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Environmental analysis of alternative food waste solutions in the urban waste system of BIR

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MASTER THESIS

for

Kaja Daviknes Sørgard

Spring 2018

Environmental analysis of alternative food waste solutions in the urban waste system of BIR

Miljøanalyse av ulike matavfallsløsninger i avfallssystemet hos BIR

Background and objective

The waste sector is considered a cornerstone in the transition towards a circular economy. The role of the sector is shifting from solely handling the waste in an environmental and safe way, to encompass the preparation of resources being re-introduced to the manufacturers and the economy. In accordance with circular economy thinking, the sector holds a pool of secondary material and energy resources that can be recovered and made available for new products and value creation.

The waste generation in Norway is steadily increasing, due to economic growth, a strong purchasing power in the population and shorter life cycles of products. The average Norwegian generates some 433 kg of household waste per capita every year. One of the largest waste fractions from the households is wet organic waste, and it holds a substantial recycling and recovery potential. The fraction is regarded as a particularly important fraction to remove from the residual waste by the Norwegian authorities, as it can release potential to sort out other fractions in the residual waste, in addition to the fraction itself being an important part of the national biogas strategy.

The objective of this master thesis is to examine alternative solutions for sorting of food waste in the households in the urban waste system of BIR (Bergenområdets interkommunale renovasjonsselskap). The purpose is to determine the critical factors of the environmental performance of alternative system solutions for the organic food waste. Different collection and end-treatment alternatives are assessed with the use of scenarios developed. This is done by analysing the waste management system's mass flows, energy use, energy efficiency, greenhouse gas emissions and costs. The master thesis is a continuation of work started in the autumn of 2017.

In 2016 a model was developed by students at NTNU, as a tool to quantitatively describe the current state and selected future scenarios in a waste system with respect to environmental performance. The model was developed to cover a gap in the knowledge concerning critical factors for the environmental efficiency of processes in a waste system. The model is a tool used to measure and explore what factors and variables that influence the environmental efficiency, or more specifically, the performance of given value chains within a waste system towards 2030 and in view of targets embedded in the circular economy policy package. For this, a system definition in combination with appropriate performance indicators are defined for the waste system studied.

Based on Industrial ecology methods, the model uses material flow analysis (MFA), energy analysis (EA) and life cycle assessment (LCA) to identify critical system variables and factors for system performance. In addition, life cycle cost (LCC) methods are applied to identify the costs of the alternative solutions. Data from the urban waste management company is used to analyse the current situation, while several desired future scenarios are developed and analysed in cooperation with the collaborating waste management company.

The starting point of the analysis carried out the spring of 2018 will be a model representation of BIR Privat AS's system in 2017, and a set of defined solutions in order to try to comply with targets for future increased material recovery from waste towards 2030. The purpose is to study what the environmental performance are for some alternative future scenarios that can be implemented by BIR Privat AS, compared with the current system.

The work will be carried out in collaboration with BIR Privat AS, with Barbro Relling as co-supervisor.

The following tasks are to be considered:

1. Carry out a literature study on topics of relevance to this project, with a focus on solutions for food waste and energy use, energy efficiency and GHG emissions in urban waste systems.
2. Collect the information needed to describe the recent and current management of selected waste categories for the case study, as well as possible new solutions in line with BIR Privat AS's plans for how to comply with (circular economy or other policy motivated) targets for future material recovery towards 2030. Collect the data needed to model and analyse the system performance with respect to energy use, energy efficiency, GHG emissions and costs for these situations from BIR Privat AS and other relevant sources.
3. Study critically the MFA-based model developed by students in 2016. Consider the system boundary and resolution of processes for the given urban household waste flows, including a mass flow layer, an energy layer, a GHG emission layer and a cost layer. Consider adding technologies in line with BIR Privat AS's targets. Define the criteria and indicators appropriate to determine the system and possible sub-system efficiencies for materials, energy, emissions and costs.
4. Use the model, with its constituent processes and flows, to analyse the current situation and selected scenarios for future management of the given waste flows towards 2030. Assess and compare the system performance for each scenario and examine critical system variables and factors that highly influence relevant performance levels.
5. Discuss the main findings of your work; i.e. levels of performance for different waste categories, influencing variables and factors, the effect of possible new solutions, and agreement with literature. Discuss the strengths and weaknesses of your work and the methods you applied. Finally, suggest recommendations for future work.

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

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

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

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

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

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

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

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

Department of Energy and Process Engineering, 15. January 2018

Professor Helge Brattebø
Academic Supervisor

Research Advisors: Assoc. Professor Sigrun Jahren, NTNU
Barbro Relling, BIR Privat AS

Preface

This thesis concludes my master's degree in Energy and Environmental Engineering at The Norwegian University of Science and Technology (NTNU). The thesis is an environmental analysis of the urban municipal waste management system in Bergen, Norway. The analysis is performed with an MFA-based model developed in 2016/2017 by students at NTNU. This particular case study was initiated in the fall of 2017. The thesis project has been conducted in cooperation with supervisor Professor Helge Brattebø at NTNU, co-supervisor Sigrun Jahren (NTNU) and Barbro Relling in BIR Privat AS.

In general, I would like to thank BIR Privat AS for their interest in my project and willingness to participate in this project. A particular thanks to Barbro Relling for being my patient co-supervisor. Barbro has, together with head of research and development in BIR AS, Thoralf Igesund, answered all questions regarding BIR's general interests and strategy. Thank you Kirsten Grevskott for all your detailed knowledge and providing of data about BIR Privat AS's operation. I would also like to Alexander Ringstad Solheim and Invertapro for sharing knowledge and providing insight in their technology development in regards to insect rearing. Thanks to co-supervisor Sigrun Jahren for a thorough review. Finally, thanks to my supervisor Professor Helge Brattebø, for constructive criticism, thorough guiding where needed and evident, useful feedback.

Trondheim, June 10, 2018

Abstract

The waste sector is appointed a key role in the transition towards a circular economy. The objective of the sector is shifting from solely handling the waste in an environmental and safe way, to encompass the preparation of resources being reintroduced in the economy. The waste sector is an important facilitator to achieve an increased recycling rate and circulation of resources and materials.

The objective of this thesis is to provide a holistic assessment of the effects of introducing food waste sorting in an urban waste management system. The main activity has been related to collection and processing of data used to analyze the urban waste management system of BIR (Bergen interkommunale renovasjonsselskap). The data is assessed with the use of a model developed and finalized by student Pieter G. Callewaert in 2017, under the supervision of Professor Helge Brattebø. The model is based on material flow analysis (MFA) methodology, with waste flow data from BIR as the input foundation. Furthermore, the model has an energy layer that requires case specific energy data for transport, incineration and other process treatment of the waste. In addition, an emission layer and a cost layer is computed manually based on the material and energy results computed by the model in addition to external, generic data.

Three system solutions for food waste treatment are modelled. The assessed alternatives are biogas production in Bergen, biogas production in Oslo and insect rearing in Voss (Hordaland). The performance in material recycling, energy recovery, emissions and cost-efficiency are compared and assessed.

The holistic assessment provides a range of results for all scenarios. The common features are an increase in material recycling and a decrease in the net emissions at the costs of a lower energy recovery rate (ERR) and higher expenses. The findings prove a rise in the overall material recycling efficiency in the BIR system from 20.9 % (reference) to 29.6 %, at best, when introducing food waste treatment. The best recycling result is achieved for the insect scenario, due to a simple technology with a low degree of losses. The insect scenario achieves a lower ERR than the biogas scenarios, and here the biogas Bergen scenario performs best due to lower transport requirements than biogas Oslo. The biogas scenarios also results in lower net emissions than the insect scenario, when including the fossil fuels that the biogas replaces. At the same time the insect scenario provides the cheapest system solution, but the biogas Bergen scenario is found to be the most cost-efficient scenario when comparing increase in costs per emission reduction. In other words, the results reflect the complex system assessed. Moreover, this assessment report provides an assessment of a novel treatment technology, insect rearing, to the field of food waste treatment in urban waste management systems.

Sammendrag

Avfallssektoren har en nøkkelrolle i overgangen til sirkulær økonomi. Sektorens hovedformål er i endring. Fra å skulle håndtere avfall på en miljøvennlig og trygg måte, er det et økende fokus på bransjens rolle i å fremstille materialer som kan gjenintroduseres som råmaterialer på markedet. Avfallssektoren er en viktig tilrettelegger for økt resirkulering og dermed også sirkulasjon av ressurser på markedet.

Formålet med denne oppgaven er å utføre en helhetlig analyse av effektene av å introdusere matavfallssortering i Bergensområdets interkommunale renovasjonsselskap (BIR) sitt innsamlingssystem for husholdningsavfall. Hovedbeskjeftigelsen i arbeidet med denne masteroppgaven har vært å samle inn og prosessere data som danner grunnlaget for systemanalysen. Dataen er analysert ved bruk av en modell utviklet av student Pieter G. Callewaert sammen med veileder Professor Helge Brattebø i 2017. Modellen er basert på materialstrømsanalyse (MFA) -metodikk, med avfallsdata fra BIR som det viktigste datagrunnlaget. Videre har modellen et energilag, som krever systemspesifikk energidata knyttet til transport, forbrenning og annen prosessering av avfallet. I tillegg er klimagassutslipp og kostnader beregnet manuelt, men ved bruk av blant annet material- og energiresultater fra modellen.

