)	(Ragn-Sells, 2015)	Biogas (sold to Trollhättan) (dummy flow)	1194.59	kWh	
Biogas (burned in torch)	(Ragn-Sells, 2015)	Biogas (burned in torch) (dummy flow)	52.14	kWh	
Biogas (burned in boiler)	(Ragn-Sells, 2015)	Biogas (burned in boiler) (dummy low)	382.60	kWh	
Recirculated process water	(Ragn-Sells, 2015)	Recirculated process water (dummy flow)	0.72	m3	
Liquid biofertilizer	(Ragn-Sells, 2015)	Liquid biofertilizer (dummy flow)	3.24	tonne	
Solid biofertilizer	(Ragn-Sells, 2015)	Solid biofertilizer (dummy flow)	0.01	tonne	
Solid residue	(Ragn-Sells, 2015)	Solid residue (dummy flow)	0.13	tonne	

Table E.4: Modelled data for the process Steam production in boiler.

Steam production in boiler (1.4)					
Inputs	**Data reference**	**Used flow in openLCA**	**Data for 1 tonne of processed food waste (dry mass)**	**Unit**	**Comment**
Landfill gas	(Ragn-Sells, 2015)	Landfill gas (dummy flow)	364.12	kWh	
Oil	(Ragn-Sells, 2015)	heavy fuel oil, burned in refinery furnace - Europe without Swizerland	1.67	kWh	
Biogas (burned in boiler)	(Ragn-Sells, 2015)	Biogas (burned in boiler) (dummy low)	382.60	kWh	
Outputs					
Steam from boiler	(Personal communication Ragn-Sells, 2016)	Steam from boiler (dummy flow)	1.0	tonne	Set to 1 since no data available on amount of steam
Carbon dioxide	(Ragn-Sells, 2015) (Ragn-Sells, 2007)	carbon dioxide - air, unspecified	0.58	tonne	Use this for the base case modelling
Carbon dioxide	(Ragn-Sells, 2015) (Ragn-Sells, 2007)	carbon dioxide, fossil - air, unspecified	0.58	tonne	Use this for the sensitivity analysis

Table E.5: Modelled data for the process Combustion in torch.

Combustion in torch (1.5)					
Inputs	**Data reference**	**Used flow in openLCA**	**Data for 1 tonne of processed food waste (dry mass)**	**Unit**	**Comment**
Biogas (burned in torch)	(Ragn-Sells, 2015)	Biogas (burned in torch) (dummy flow)	52.14	kWh	
Outputs					
FU dummy flow	n/a	FU dummy flow: 1 tonne mixed food waste, dry mass (dummy flow)	1.00	tonne	Assumption, flow to be able to calculate product system per the Functional unit 1 tonne of mixed food waste, dry mass
Carbon dioxide	(Ragn-Sells, 2015) (Ragn-Sells, 2007)	carbon dioxide, air, unspecified	0.04	tonne	Use this for the base case modelling
Carbon dioxide	(Ragn-Sells, 2015) (Ragn-Sells, 2007)	carbon dioxide, air, unspecified	0.04	tonne	Use this for the sensitivity analysis

E.2 Option 2: FW to SA

Table E.6: Modelled data for the process Protease production.

Protease production					
Inputs	**Data reference**	**Used flow in openLCA**	**Data for 1 tonne of processed food waste (dry mass)**	**Unit**	**Comment**
Use of agricultural land	(Nielsen et al., 2006)	unknown land use - GLO	0.28	m2	
Primary energy consupmtion	(Nielsen et al., 2006)	electricity, medium voltage - SE	55.128	MJ	
Outputs					
Protease	(Nielsen et al., 2006)	Protease (dummy flow)	0.919	kg	
Global warming potential	(Nielsen et al., 2006)	Carbon dioxide - air, unspecified	3.675	kg	
Acidification potential	(Nielsen et al., 2006)	Sulfur dioxide - air, unspecified	13.78	g	
Nutrient enrichment	(Nielsen et al., 2006)	Phosphorus pentaoxide - air, unspecified	0.919	g	
Photochemical ozone formation	(Nielsen et al., 2006)	Ethylene oxide - air, unspecified	1.011	g	

Table E.7: Modelled data for the process Glucoamylase production.