To behandlingsløsninger for matavfall i tre ulike systemer er modellert. De analyserte alternativene er biogassproduksjon i Bergen, biogassproduksjon i Oslo og insektsdyrking på Voss (Hordaland). De ulike systemløsningenes ytelse i materialgjenvinning, energigjenvinning, klimagassutslipp og kostnadseffektivitet er analysert og sammenlignet.

Analysen er av helhetlig karakter, og gir et bredt spekter av resultater for alle scenarioer. Fellestrekk i funnene er at en økning i materialgjenvinning og lavere netto klimagassutslipp oppnås på bekostning av lavere energigjenvinning og økte kostnader. Funnene viser at utsorting og behandling av matavfall øker BIRs materialgjenvinningsgrad (MRE) fra 20.9 % (referanse) til 29.6 % som beste resultat. Det beste resultatet for materialgjenvinning oppnås for insektsscenarioet, takket være en enkel anvendt teknologi og lite tap underveis. Insektsscenarioet oppnår en lavere energigjenvinningsgrad (ERR) enn biogassløsningene, og her har biogass Bergen best resultat grunnet et mindre transportbehov sammenligent med biogass Oslo. Biogasscenarioene har også mindre netto klimagassutslipp, når det fossile drivstoffet som biogass erstatter er medregnet. Samtidig er insektsscenarioet den billigste løsningen, men biogass Bergen oppnår bedre kostnadseffektivitet når kostnad per reduksjon i kg CO_2-ekvivalenter sammenlignes. Med andre ord reflekterer resultatene kompleksiteten i et avfallssystem. Dessuten tilfører denne analysen en ny teknologi, innsektsdyrking, som et alternativ til matavfallsbehandling.

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List of Abbreviations

AD anaerobic digestion.

AP Acidification potential.

BIR Bergensområdets interkommunale renovasjonsselskap.

BSF Black Soldier Fly.

CE circular economy.

DH district heating.

DM dry matter.

EC European Commission.

EEA European Economic Area.

EPAV Environmental Protection Authority Victoria.

EPRS European Parliament Research Service.

ERR energy recovery rate.

EU European Union.

F2W2F Food2Waste2Food.

FAO Food and Agriculture Organization.

FCR feed conversion ratio.

FW food waste.

G&M Glass and metals.

GB green bags.

GDP gross domestic product.

GHG greenhouse gas.

GWP Global warming potential.

HTP Human Toxicity Potential.

HWEEE Hazardous waste and WEEE.

IRIS Interkommunalt Renovasjonsselskap i Salten.

ISWM integrated sustainable waste management.

IVAR Interkommunalt vann, avløp og renovasjon.

IWM integrated waste management.

LCA life cycle assessment.

LCC life cycle costs.

LCI life cycle inventory.

LCIA life cycle impact assessment.

LCM lightly contaminated masses.

LHV lower heating value.

MFA material flow analysis.

MRE material recycling efficiency.

MRF mterial recycling facility.

MSW municipal solid waste.

NIBIO Norsk institutt for bioøknomi.

NOK norsk krone (Norwegian krone).

NTNU Norwegian University of Science and Technology.

ODP Ozone depletion potential.

P&C Paper and cardboard.

PMFP Particulate matter formation potential.

RBA Romerike biogassanlegg (biogas plant).

RfD Renovasjonsselskapet for Drammensregionen.

RiG Renovasjon i Grenland.

ROAF Romerrike Avfallsforedling.

RS recycling station.

RUL recycling–utilization–landfilling.

RW residual waste.

SFA substance flow analysis.

SSB Statistisk Sentralbyrå.

SWM solid waste management.

TC transfer coefficient.

tkm tonne-kilometres.

TS total solids.

UFF U-landshjelp fra Folk til Folk.

UNFCCC United Nations Framework Convention on Climate Change.

WEEE Waste Electrical and Electronic Equipment.

WFD Waste Framework Directive.

WRAP Waste Resources Action Plan.

WtE waste to energy.

1 Introduction

The industrialization over the last century has resulted in economic growth and increased welfare for economies across the world. Over the last few decades, the negative impacts following this development has gained increased awareness amongst the public and policy-makers. With the dominating linear economic system of today, the economic growth is coupled with increased consumption and thus also results in an increased resource use and waste generation (Sjöström and Östblom 2010). The virgin resources and materials continuously entering the global economy are causing a growing stock of waste, and increasing welfare is attributed much of the responsibility for this development in Western Europe. The concept of circular economy (CE) is introduced to counteract this established correlation and facilitates an economic growth without increasing resource exploitation, resting on reuse and recycling of the resources already in circulation. Circular economy has gained importance on the political agenda, particularly within the European Union (EU). The waste management sector plays an important role in the transition towards a circular economy (Hollins et al. 2017), where resources circulates within the economy instead of being disposed of and thus leaving the economy, as in a linear economy.

The solid waste management (SWM) sector is undergoing a paradigm shift and has profoundly changed its agenda the last three to four decades. The increasing attention drawn towards environmental concern and resource efficiency in the public and political landscape has also influenced the direction of the SWM sector. From focusing on disposal methods, the municipalities are increasingly targeting measures for a growth in prevention and recycling rates (European Environment Agency 2016). Sustainable development is being put in the front-seat by European decision-makers with the Circular Economy Package passed by the European Commission (EC) (EC 2015). A significant responsibility is assigned the waste sector for facilitating an increased level of material recycling in EU's Circular Economy Package. In the action plan released in 2015 (ibid.), food waste is presented as a priority area, due to the significant environmental, economic and social consequences of resource loss along the value chain in the food sector and consumption. Food waste is currently a large part of the waste flow from the Norwegian households (Relling and Grevskott 2017), and represents a significant potential for *prevention* through increased consumer awareness and *material recycling* through alternative treatment and recycling technologies.

Through the European Economic Area (EEA) agreement, Norway is required to follow the waste directives introduced by the EU. The targets for municipal solid waste (MSW) and packaging waste is a recycling rate of 65 % and 75 %, respectively, by 2030. This means that municipalities and the respective companies that are responsible for handling the municipal waste are committed to facilitate an increase in the recycling rate by supplying services and treatment technologies to reach the targets. Today the total material recycling and energy recovery amounts to 63 % in Norway, and of this 35 % is energy recovered (Miljødirektoratet 2017a). The intermediate target for MSW is an material recycling rate of 50 % by 2020. Norway is still far away from reaching this target, with material recycling of household waste being 38 % in 2017 (Sæther and Skjerpen 2018). This demonstrates that extensive measures must be introduced to reach the 2030 recycling targets.

In a White Paper released in June 2017 regarding waste management related to the circular economy efforts (Meld. St. 45 (2016-2017)), the Norwegian authorities points to wet organic waste (hereby food waste) as a fraction of great importance to achieve increased material recycling from the households. Eliminating the food waste fraction from the residual waste (RW), is an important measure because of the significance the fraction represents in the RW, and hence a significant potential increase in material recycling. In addition, this will contribute to a more pure residual waste type, in which the remaining fractions become easier to segregate at central sorting facilities, for instance. Furthermore, the fraction itself is an important part of the national strategy for biogas production. Today, approximately 70 % of the Norwegian population is offered source segregation of food waste in the households, but recently passed EU legislation forces solid waste management companies to offer food sorting services in Norway and the EU by 2023 (Wilsgaard 2017).

BIR Privat AS is a branch company of BIR AS (Bergen interkommunale renovasjonsselskap) and is responsible for handling household waste in the municipality of Bergen, Norway's second largest city, and eight surrounding municipalities. BIR stresses that the objective of their activities is to obtain quality recycling, *kvalitetsgjenvinning*, to ensure that the sorting and recycling they facilitate for has a beneficial result for the society and the natural environment (Raadal, Soldal, et al. 2015; BIR Privat AS 2016). The motivation for initiating measures to improve the environmental performance of the system should result in net benefits in a system perspective, obtain secondary raw materials of a beneficial quality and seek to avoid problem shifting.

Waste systems perform differently as a consequence of the technologies applied, transport and collection methods and chosen downstream solutions. These solutions affect the resulting recycling rates, greenhouse gas emissions and energy use, and thus the environmental performance. Today, BIR does not offer source segregation of food waste from households, and the fraction is sent with the residual waste to incineration. Alternative solutions for food waste sorting and anaerobic digestion treatment was investigated in 2013, and the results shows that the environmental returns were minor compared to the costs (Igesund et al. 2014). This was particularly due to the lack of a market for bioresidues in the Western region of Norway. However, as part of BIR's strategy towards 2020, a target is to initiate another study of possibilities for organic waste treatment (BIR Privat AS 2016). For this reason, and since organic waste sorting will soon be mandatory for all SWM companies through the EEA agreement (Wilsgaard 2017), alternative food waste treatment solutions are studied in this report.

The main objective of this research paper is to represent the system of BIR Privat AS for 2017 and in future scenarios, showing how the recycling rates, emissions and energy use can change for different food waste treatment solutions. In addition, an assessment of the costs of the different scenarios is performed. Cost-effectiveness is an inevitable premise for a sustainable society, and therefore included in this analysis. The main research question to be answered is *How does initiated measures affect the environmental performance of the solid waste management system of BIR in terms of recycling rate, energy efficiency and greenhouse gas emissions?* More specifically, *What are the overall effects and/or consequences of introducing food waste sorting and treatment?*, *Which system solution performs best of the scenarios introduced and the reference scenario in the applied model?* and finally, *Is there a cost-effectiveness in suggested implemented solutions compared to the current?*. This study models the system with regards to the given indicators, and compares different future scenarios.

A reference scenario based on data and statistics for 2017 is developed for a time frame from 2017 to 2030, in addition to three scenarios where alternative treatment solutions for organic waste are implemented. The assessed treatment solutions for organic household waste are biogas- and insect production. The purpose of this study is to compare the environmental performance of the different system solutions. With the fundamental results obtained from the model, the critical performance indicators of the system can be identified and the effect of measures will be analyzed.

The system is represented and calculations are performed with the use of a material flow analysis-based model developed by students at NTNU in 2016/2017. This is a generic model for municipal waste management systems, using Microsoft Excel for system implementation and output, while a Matlab program handles the calculations. The model uses input of material flows and specific energy data to calculate the energy and material layer of the system. In addition, emissions and costs are calculated manually outside of the model to provide results for the environmental and economic impacts and/or savings of the alternative scenarios.

2 Literature Review

In this section an introduction to circular economy and the solid waste management (SWM) sector is given, before going in-depth on the challenges of food waste.

2.1 The Circular Economy

circular economy (CE) is a concept where an industrial society have closed supply chain loops. Resources are reused over and over again, and hence the economy is regenerating itself (Ellen MacArthur Foundation 2017). The circular economy model is, in contrast to the dominating linear economy of today ("take, make, dispose"), designed with a primary objective to retain resources that enters the economy in circulation and in the best quality possible. It aims to minimize or, at best, eliminate generation of waste. A functional circular economy is designed so that the residues created are redirected back into a production process, creating a loop where resources circulate.

CE is increasingly recognized among policy-makers as an important measure towards a sustainable industrial society, and has particularly prospered the last decade. However, the fundamental ideas are not new. The modern origins can be traced back to Europe in the 1970s (Geissdoerfer et al. 2017). The CE term is a later addition, but central ideas like closed loops, waste prevention, dematerialisation and resource efficiency were discussed also then.

The circular economy model emphasize life-cycle thinking to manage the economic and environmental challenges we face today. The concept represents new business strategies, which in turn can stimulate innovations and introduce new services (Ritzén and Sandström 2017). However, a successful implementation of a CE model is highly dependent on targeted policies and large structural changes in all parts of *the loop*, including changes in consumer behaviour.

Germany and the Netherlands are considered pioneers in implementing the policies that are the foundation of the EU circular economy strategy (Geissdoerfer et al. 2017). The policies were established by the European Commission (EC) for the first time in 2011 and adopted in 2015 with the EU *Action Plan For The Circular Economy* (EC 2015). The Action Plan is a strategy measure to support the transition towards a circular economy, and establishes generic measures as well as sector and material specific efforts that should be implemented (EC 2017a).

Waste management is one of the four key action areas in the commission's circular economy Package (Henry 2016). The three other key action areas are production, consumption and secondary raw materials, and constitute the complete supply chain of an industrial society. The waste sector is an important facilitator for the closing of loops and reintroduction of the waste to the economy as resources in a circular economy. As the waste diminishes, the sector increasingly attains a role as a resource manager. The activities in the waste sector are dependent on how much we consume, what we consume and how the commodities that become waste are produced and assembled. In other words, for it to become possible to exploit its full potential, it is vital that all parts of the economy are designed to comply with the CE concepts. This calls for an intertwined economy where sectors and the population work together across life stages to implement closed loops.

The quantitative targets established in the EU Circular economy package for the waste management sector are to increase the recycling rate of municipality waste and packaging waste to 65 % and 70%, respectively, in addition to the target of reducing the amounts of municipal waste in landfills to a total 0f 10 % within the EU (EC 2017b).

The European circular economy targets are formed with the purpose of reducing the input of resources to the economy and releasing pressure on the environment. The products, components and materials already introduced in the economy are utilized at a better rate. In addition to the environmental benefits of CE, the European Commission expect a growth in GDP for member states, encouraged innovation and increased security of supply (Henry 2016). As of today, Europe is a net importer of goods, making the region fragile for minor disruptions in the global market. Implementing a circular economy would mean to expand today's waste management sector within Europe, creating jobs, stimulating a regional secondary raw material market and on a longer term creating not only a circular path, but also a shorter loop, for an increased amount of products and resources in circulation within Europe. This increases the security of supply and reduces the environmental impacts caused by long intercontinental transport routes, among other beneficial consequences.

2.2 Solid Waste Management in a Circular Economy

The waste sector is an important subsystem in the industrial metabolism of an urban area (Zhou et al. 2015). The public health is dependent on a functional solid waste management (SWM) system, and hence the sector's primary objective is to maintain the inhabitants' safety and health. However, there is an ongoing paradigm shift in the SWM sector, where companies are moving from only facilitating a reduction in the accumulated waste amounts, to also encounter a minimum of environmental stress (Liu, Xing, and Liu 2017). In recent decades, new indicators are increasingly emphasized within the modern waste management sector: environmental, spatial and aesthetic factors. In western Europe today, most municipal solid waste management systems follow strict regulations to ensure human health. While it is not subject to the same strict regulations, the environmental performance of a solid waste management system is climbing on the political agenda. The importance of the sector is substantiated in the EU Circular Economy Action Plan. Furthermore, the potential for improvement of the environmental performance is being recognized by decision-makers, yet it is still greatly dependent on the individual waste management companies.

The typical modern urban waste management system of today is a well-engineered and complex system of modern technology and infrastructure processes. The complexity of an urban waste management system lies in the variety of problems to be solved. In addition, the objectives of the processes often are in conflict (Caruso, Colorni, and Paruccini 1993). Hollins et al. (2017) identify two main challenges for Europe in the future: (a) reduce waste generation and (b) align the objectives of SWM with those of CE. While (a) is an industry and behavioural dependent change, (b) should be targeted by the SWM system. Succeeding with this will be an important step towards reducing the environmental stress induced by a waste management system.

2.2.1 The Waste System

The municipal waste management system comprises of the value chain from household disposal and source segregation of resources, to collection, transport, treatment, and the final use or disposal. Generally, the processes are separated in four stages. Generation, Collection and Transport, Treatment and finally, Recycling - Utilization - Landfill(RUL) (Christensen 2010a). In the following paragraphs these stages will be defined for an urban waste management system handling household waste as they are defined in Christensen (ibid.), and related to their relevance in adopting to a circular economy.

Waste Generation
Generation of waste is the initial phase of the waste management system, and happens when goods goes from being a useful item valuable to its owner, to becoming a residual or redundant material of no value to the owner (Christensen 2010b). Waste is a consequence of human lifestyles (McDougall 2008), and currently an economy with a large GDP also results in large generation of waste per capita (Hollins et al. 2017). The main objective of CE is to reduce the environmental pressure that economic growth puts on the planet today (Ghisellini, Cialani, and Ulgiati 2016) by decoupling, i.e. to reduce waste generation while maintaining a growth in GDP. However, the waste generation per capita is increasing in approximately one-third of the EU member states. This is also the case in Norway, where it was generated approximately 433 kg waste per capita in 2016 (SSB 2017a).

Waste Collection and Transport

The infrastructure of a waste management system can be designed in numerous ways. In Norway it is common with at least some kind of source segregation with separate pickup at the households. Source segregation of paper and cardboard (P&C) is the most common, as well as plastics and organic waste in addition to the residual waste. These waste types are picked up at the households on a regular basis. Central collection points are also a much used in a collection system. This usually means shared containers for waste types on public parking lots and other central spots in a neighbourhood, which is emptied irregularly. To bring the waste from its source to a treatment facility, transport is necessary. The conventional solution is diesel trucks accommodating the collection technology (e.g. bins or containers). In pneumatic systems, a part of the transport is done in vacuum pipes, but many of these systems also needs motorized vehicles at the following stage, unless co-located with a treatment plant.