Glucoamylase production					
Inputs	**Data reference**	**Used flow in openLCA**	**Data for 1 tonne of processed food waste (dry mass)**	**Unit**	**Comment**
Use of agricultural land	(Nielsen et al., 2006)	unknown land use - GLO	3.216	m2	
Primary energy consupmtion	(Nielsen et al., 2006)	electricity, medium voltage - SE	82.692	MJ	
Outputs					
Glucoamylase	(Nielsen et al., 2006)	Glucoamylase (dummy flow)	0.919	kg	
Global warming potential	(Nielsen et al., 2006)	Carbon dioxide - air, unspecified	6.891	kg	
Acidification potential	(Nielsen et al., 2006)	Sulfur dioxide, air, unspecified	22.051	g	
Nutrient enrichment	(Nielsen et al., 2006)	Phosphorus pentaoxide - air, unspecified	20.214	g	
Photochemical ozone formation	(Nielsen et al., 2006)	Ethylene oxide - air, unspecified	2.297	g	

Table E.8: Modelled data for the process E.Coli propagation.

E.Coli propagation (2.2)					
Inputs	**Data reference**	**Used flow in openLCA**	**Data for 1 tonne of processed food waste (dry mass)**	**Unit**	**Comment**
Whey, at dairy	(Jungbluth et al., 2007)	whey – GLO	1117.653	kg	
Water, cooling, unspecified natural origin	(Jungbluth et al., 2007)	Water, cooling, unspecified natural origin - resource, in water	0.028	m3	
tap water, at user	(Jungbluth et al., 2007)	tap water - Europe without Switzerland	131.385	kg	
sodium sulphate, from natural sorces, at plant	(Jungbluth et al., 2007)	sodium sulphate, various forms, in ground - Resource, in ground	0.043	kg	
sodium phosphate, at plant	(Jungbluth et al., 2007)	sodium phosphate – RER	0.114	kg	
soda, powder, at plant	(Jungbluth et al., 2007)	soda ash, light, crystalline, heptahydrate – RER	0.895	kg	
sulphuric acid, liquid, at plant	(Jungbluth et al., 2007)	sulfuric acid – RER	0.496	kg	
transport, lorry 28t	(Jungbluth et al., 2007)	transport, freight, lorry 16-32 metric ton, EURO3 - RER	0.056	tkm	
transport, lorry 16t	(Jungbluth et al., 2007)	transport, freight, lorry 16-32 metric ton, EURO3 - RER	111.765	tkm	
transport, freight, rail	(Jungbluth et al., 2007)	transport, freight train - Europe without Switzerland	0.670	tkm	
electricity, medium voltage, at grid	(Jungbluth et al., 2007)	electricity, medium voltage - RER	8.163	kWh	

heat, at cogen with gas engine, allocation exergy	(Jungbluth et al., 2007)	heat, district or industrial, natural gas - Europe without Switzerland	75.415	MJ	
heat, natural gas, at industrial furnace >100kW	(Jungbluth et al., 2007)	heat, district or industrial, natural gas - Europe without Switzerland	138.289	MJ	
Outputs					
Ethanol, 95% in H2O, from whey, at fermentation plant	(Jungbluth et al., 2007)	ethanol, without water, in 95% solution state, from fermentation – RER	19.58	kg	
Protein concentrate, from whey, at fermentation	(Jungbluth et al., 2007)	protein concentrate, from whey, at fermentation (dummy flow)	116.054	kg	
Yeast paste, from whey, at fermentation	(Jungbluth et al., 2007)	Yeast paste (dummy flow)	22.4	kg	Assume yeast paste is propagated E.Coli
Heat, waste	(Jungbluth et al., 2007)	heat, waste - air, high population density	29.388	MJ	
Carbon dioxide, biogenic	(Jungbluth et al., 2007)	Carbon dioxide, biogenic - air, high population density	56.259	kg	
BOD5, biological oxygen demand	(Jungbluth et al., 2007)	BOD5, Biological Oxygen Demand - water, unspecified	0.881	kg	
COD, Chemical oxygen demand	(Jungbluth et al., 2007)	COD, Chemical Oxygen Demand - water, unspecified	0.881	kg	
DOC, Dissolved organic carbon	(Jungbluth et al., 2007)	DOC, Dissolved Organic Carbon - water, unspecified	0.352	kg	
TOC, total organic carbon	(Jungbluth et al., 2007)	TOC, Total Organic Carbon - water, unspecified	0.352	kg	

Table E.9: Modelled data for the process Ethanol production.

Ethanol production					
Inputs	**Data reference**	**Used flow in openLCA**	**Data for 1 tonne of processed food waste (dry mass)**	**Unit**	**Comment**
n/a	n/a	n/a	n/a	n/a	
Outputs					
Drank	(Bernesson and Strid, 2011)	DDGS (dummy flow)	1.08	tonne	
Ethanol	(Bernesson and Strid, 2011)	Ethanol (dummy flow)	0.998	tonne	
CO_2	(Bernesson and Strid, 2011)	carbon dioxide - air, unspecified	529.473	kg	
CO	(Bernesson and Strid, 2011)	carbon monoxide - air, unspecified	1.144	kg	
HC	(Bernesson and Strid, 2011)	NMVOC, non-methane volatile organic compounds, unspecified origin - air, unspecified	0.363	kg	
CH_4	(Bernesson and Strid, 2011)	methane - air, unspecified	0.351	kg	
NOx	(Bernesson and Strid, 2011)	nitrogen oxides - air, unspecified	3.915	kg	
SOx	(Bernesson and Strid, 2011)	sulfur oxides - air, unspecified	0.889	kg	
NH_3	(Bernesson and Strid, 2011)	ammonia - air, unspecified	0.912	kg	
N_2O	(Bernesson and Strid, 2011)	Dinitrogen monoxide - air, unspecified	2.291	kg	
HCl	(Bernesson and Strid, 2011)	Hydrogen chloride - air, unspecified	0.054	kg	
Particles	(Bernesson and Strid,	Particulates, unspecified - air, unspecified	0.180	kg	

	2011)				
Phosphate PO_4^{3-}	(Bernesson and Strid, 2011)	Phosphate - water, ground water	5.487	kg	

Table E.10: Modelled data for the process Pre-treatment.

Pre-treatment (2.1)					
Inputs	**Data reference**	**Used flow in openLCA**	**Data for 1 tonne of processed food waste (dry mass)**	**Unit**	**Comment**
Mixed food waste	(Lam et al., 2014)	Mixed food waste (dummy flow)	3.333	tonne	
Water	(Lam et al., 2014)	Water, process, unspecified natural origin - resource, in water	3.333	m3	
Glucoamylase	(Lam et al., 2014)	Glucoamylase (dummy flow)	0.919	kg	
Protease	(Lam et al., 2014)	Protease (dummy flow)	0.919	kg	
Electricity	(Lam et al., 2014)	electricity, medium voltage - SE	91.278	kWh	
Steam	(Lam et al., 2014)	Steam, in chemical indusrty - RER	3.665	tonne	
Outputs					
Food waste slurry	n/a	Food waste slurry (dummy flow)	6.667	tonne	
CO_2-eq from food production	(SIK, 2008)	carbon dioxide, fossil - air, unspecified	6.889	tonne	Include in sensitivity analysis

Table E.11: Modelled data for the process bacterial fermentation.