Treatment

The treatment processes referred to in this stage are either mechanical, thermal or biological. This can be e.g. mechanical shredding and/or sorting, incineration or composting. The objective of these processes are either to prepare the waste for the next process, or to extract energy. As part of a transition to circular economy, this treatment should prepare the resources to remain in the economy loop and facilitate the use of a material in manufacturing of new products. Similarly, producers should develop product design that facilitates material recycling (Hollins et al. 2017) and easy dismantling. This underlines the increased need for cooperation between the producers and waste management (or resource management) to realize a circular economy. The concept of circular economy favours treatment with the lowest degree of transformation, while a maximum conservation of the resource quality is preferred. Incineration with energy recovery is the preferred treatment for those fractions that cannot be prepared for material recycling due to poor quality or marginal quantities.

Recycling – Utilization – Landfilling (RUL)

The final stage in the waste system is the end-treatment of the waste, where it either is sold as a resource (and not considered waste anymore) or sent to become part of a final stock of accumulated waste. Thus, the processes introduced here are either material recycling, utilization of the waste in a new product or landfilling.

Recycling is when a material re-enters the production of similar goods as it originates from, or in other words, substitutes any virgin material (Christensen 2010a). Examples of such is paper waste used in paper production. Utilization is the use of waste fractions in a way or form that differs from its origin, such as using compost as fertilizer in the agriculture or bottom ash used in the base of roads. With such treatment, the waste usually returns to the economy as a resource with lower quality than its origins. Landfilling is the use of land area to dispose of and store waste indefinitely (Christensen 2010a). Today these areas are engineered to store the masses in a safe way with minimal environmental impact. However, due to the definite state of areas for such use, there is a need for surveillance for decades or even centuries after the filling is closed, to ensure the health and safety of surrounding areas and the environment in general. Due to this and the increasing landfill area needed for the accumulating masses, landfilling is the least desired end-treatment in a modern solid waste management system today.

2.2.2 The Waste Hierarchy

As mentioned already, the waste management sector is transitioning into a resource management sector. This process has been in development since the 1970s in some European countries (Geissdoerfer et al. 2017), and is today an important step towards a circular economy within Europe. The EU has announced their commitment towards a circular economy with the *Action Plan for a Circular Economy* released in 2015 (EC 2015). The overarching legislation regulating the waste sector is the Waste Framework Directive (WFD) 2008/98/EC (European Union 2008). The directive defines the most relevant terms in waste management, draws up in detail how the member states should handle the different waste types, implement efficient management and establish recycling targets for 2020 to ensure human health and environmental protection. The targets established in 2008 are now re-evaluated and enhanced with the CE Action plan.

The current targets for the waste treatment are ambitious and encourages the member states to develop the sector in a innovative manner. In the following paragraphs the relevant waste treatment technologies for an urban solid waste management system are presented in the context of the waste hierarchy and related to a transition towards CE. The waste hierarchy is a central principle in European waste legislation. The EC has adopted the waste hierarchy in their official documents as a figure of guidance (EC 2015). The EU Waste Framework Directive (WFD) 2008/98/EC, the up to present most important directive for the European waste sector, also uses the waste hierarchy as a guidance to the priorities and strategy of waste management.

The waste hierarchy is a ranking of the desired treatment tools in solid waste management, visualized in a typical manner in Figure 2.1. There exists different variations of the waste hierarchy, and Figure 2.1 represents a generic version. In EU's waste directive from 2008 (European Union 2008), some specifications are included. The same version is utilized in the CE action plan (EC 2015), and these specifications are commented on below.

Figure 2.1: A representation of the waste hierarchy

The upper section of the figure represents *Prevention*. Waste prevention is the most desired outcome, because it is the most efficient path to save resources and environmental pressure (Salhofer, Unger, and Bilitewski 2010). This is, among the most recent, repeated in Hollins et al. (2017) report for the European Parliament Research Service (EPRS) as one of two main targets to comply with the environmental stress of the sector. Even though the prevention has been on top of the waste pyramid from the beginning, it is traditionally not subject to priority of waste management companies, as it is difficult to see and measure (Lasaridi and Stentiford 2011). It is obvious that avoided waste represents less environmental stress than any treatment process in the waste management system. For instance, eliminating all avoidable food waste would have a more beneficial outcome than any biological treatment of this waste, but these effects are not visible in the waste statistics. Minimization is also a common addition to this section. When waste cannot be completely avoided or *prevented*, then a minimal amount of waste generation should be strived for. The WFD 2008/98/EC defines prevention as a measure taken to reduce the quantity of waste. Critics emphasis that reduction and prevention are two different measures, and emphasis that the action of prevention should focus on avoiding raw material depletion, and not just avoid that products become waste (van Ewijk and Stegemann 2016; Gharfalkar et al. 2015).

The following piece of the pyramid is *Reuse*, i.e. when a good is not useful for the owner, but a new owner finds use of it. This calls for input of resources (e.g. reparation) and infrastructure (transport). When put in system, a good can be delivered to secondhand retailer at a delivery point or directly. Either way, transport is involved. WFD 2008/98/EC defines this step as "Preparing for reuse", to underline that it includes the activities of repairing. However, this definition might not clarify that reuse is also a waste prevention operation when the product or component can be reused directly, and hence not subject to any kind of pre-processing (Gharfalkar et al. 2015). Thus, this piece in the waste hierarchy plays the same role as prevention and contributes to avoid that waste enters the SWM system.

For *Recycling*, some kind of treatment process is involved, since this includes dismantling and processing of a good or material fractions into a new substance, material or product. The adapted EU waste directive states the target for municipal waste recycling to be 65 % and 75 % for packaging waste by 2030 (EC 2015). Material recycling, is an essential part of a successful circular economy. This is the process where waste is transformed into resources that can be re-introduced in the economy. In a functional circular economy, producers are dependent on the supply of high-quality secondary materials, and material recycling facilities should be cooperating closely with producers to recycle waste so that it becomes resources with a demand. According to the current European legislation, organic food waste is material recycled through anaerobic digestion, when the treatment generates digestate that is used as a product, material or substance benefiting the agriculture or natural environment (Potocnik 2011). This way, anaerobic digestion of organic waste contributes to increasing the material recycling rate.

The *Recovery* piece of the pyramid refers to energy recovery. This is any process producing energy from waste, i.e. the product being power, heat or fuels. All over Europe, including Norway, there are waste incineration facilities that burn waste and simultaneously recovers energy from the heat generated by the incineration process. The main end-product is usually heat used for district heating in local residential and commercial buildings. In addition, there is usually electricity generated and sold, while some energy is used at the facility. The incineration plants of this type that are found in Europe are under strict legislation regarding emissions and gas treatment. The WFD 2008/98/EC Directive defines this part as "Other recovery" and only refers to energy recovery as an example. Hence, the definition above is more specific as provided by Waste Resources Action Plan (WRAP) (Gharfalkar et al. 2015).

When moving further down the upside down pyramid, the treatment becomes less desired because it degrades the resource quality. Simultaneously, the need for transport, processing, treatment energy and additives increases at the same time as the potential for contributions to the environmental pressure grows. *Disposal* or landfill is the least desired final treatment since it is an accumulation of resources in the environment that cannot be recovered (Christensen, Scharff, and Hjelmar 2010). Norway and many other European countries have introduced a ban on biodegradable compounds in landfills. This was implemented in Norway in 2009 as a measure to cut greenhouse gas emissions and to encourage better resource utilization (Avfallsforskriften 2004).

Landfill as the end-treatment solution is by many seen as incompatible with the circular economy concept, since it represents a finite process that is not designed to be linked with a new production process. It is however unavoidable for some compounds, e.g. toxic compounds that we do not want to circulate and spread. These are stabilized and stored in a safe manner KILDE. Among the circular economy targets, EU aims to decrease the amount of waste to landfills in all member states to less than 10 % of the total amounts of municipal waste. Today there is a great gap between member states regarding the amounts going to landfill. Six countries only landfill 3 % of the municipal waste, while some states exceed 90 % (European Commission 2015).

2.3 Performance of a Solid Waste Management System

The performance of a SWM system can be measured according to the three pillars of sustainability, namely economical development, environmental protection and social development. In this study, the two former are of main interest and analysed for a given system. It is assumed that the Norwegian waste sector's impact on social development is limited and that the services are similar for all inhabitants. With this, it is meant that the social sustainability of the Norwegian municipal waste management systems is assumed to be good.

A literature search on SWM system analyses shows that there is a lack of holistic assessment models. Morrissey and Browne (2004) reviewed models utilized in municipal waste management for decision-making and concluded that there was none of them that evaluated all the three aspects of sustainability at once. A waste management system is of a complex character, and this adds to the challenge of developing a comprehensive model. The complexity is reflected in the number of dimensions to take into account (ecological,economic, technical, social and political), as well as the spatial (single households, urban area, municipalities, national, regional or global) and temporal (short-term or long-term) scales, and complicates a holistic approach to modelling such systems (Chifari et al. 2016).