Bacterial fermentation (2.3)					
Inputs	**Data reference**	**Used flow in openLCA**	**Data for 1 tonne of processed food waste (dry mass)**	**Unit**	**Comment**
Food waste slurry	n/a	Food waste slurry (dummy flow)	6.667	tonne	
Propagated E.Coli	(Lam et al., 2014)	Yeast paste (dummy flow)	0.075	kg	Assume yeast paste is propagated E.Coli
$MgCO_3$	(Lam et al., 2014)	potassium carbonate - RER	74.167	kg	
NaOH	(Lam et al., 2014)	sodium hydroxide, without water, in 50 % solution state - RER	0.053	tonne	
CO_2	(Lam et al., 2014)	carbon dioxide, liquid - RER	139.135	kg	Use for base case
CO_2		carbon dioxide, biogenic - air, unspecified	139.135	kg	Use in sensitivity analysis
CO_2		carbon dioxide, fossil - air unspecified	139.135	kg	Use in sensitivity analysis
DDGS	n/a	DDGS (dummy flow)	1.08	tonne	
Electricity	(Lam et al., 2014)	electricity, medium voltage - SE	91.278	kWh	
Steam	(Lam et al., 2014)	Steam, in chemical indusrty - RER	3.665	tonne	
Outputs					
SA in broth	n/a	SA share in broth (dummy flow)	0.747	tonne	
SA broth	n/a	SA broth (dummy flow)	10.932	tonne	

Table E.12: Modelled data for the process SA upgrading.

SA upgrading (2.4)					
Inputs	**Data reference**	**Used flow in openLCA**	**Data for 1 tonne of processed food waste (dry mass)**	**Unit**	**Comment**
SA in broth	n/a	SA share in broth (dummy flow)	0.747	tonne	
SA broth - change to elementary flow?	(Lam et al., 2014)	SA broth (dummy flow)	10.932	tonne	
Water	(Lam et al., 2014)	Water, process, unspecified natural origin - resource, in water	28.44	m3	
Electricity	(Lam et al., 2014)	electricity, medium voltage - SE	91.278	kWh	
HCl	(Lam et al., 2014)	hydrochloric acid, without water, in 30% solution state - RER	0.021	tonne	
NaCl brine	(Lam et al., 2014)	sodium chloride, brine solution . RER	4.071	tonne	
Steam	(Lam et al., 2014)	Steam, in chemical indusrty - RER	3.665	tonne	
Outputs					
SA crystals	(Lam et al., 2014)	SA crystals (dummy flow)	0.448	tonne	Use for base case
	(Lam et al., 2014)	SA crystals (dummy flow)	0.224	tonne	Use in sensitivity analysis
	(Lam et al., 2014)	SA crystals (dummy flow)	0.672	tonne	Use in sensitivity analysis
Biomass	(Lam et al., 2014)	Biomass (dummy flow)	2.886	tonne	

Appendix F – Model graphs from openLCA

Appendix F includes screenshots of the product system model graphs of Option 1: FW to biogas and Option 2: FW to SA from openLCA.