Nevertheless, the concept of a holistic approach to evaluating the sustainability of an SWM system has gained foothold in the literature and research community. integrated waste management (IWM) became standard within SWM research in the early 2000s (Wilson, Velis, and Rodic 2013). 'Integrated' points to a management model where the complete system is evaluated as a total unit, from collection methods to end-treatment (Chifari et al. 2016). This is in contrast to the practice of considering each process' performance individually. A critique to the early IWM is that the performance assessment is mainly technically rooted (Wilson, Velis, and Rodic 2013). Furthermore, adding 'sustainable' to *integrated and sustainable waste management* (ISWM) is a more recent development, arising as an expansion of the technical oriented IWM. ISWM[1] identifies three joint dimensions; the physical (technical) system, the sustainability aspects and the involved stakeholders (users and providers, governance) (Wilson, Rodic, et al. 2015). To analyse a SWM system holistically, the governance aspects should be included. Wilson, Velis, and Rodic (2013) points to the behavioural response to a new sorting scheme e.g., as an essential factor in evaluating the success and performance of a SWM system, and therefore should be more emphasised in analyses. Behaviour can be included by modelling different levels of success, but an accurate response is difficult to predict for introduction of e.g. a new recycling scheme to a new crowd.

In the following section environmental-, whereof energy is a natural factor, and economical aspects of a solid waste management system in a circular economy is looked into.

2.3.1 Environment and Energy

The energetic and environmental factors of any system are intertwined. A SWM system is both a consumer and producer of energy. Also, each segment in a waste management system, such as a process or the infrastructure, represents emissions caused or avoided.

The infrastructure and operation of a collection and treatment system, requires energy input. Any SWM system consumes energy during operation due to its dependency on transport in all parts of the system. Collection of generated waste, transport between facilities, transport to companies, to material and energy markets and to customers all requires energy. Transport cannot be eliminated from a waste system, but it can be optimized. This can be achieved by revising collection system (kerbside or bringing), collection frequency, car types and routes. For instance, level measurement tools in the bins can reduce the tonnekilometers if the information is used to customize a route that maximizes kilometer and capacity utilization each week. Wassermann (2003) found that optimization of the

[1]ISWM can also refer to integrated solid waste management.

collection frequency in urban areas is a more effective measure, while in dispersed settlements a route optimization is of a significant impact. Both the distances travelled (tonnekilometers), the motor efficiency and the fuels used influence the energy consumed, and thus also the emissions and environmental performance.

The global transport sector stands for 14 % of human-induced GHG emissions (Edenhofer et al. 2014), resulting from the use of fossil fuels. The domestic freight in Norway is dominated by heavy-duty vehicles and ocean transport (SSB 2017d). Nevertheless, the technology development is continuously increasing the performance and efficiency of vehicle engines. The trend for the development of vehicles is that the technology progress is directed at reducing carbon intensity and energy consumption for the combustion engines and alternatively fuelled motorized vehicles are on the rise. This is also true for heavy-duty vehicles, where testing of long-haul electric vehicles is recently initiated (Størbu 2018). According to Wassermann (2003) citing Wassermann, Salhofer, and Schneider (2002), the environmental stress from collection and transport of waste can stand for up to 90 % in a SWM system.

Furthermore, there are some fractions for which there is no recycling facility in Norway, and these fractions requires long-distance international transport. The share of Norwegian waste that is treated outside of Norway is increasing, but this can also mean that the resource exploitation increases when treated in a way the Norwegian waste sector currently can not offer (Miljødirektoratet 2017b). It can be argued that the transport is justified if the resources are utilized in a beneficial way. If the generation of waste keeps growing as projected, there will be large enough amounts to establish domestic facilities. This is however a dilemma for decision-makers. Should we dimension the future plans for an increase in generated waste or a reduction? Is it good management to work towards increased consumer and producer awareness, while upscaling capacity?

Secondly, the processes of the system requires energy input. In a life cycle perspective, the energy used to extract and produce raw materials (embodied energy), e.g. for construction, is also taken into account when quantifying the performance . This means that resources are needed to construct, operate and demolish a facility. Furthermore, any treatment facility needs an input of energy to e.g. operate the machines that shred waste into smaller units or optically sort fractions.

A waste management system can also produce energy. The most conventional methods for energy production (waste to energy (WtE)) is energy recovery by incineration or production of biogas (methane) through anaerobic digestion of organic waste. There are other options, such as landfill gas recovery (methane capture) (Willumsen and Barlaz 2010). Old landfills in Norway are required to capture landfill gas. However, since bio-degradable organic matter has been forbidden in landfills in Norway since 2009, the gas leakage is limited.

Incineration of waste is the most common WtE treatment of municipal solid waste worldwide (Lausselet et al. 2016). These incinerators contribute to reduced emissions of organic compounds, through the combustion of e.g. methane (CH_4) to carbon dioxide (CO_2), which is a less vigorous greenhouse gas in the atmosphere compared to methane and nitrous oxide (N_2O) (Oonincx and Boer 2012). Nevertheless, the incineration process itself generates emissions when materials are combusted. In Norway, the emissions from incinerators to the environment is regulated by the environmental legislation in Forurensingsloven (1981) (§. The incinerators have mandatory flue gas cleaning systems, and the permitted emissions are limited for numerous compounds. The purpose of these plants are to minimize the accumulated waste volumes, destroy certain toxic compounds and recover energy from the combustion of waste. Usually, heat is recovered and used in local district heating, in addition to electricity production. A positive side effect of the energy recovery is reduced emissions compared to oil burning installations that was commonly used for heating earlier (BIR Privat AS 2016).

Anaerobic digestion is the degrading of organic waste under anaerobic conditions into two main products; biogas and digestate (Angelidaki and Batstone 2010). While the biogas is used as a fuel, mainly in transport, the effluent, or residue of the production is a valuable fertilizer. The bioresidue contains nutrients like phosphorus and nitrates that makes it an attractive substitute to inorganic fertilizers for cropland. The bioresidue has less embodied energy and can contribute to closing nutrient cycles when utilized as an organic fertilizer in the agriculture (Haraldsen, Andersen, et al. 2011).

The first step of biogas production generates a biogas containing approximately 60 % methane (CH_4) and 40 % carbon dioxide (CO_2) (Måge 2017). The CO_2 is removed when the biogas is upgraded to become a transport fuel, bio-methane. To improve the beneficial effects of such a system, this CO_2 could be collected. The EU has together with collaborating partners in Norway, Netherlands and Poland been an initiator of the Food2Waste2Food (F2W2F) project (Lindum and Poznan University 2015), which studied and tested a closed cycle organic food-to-waste system. The technology integrated organic waste treatment, biogas production, digester and CO_2 capture and use. Organic municipal waste and manure was the main input to the biogas plant, and the by-products was instead of being accumulated or emitted, re-allocated to a greenhouse as nutrients for the cultivated plants in the form of CO_2 and digester.

The F2W2F demonstrated a successful implementation of a circular system. However, for this to work in practice, it is dependent on an extended collaboration between the agriculture and waste sector. The agriculture is the main consumer of the bioresidue, can be the receiver of carbon dioxide for greenhouses and can supply manure for the biogas production. However, the bioresidue is only suitable for soil with a nutrient shortage, and this varies around Norway. The farmland soil on the west coast and in Eastern Norway are subject to different usages and have different nutrient composition. The biogas residue has a beneficial composition for cropland and grain-growing (Haraldsen and Føreid 2015), of which there is much of in the eastern region of Norway. This is not case on the west coast, where the agriculture is mainly pasture and have an excess of phosphorus and nitrates (Igesund et al. 2014). This means that a biogas facility in the western region would need to transport the bioresidues to other regions in Norway who has a fertilizer demand, and results in reduced benefits due to the increased transport when compared to local treatment solutions.

There are several benefits of anaerobic digestion treatment compared to incineration with energy recovery that places AD above WtE incineration in the waste hierarchy. AD treatment preserves nutrients in the economy when the bioresidue is utilized and re-allocated to the agriculture. In comparison, the material output of the incineration process is ashes that contains some valuable metals and an inert mass that can be utilized road construction (approximately 20 % of the orginal mass).

2.4 The Challenge of Food Waste

In 2015 it was estimated that 100 million tons of food waste was disposed of in the EU-28, when including all parts of the supply chain (agriculture, food industry, wholesale, grocery trade, serving business and households) (Salomone et al. 2016; Stensgård and Hanssen 2018). Food waste is a large part of the waste generated in the households in Norway. A study performed by Østfoldforskning (Hanssen et al. 2013) showed that the average generated food waste was 78.8 kg per inhabitant in Norway in 2013. Of this, 46.3 kg is edible food, and represents both a structural and behavioural problem in Norway. Similar trends are detected in other developed countries (FAO 2015). The challenges of avoidable organic waste is being countered by the Norwegian authorities and the food sector itself with a industry-wide agreement to reduce the edible food waste with 50 % within 2030 (Stensgård and Hanssen 2018). The agreement was signed in June 2017 by 42 major companies from the food industry, grocery trade and serving business. Similar efforts such as research and awareness projects (e.g. ForMat, Matvett) has contributed to a reduction of edible food waste in the supply chain of 12 % from 2010-2015 (ibid.). There is still need for further reductions in order to achieve a significance for climate change mitigation. In 2015, the households contributed with roughly 60 % of the avoidable food waste in Norway, but also here there has been a reduction, and the awareness is on the rise (*Om matsvinn* 2018). A picking analysis performed by Bergensområdets interkommunale renovasjonsselskap (BIR) showed a potential of 71 kg organic waste per inhabitant from the household waste in the fall of 2017 (Relling and Grevskott 2017). Of this, 42 kg per inhabitant, or almost 60 % is found to be avoidable food waste.

Nevertheless, even when the edible food waste is minimized, there will always be a fraction of unavoidable food waste, such as fruit peel or bones from meat, that should be treated in an adequate and beneficial way that conserves the resources and, at best, also the nutrients. In the Bergen area the unavoidable food waste was found to be 17 % of the RW stream (food sorting is currently not offered here) (ibid.), and was the third largest fraction after Other combustibles (21 %) and Avoidable food waste (24 %). Food waste that is source segregated at households in Norway today is treated either by composting or anaerobic digestion (Hanssen et al. 2013). The main products are nutrient-rich soil and methane fuel, respectively.

Anaerobic digestion is the most common treatment technology for household organic waste. Today there are roughly 40 biogas plants in Norway (Martinussen 2017), supplied with input from various biological sources (e.g. manure) in addition to food waste. The production of biogas from food waste mainly replaces fuels for transport due to the low electricity price in Northern Europe (Jönsson and Persson 2003). If comparing incineration and biogas production treatment of organic waste, it can be argued that the environmental benefits of replacing fuels in motorized vehicles exceeds the replacement of electricity or heating in Norway due to the dominance of hydroelectric power in domestic electricity production. The transport sector is among the emitters of largest significance in Norway today (SSB 2017e), while the emissions from the Norwegian electricity production are low (NVE 2016). This is however system dependent, and the benefits and drawbacks can vary. Furthermore, using waste flows as input in biogas production is more beneficial than using produce directly. This way you do not need to allocate land for growing plants for biogas production, and more land is allocated to food production (Salomone et al. 2016). At the same time, the nutrients remain in circulation, and value is added to a reject stream that traditionally has been considered of low value. The nutrient-rich by-product bioresidues can be used as a natural fertilizer in the agriculture.

An alternative to the conventional treatment of food waste today is to use the organic waste as a nutrient source for insect farming and protein production. The insects can feed on food waste containing valuable nutrients and is, like anaerobic treatment, conserving the nutrients within the economy instead of leaving the economy indefinitely. The main output of such a process is high-value proteins, fat (oil) for feed, food (Smetana et al. 2016) or biodiesel production (Salomone et al. 2016), in addition to water, insect manure and food waste residues that can substitute fertilizer in the agriculture (Thévenot et al. 2018; Čičková et al. 2015). Insects are eaten in certain regions across the world today, but the concept of industrial insect farming for food is a more recent object of studies (FAO 2015). There are however, significant market barriers to introduce insects as human food in the western societies, as this is currently not a conventional part of western diets. In addition, there are legislative barriers within EU. E.g., it is differentiated between the use of insects as feed and as food . With that said, the potential for the products of insect farming are currently larger as a protein source in feed production. The interest for insects as a substitute for the traditional feed is rising. Insects can be large-scale cultivated with organic waste as input, and this is already being reared in China among other countries (van Raamsdonk, van Der Fels-Klerx, and de Jong 2017). The majority of the existing production farms direct their product to the animal feed industry.

In the literature on the topic of insect production, a large variety of benefits and impacts are described due to the varying contexts, technology and raw material input. Nevertheless, there are several factors that make insect farming an interesting alternative for food waste treatment. Issues from security of supply to environmental concerns motivates the introduction of alternative protein sources. The demand for animal protein is expected to increase with 70-80 % within 2050 (Oonincx and Boer 2012). While the current production of livestock stands for approximately 18 % of anthropological greenhouse gas emission (van Huis 2013), increased production of edible insects can potentially lower these emissions and add to the supply of protein feed and food for an increasing global population. Insects also have a low feed conversion ratio (FCR) (amount of feed input per produced amount of edible protein) compared to other protein sources, and the insect can be consumed as a whole without removing any residual parts (bones, head, fur) (van Zanten et al. 2015).

The performance of insect production systems are varies for different types of insects. The black soldier fly (*Hermetica illucens*), the housefly (*Musa domestica*) and the yellow mealworm (*Tenebrio molitor*) are the insects of most interest currently, due to their low FCR (van Huis et al. 2013). The insect types' preferred diets vary. The housefly thrives on a diet of manure from animals of mixed diet, and develops less well with manures from animals on a plant diet (Čičková et al. 2015). The black soldier fly BSF is more adaptable and can consume nutrients from masses with a high moisture content compared to the other two (Ringstad, A.S., 2018, personal communication). It also develops well in mixed diets of large variety, and organic food waste is therefore a good match. Furthermore, the BSF requires less energy for production and has a high nutrient content (de Boer et al. 2014), and is currently considered the most promising specie for insect protein production.

There are currently few studies that assess the performance of insect rearing with a life cycle perspective in the literature. Due to differences in system boundaries, comparative grounds, location and technology the resulting performance of the studied systems are varying.

One of the first LCAs on insect rearing was performed by Oonincx and Boer (2012). The environmental impacts of production of mealworms as a human protein source is compared to conventional protein sources (milk, chicken, pork and beef). The study has a cradle-to-gate approach, and studies all upstream inputs to the mealworm production, which is then viewed up against the literature on similar studies for the other protein sources. It is shown that the farming of insects have considerable lower GHG emissions and land use than the protein sources it would substitute. The energy use is comparative to the other systems, mainly due to the need for a high ambient temperature during rearing. This study assumes grain and carrots crops as the feed, meaning that similar systems utilizing reject streams such as food waste should have an improved environmental performance compared to this study. While Oonincx and Boer (ibid.) performed the research on a laboratory scale, Smetana et al. (2016) performs an LCA on an industrial scale system for insect rearing, comparing the environmental impacts of different diets. This study found that the environmental benefits of the system were highly dependent on the source of the insect diets. The system based on the low-value flow of municipal organic waste performed best when assessing 17 different midpoint impact categories. The authors conclude that low value insect feed with high protein content are most promising. The results were partly due to the benefits of utilizing the municipal food waste being considered a reject stream, and partly due to the favourable nutritional composition and density of the material.

Furthermore, van Zanten et al. (2015) studied rearing of the housefly on poultry manure and food waste. This study included the indirect consequences of producing less bioenergy due to less available organic mass, and found an increase in energy use and GWP (CO_2-eq), while the land use was reduced when substituting the insect proteins for fishmeal and soybean meals. However, the authors acknowledge that the environmental impacts are system specific and could be different in another system. With the current situation in Norway, for instance, the allocation of food waste to insect production would not mean a reduced biogas production as assumed by van Zanten et al. (ibid.), because the quantities of FW are far larger than the input to the existing biogas facilites.

Salomone et al. (2016) analyses insect rearing at a bioconversion pilot plant located in Southern Italy. The study concerns the rearing of the BSF larvae, performing an analysis of a system where the output of proteins are put to use as livestock feed, the fat as biofuel and the residues as fertilizer. The study found higher environmental impacts from energy use and GWP when comparing insect proteins to the production of soybean flour for feed. The impact category of land use was significantly reduced in the case of insect production. Nevertheless, the authors emphasize that there still are limited available data in the literature, and the sensitivity analysis changes the results significantly, moving towards a lower GWP when the comparison is done per mass gain of insects. The limited data and variation in performance results bear witness to a relatively immature field of research and technology. However, the potential for technical improvements of the developing technology, gives reason to believe there also is potential for a reduction in the specific environmental impacts of such a system.

3 Methodology

In this section the theory that the model is based on is introduced before the model itself is presented. The final part of this section is devoted to the case study; BIR Privat AS. The data collection method is accounted for and the scenarios selected in collaboration with the company are presented.

3.1 Material Flow Analysis

Material flow analysis (MFA) is a systematic assessment method for material flows and stocks within a defined system, designed to map the metabolism of an anthropogenic system (Brunner and Rechberger 2016). An MFA's objective is to represent the material movements in a system by applying mass balance to the processes in view, and yields an account of material pathways (Allesch and Brunner 2015). The basic theoretical foundation of the methodology makes MFA suitable as decision support for policy makers. Based on the law of mass conservation, the MFA methodology utilizes mass balance to define a process' input, output and stock change. This can be done for an entire system, where the mass balance of each process within the system sums up to a total mass balance for the entire system. In this way, an MFA shows the sources, sinks and the pathways of the resources within a system for a given time frame. The mass balance of a system or a process with k_i input flows and k_o output flows is given in Equation 3.1 (ibid). All inputs, subtracted the outputs and stock change should summarize to zero. In other words, what goes in must either come out or accumulate in the system.

$$\sum_{k_i} \dot{m}_{in} = \sum_{k_o} \dot{m}_{out} + \dot{m}_{stock} \tag{3.1}$$

Furthermore, transfer coefficients (TC) describe the partitioning of the output of a material leaving a process (Brunner and Rechberger 2016). Equation 3.2 shows the transfer coefficient for material k in process i, leaving to process j, defined by the mass flow X of material k. All transfer coefficients of a material leaving a process sums to 1, and shows the distribution of a material exiting a process.

$$TC_{k,ij} = \frac{X_{k,j}}{\sum_k X_{k,i}} \tag{3.2}$$

Transfer coefficients are useful to identify the destination of materials, and describes a process' performance on delivering materials to a desired end-process. In waste management mapping, this can be useful to get an overview of a system's sorting degree towards a specific market.