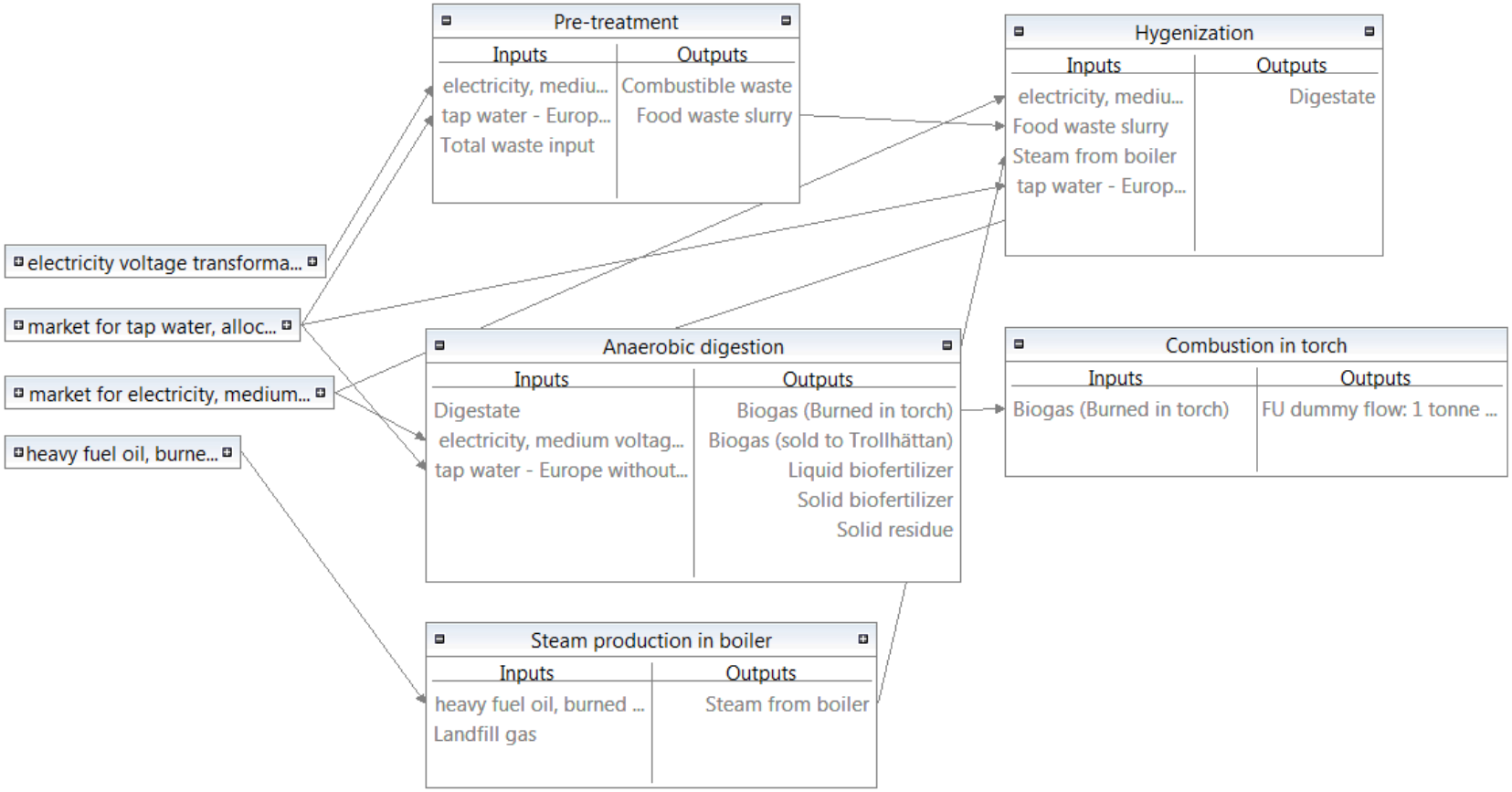

Figure F.1: Screenshot of product system model graph of the openLCA model for Option 1: FW to biogas.