Figure 3.1 is reproduced based on research by Allesch and Brunner (2017) and shows the general features of a waste system, visualized in a format commonly used in MFA. The figure can be read from left to right, where the main inputs are on the left side, and the outputs gathered on the right side. The scheme displayed here is valid for all collected waste of a generic SWM system. Inputs and outputs of the system are those crossing the stippled lines of the system boundaries, while there also exists a mass balance for each process.

MFA is much used in environmental studies of SWM systems. The method supplies a set of quantitative results of flow composition and pathways, which can be assessed with regards to targets of material recycling for instance, such as those put forward by the EC (Allesch and Brunner 2015). The growth of MFA in the field of industrial ecology is attributed the increasing demand for resource- and environmental assessments from industry, commerce and authorities, and can e.g. contribute to reach increased recycling rates by identifying areas of secondary raw material loss.

3.2 Life Cycle Assessment

Life cycle assessment (LCA) is a methodology that examines a product's or system's energy and environmental performance through-out the life cycle. By studying and characterizing the mass and energy flows of a product, one can map the environmental impacts. This involves the production, use and disposal or recycling

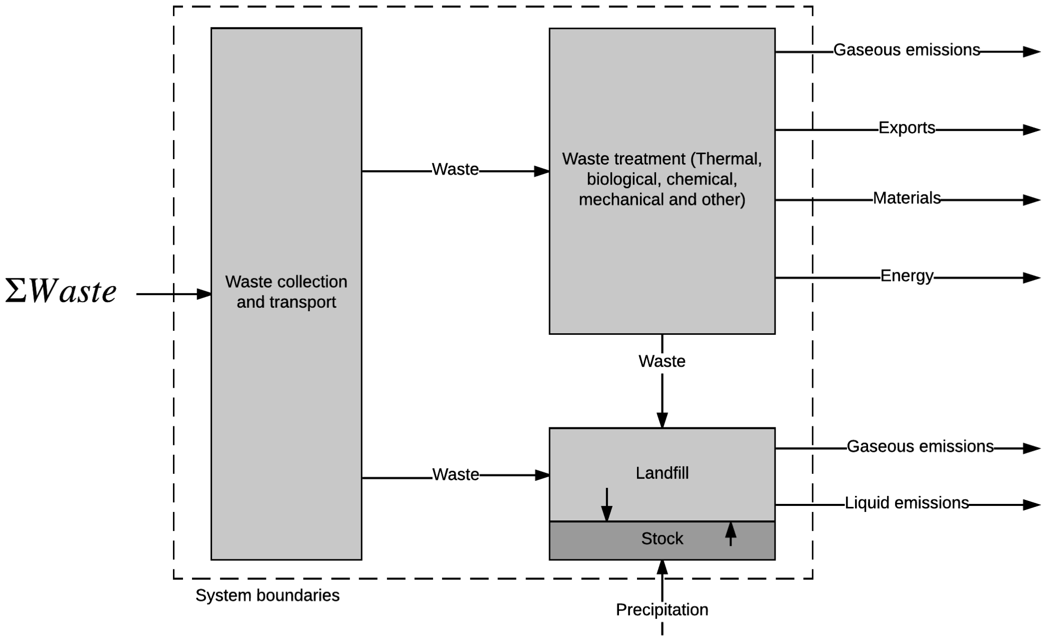

Figure 3.1: Material flows of a generic waste system.

stages for a product lifetime or process system. Environmental impact types such as global warming, particulate matter formation or acidification - among numerous indicators - are selected and represents the link between the system's emissions and/or resource use and the impacts generated. The objective of an LCA is usually to understand the stress the system studied inflict on the natural environment (Graedel and Allenby 2010).

The life stage of waste management is often simplified or completely disregarded in an LCA of a product system (Hauschild and Barlaz 2010), but more recently LCA is increasingly performed on waste management systems exclusively to map their environmental performance. This is a development underlining the increasing recognition of the sector as an important contributor to a sustainable future, while the impact of the end-treatment life stage is acknowledged. In the following paragraphs we introduce the LCA methodology with an emphasis on the practice for LCA on SWM systems.

The LCA methodology framework is standardized in the ISO 14040:2006, and constitutes of four stages (ISO 1997). These and the flow of information between the stages are shown in Figure 3.2, and will be shortly introduced for SWM systems in particular in the following paragraphs. (ibid.)

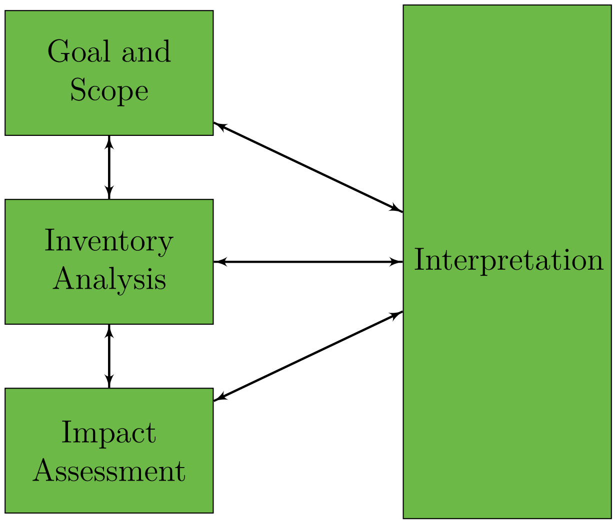

Figure 3.2: The methodology of LCA.

3.2.1 Goal and Scope

The initial phase of a life cycle assessment is to determine the goal and scope of the study; what are we studying and why. Furthermore, specifications to how the assessment is performed is also established in this stage.

The goal of the study should make clear what the purpose is and in what context the findings are to be used, i.e. how the assessment is providing environmental decision support (Hauschild and Barlaz 2010). The goal can be to uncover the life stage that contributes the most to environmental stress or to compare the impacts of two product system.

The functional unit is defined to frame the scope of the analysis. The functional unit represents the object of the study, and is a reference unit to which the inventory data is aligned with (Roy et al. 2009). It reflects an amount, a volume or another reasonable measure of the service the product provides to a user (Rebitzer et al. 2004). For a SWM system LCA this can be e.g. the amount of waste treated or the composition of the waste handled (Hauschild and Barlaz 2010).

A system boundary is usually determined by the degree of contribution for each process in the total system, relevant to the goal of the study. For an SWM LCA, the system boundary deviate from the system boundaries of a product system LCA. For an SWM system, the primary objective is the end-of life treatment stage, and thus the system boundaries only encompass processes in this life stage.

3.2.2 Inventory Analysis

The *Life Cycle Inventory (LCI) analysis* is the stage of data collection, and is the most time consuming part of an LCA. The work here comprises of mapping all inputs and outputs of the system (Roy et al. 2009). This includes energy, water, material inputs in addition to outputs of products, bi-products and emissions.

3.2.3 Life Cycle Impact Assessment

Life cycle impact assessment LCIA consists of sorting and allocating the raw data acquired in the inventory to the right impact categories to be able to assign emitted (or consumed) quantities to a corresponding effect. There exists numerous impact categories or indicators, but usually a selection of indicators relevant for the goal of the study are utilized. Some of the most common indicators are Acidification potential (AP), Global warming potential (GWP), Ozone depletion potential (ODP) and Particulate matter formation potential (PMFP). The objective of this stage is to interpret the quantities and link the emissions to the impacts caused.

The first step of LCIA is *classification*, where the raw data is assigned to an impact category (Graedel and Allenby 2010). Secondly, the impact of the sorted data is quantified in the *characterization* step. This step is presented in Equation 3.3 as given in Graedel and Allenby (ibid.).

$$S_j = \sum_i C_{i,j} \times E_i \tag{3.3}$$

The total stress indicator S_j for impact category j is found by multiplying the mass flow E_i for i data sources with a characterization factor $C_{i,j}$, given by source i's allocated impact in impact category j. The characterization factor is usually given specifically, i.e. as equivalents per mass.

3.2.4 Interpretation

The final stage of an LCA comprises of reviewing the findings in the impact assessment and establishing if the findings are in line with the goal of the study. If not, one should go back to review and adjust what is causing this mismatch, and reconsider the modelling of the system. The LCA methodology rests on a dynamic framework, and thus the assessment involves feedback loops to assure the objective to be fulfilled. The interpretation is therefore interconnected with all the other stages in Figure 3.2.