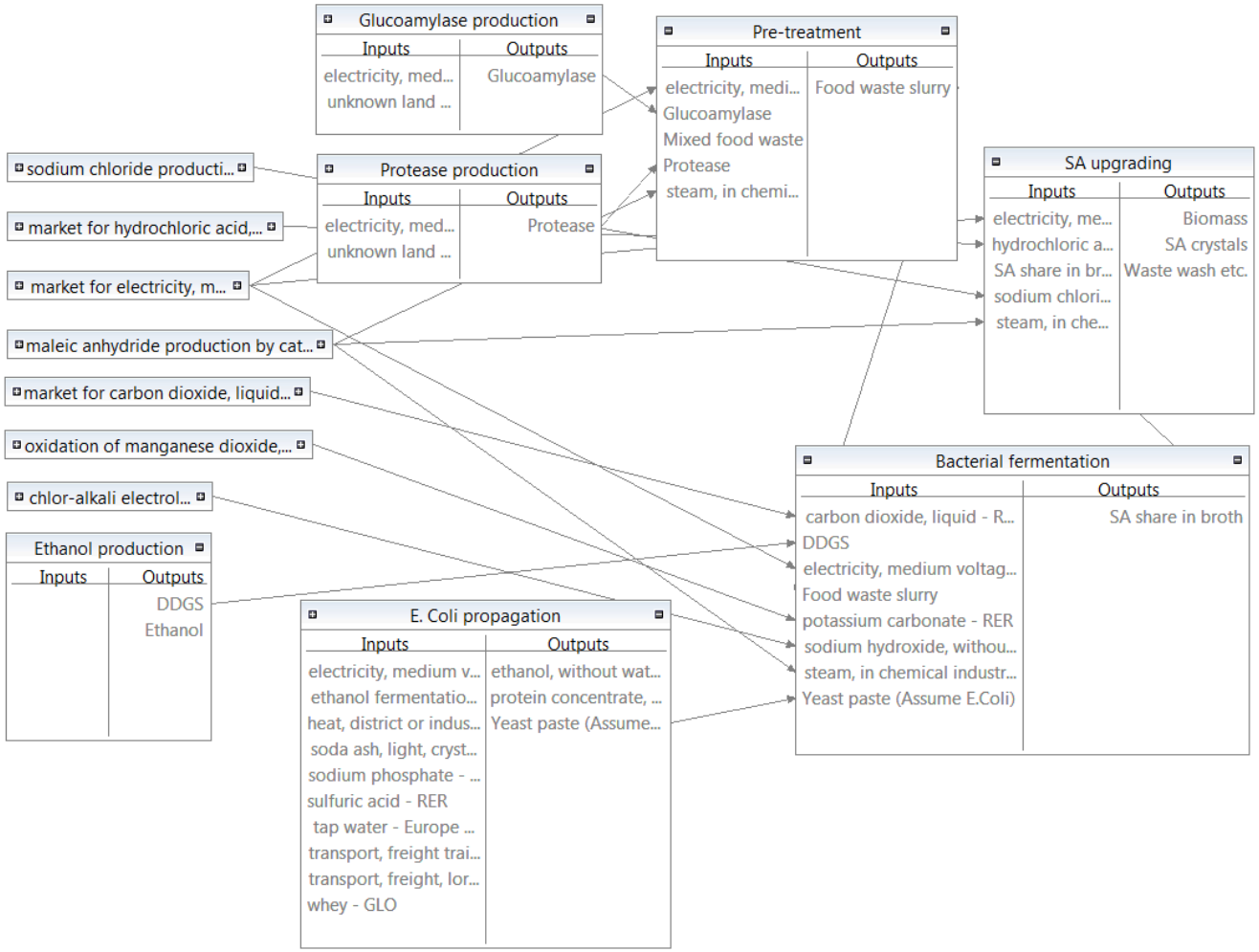

Figure F.2: Screenshot of product system model graph of the openLCA model for Option 2: FW to SA.

Appendix G – Used and Created Processes

Appendix G lists the processes used in the modelling in Open LCA. Both the used processes from the database Ecoinvent 3.1, attributional version and created processes based on other data sources are presented.

G.1 Option 1: FW to biogas

Table G.1: Used processes from Ecoinvent 3.1 for the modelling of Option 1: FW to biogas.

Used flow in openLCA	**Process from Ecoinvent 3.1**	**Data reference (as stated in Ecoinvent 3.1)**
electricity, medium voltage - SE	market for electricity, medium voltage, alloc. default, U	Itten R. et al. 2012
electricity, medium voltage - SE	electricity voltage transformation from high to medium voltage, alloc. default, U	Itten R. et al. 2012
tap water - Europe without Swizerland	market for tap water, alloc. default, U	n/a
heavy fuel oil, burned in refinery furnace - Europe without Swizerland	heavy fuel oil, burned in refinery furnace, alloc. default, U	Jungbluth, N. 2007

Table G.2: Created processes for the modelling of Option 1: FW to biogas.

Created process	**Data reference**
Pre-treatment (1.1)	(Ragn-Sells, 2015) (Personal communication Ragn-Sells, 2016) (SIK, 2008)
Hygenisation (1.2)	(Ragn-Sells, 2015)
Anaerobic digestion (1.3)	(Ragn-Sells, 2015)
Steam production in boiler (1.4)	(Ragn-Sells, 2015) (Ragn-Sells, 2007)
Combustion in torch (1.5)	(Ragn-Sells, 2015) (Ragn-Sells, 2007)

G.2 Option 2: FW to SA

Table G.3: Used processes from Ecoinvent 3.1 for the modelling of Option 2: FW to SA.

Used flow in openLCA	Process from Ecoinvent 3.1	Data reference (as stated in Ecoinvent 3.1)
unknown land use - GLO	market for unknown land use, alloc. default, U	n/a
electricity, medium voltage - SE	market for electricity, medium voltage, alloc. default, U	Itten R. et al. 2012
whey - GLO	cheese production, soft, from cow milk, alloc. default, U	Kim, D. et al. 2013
tap water - Europe without Switzerland	market for tap water, alloc. default, U	n/a
sodium phosphate - RER	sodium phosphate production, alloc. default, U	Zah R. et al. 2007
soda ash, light, crystalline, heptahydrate - RER	soda production, solvay process, alloc. default, U	Althaus H.-J. et al. 2007
sulfuric acid - RER	sulfuric acid production, alloc. default, U	Althaus H.-J. et al. 2007
transport, freight, lorry 16-32 metric ton, EURO3 - RER	transport, freight, lorry 16-32 metric ton, EURO3, alloc. default, U	Keller, M. 2010
transport, freight train - Europe without Switzerland	market for transport, freight train, alloc. default, U	n/a
electricity, medium voltage - RER	fluting medium production, semichemical, alloc. default, U	FEFCO et al. 2009
heat, district or industrial, natural gas - Europe without Switzerland	market for heat, district or industrial, natural gas, alloc. default, U	n/a
Steam, in chemical indusrty - RER	maleic anhydride production by catalytic oxidation of benzene, alloc. default, U	Althaus H.-J. et al. 2007
potassium carbonate - RER	oxidation of manganese dioxide, alloc. default, U	Althaus H.-J. et al. 2007
sodium hydroxide, without water, in 50 % solution state - RER	chlor-alkali electrolysis, mercury cell, alloc. default, U	Althaus H.-J. et al. 2007
carbon dioxide, liquid - RER	market for carbon dioxide, liquid, alloc. default, U	n/a
hydrochloric acid, without water, in 30% solution state - RER	market for hydrochloric acid, without water, in 30% solution state, alloc. default, U	Althaus H.-J. et al. 2007
sodium chloride, brine solution . RER	sodium chloride production, brine solution, alloc. default, U	Althaus H.-J. et al. 2007

Table G.4: Created processes for the modelling of Option 2: FW to SA.

Created process	**Data reference**
Protease production	(Nielsen et al., 2006)
Glucoamylase production	(Nielsen et al., 2006)
E.Coli propagation (2.2)	(Jungbluth et al., 2007)
Ethanol production	(Bernesson and Strid, 2011)
Pre-treatment (2.1)	(Lam et al., 2014) (SIK, 2008)
Bacterial fermentation (2.3)	(Lam et al., 2014)
SA upgrading (2.4)	(Lam et al., 2014)