3.3 The Model

The model applied in this study is a model developed by a group of NTNU students in 2016, in a project led by supervisor Professor Helge Brattebø. The development of this model is motivated by a lack of holistic assessment methodologies for SWM systems. This case study is a continuation of the project and applies the model developed on new case data. The model is designed to represent an urban waste system from the initial stage of waste collection or delivery from the households, to the market of final use or disposal. It employs Microsoft Excel as the main user platform. An excel file is used as input in a Matlab script, which reads, calculates and generates results in a new excel file. The generic format allows for easy adaption to any SWM system. The format enables seamless tailoring of existing and simulated scenarios.

When assessing the performance of a SWM system, literature shows that LCA is the most common methodology, providing a comprehensive system description and quantitative results that are easily translated into concrete measures (Laurent et al. 2014). Brunner and Rechberger (2016) favours MFA as a decision support tool pointing to the coherent system boundaries and holistic system definition that supplies reliable numerical flow and transfer coefficient data. Hence, the difference between the two can be said to be that LCA is an evaluating assessment methodology, while MFA provides numerical values of every flow in the system. Few studies combine MFA and LCA for SWM studies. Turner, Williams, and Kemp (2016) quantifies the waste flows, before analyzing the case with LCA. Similarly, the model developed and applied here, implements a material layer (MFA) and an energy layer (LCA). The energy layer is based on the flows in the material layer, and hence the material flows serves as the basis inventory for the energy analysis (Allesch and Brunner 2015). The workings of the model and its indicators are described in detail in the following sections.

3.3.1 The Material layer

The material layer is the fundamental layer of this model. The generated and collected waste flows represents the initial fundamental data input, and together with the transfer coefficients the pathways for waste types are partly defined and partly calculated by the model in Matlab.

Two performance indicators are applied in this analysis to describe the performance of the material flows, and these are part of the output the model generates. The indicators identify the processes or waste types that are critical for the material recycling performance of the system. These indicators are introduced and defined here, following the definition introduced by P. Callewaert in the model documentation (Callewaert 2017b). A summary of the parameters utilized in this section is provided in Table 3.1 below.

Table 3.1: Overview of the parameters defining the material layer of the model.

Parameter	Definition
a	Collection processes
c	Material markets
d	Bioenergy markets
e	Compost processes
i	Waste types
j	Waste fractions
0	Waste generation process

3.3.1.1 The Collection Efficiency

The collection efficiency describes the performance of the initial part of the waste system; the collection processes. This can be a pointer to if BIR's collection system is understood and correctly utilized, i.e. it tells us to what degree the residents source segregate their household waste correctly. To evaluate this, the model calculates the amount of waste that is collected in the correct waste type bin, relative to the total amount of waste generated, as shown in Equation 3.4. For each fraction j, the annual amounts $X_{0a,i=j}$ collected at a collection processes in the correct bin (waste type) i, is summarized and divided on the total collected waste amounts.

$$\eta_{coll} = \frac{\sum_j \sum_a X_{0a,i=j}}{\sum_j \sum_i \sum_a X_{0a,ij}} \tag{3.4}$$

Here 0 represents the waste generation process, and a represents all the collection processes (home collection, central collection points or recycling stations). Accordingly, the numerator sums all waste of fractions j collected in waste type i when the fraction matches the waste type ($i = j$). Similarly, the denominator sums all waste generated (0) and collected in the system studied for all waste fractions j in all waste types i.

3.3.1.2 The Material Recycling Efficiency

The material recycling efficiency (MRE) is the total amount of waste that is material recycled and ends up on a marked as new raw material, at a bioenergy market or composted over the total amount of waste generated. The indicator is defined in Equation 3.5. The numerator represents the summation of all waste entering the different markets. Here, vector c is the material market, d the bioenergy market and e the compost processes. E.g., $X_{xc,ij}$ is the total waste flow entering the material market c from x treatment processes in tons per year. In similar fashion as for the collection efficiency, the denominator is a sum of all waste entering the system.

$$MRE = \frac{\sum_j \sum_i (\sum_c X_{xc,ij} + \sum_d X_{xd,ij} + \sum_e X_{e0,ij})}{\sum_j \sum_i \sum_a X_{0a,ij}} \tag{3.5}$$

3.3.2 Energy layer

The energy layer calculates the energy consumption and production in the system. The equations that defines the energy layer are described here in a generic way. Also this section is based on the layer presented in the documentation accompanying the model (Callewaert 2017b).

The model requires system specific energy data input for each transport lap and process. The required input to the energy layer is listed in Table 3.2, while the main energy data for the performed analysis is given in Appendix C and D.

Table 3.2: Model input to energy layer.

Data	Unit
Energy carriers	[name]
Calorific value of feedstock	KJ/kg
Methane yield	Nm^3/t
Incinerator efficiency	%
Distance	km/trip
Transport energy intensity per fuel	kWh/tkm
Process energy consumption per fuel	Kwh/t

In addition to the parameters defined in Table 3.1, some new parameters are applied to define the energy layer, and these are presented in Table 3.3.

Table 3.3: Overview of the parameters defining the energy layer of the model.

Parameter	Definition
tr	Transport
pr	Processes
fs	Feedstock
AD	Anaerobic digestion
INC	Incineration
f	Fuel type
h	Incineration process
k	anaerobic digestion process
l	Fertilizer markets
m	process
y,z	processes (used in transport calculations)
LHV_j	lower heating value (LHV) for compound j

The overall energy recovery rate (ERR) is the total output of energy (generated) over the energy input to the system per year and given in Equation 3.6.

$$ERR = \frac{E_{out}}{E_{in}} = \frac{E_{inc} + E_{AD}}{E_{tr} + E_{pr} + E_{fs}} \tag{3.6}$$

The variables in Equation 3.6 are defined separately for output and input of energy, respectively, below.

3.3.2.1 Produced Energy

E_{out} is the energy produced in the system in kWh per year. As described in section 2.3.1, the two conventional components contributing to the energy production in an solid waste management (SWM) system is the waste-to-energy incineration (E_{inc}) and biogas fuel production by anaerobic digestion (E_{AD}). The energy recovered from incineration process h is estimated by multiplying the flow of fraction j in waste type i going into the incineration process with the lower heating value LHV_j (KJ/kg) of fraction j and the efficiency of the plant, η_h^{inc}. Here $X_{xh,ij}$ describes the waste flow of waste fraction j in waste type j coming from process x and going to the incineration process h. The total annual E_{inc} in kWh/year is then found by summarizing Equation 3.7 over h incineration processes in the system.

$$E_h^{inc} = \sum_i \sum_j (X_{xh,ij} \cdot LHV_j \cdot \eta_h^{inc}) \tag{3.7}$$

Similarly, the annual energy produced in anaerobic digestion (AD) process k is the sum of j fractions in i waste types leaving the biogas process k and entering the bioenergy market d and fertilizer market l, multiplied with the LHV for CH_4 and the methane yield (Nm^3/t), $Yield_k$.

$$E_k^{AD} = \sum_i \sum_j ((X_{kd,ij} + X_{kl,ij}) \cdot LHV_{CH_4} \cdot Yield_k) \tag{3.8}$$

The reason for including the fertilizer market in Equation 3.8 is that the methane yield filled in by the user is based on the total input to the biogas facility. In other words, $Yield_k$ is given per ton organic waste treated, and supplied by the user for each AD treatment process k in the system studied. An LHV value for CH_4 of 6.5 kWh/Nm^3 is pre-defined in the model.

3.3.2.2 Energy Input

E_{in} is the total annual energy input to the system and consists of three contributions. These are transport energy, process energy and feedstock energy and given in kWh/year.

The consumed transport energy $E_{yz,i}^{tr}$ in kWh/year for the transport of waste type i from process y to z is defined in Equation 3.9. The variables in Equation 3.9 are defined in Table 3.4.

$$E_{yz,i}^{tr} = \sum_{f_t} (I_{yz,i,f_t} \sum_i X_{yz,i} \cdot D_{yz,i} \cdot S_f) \tag{3.9}$$

The process energy E_{pr}, is the total energy consumption in kWh/year. Equation 3.10 shows the energy requirement for treating waste type i in process m. This is found by multiplying the waste flow $X_{m,ij}$ entering process m with the specific energy input $E_{req_{m,i,fp}}$ of fuel type f_p required in process m to treat waste type i. The model allows for multiple fuel types and thus the calculation is done for each specified fuel type (f_p) and summarized. E_{pr} is then found by summarizing over m processes in the system treating i waste types.

Table 3.4: Variables defining the transport energy calculations.

I_{yz,i,f_t}	Energy intensity for transport lap y to z for waste type i and with fuel type f_t
$X_{yz,i}$	Waste quantity transported from y to z for waste type i
$D_{yz,i}$	Distance travelled from y to z with waste type i in kilometers per trip
S_f	Share of tkm transported with fuel type f

$$E_{m,i}^{pr} = \sum_{f_p}(\sum X_{m,ij} \cdot E_{req_{m,i,fp}}) \tag{3.10}$$

The feedstock energy E_{fs}, is the energy content of the waste entering the system in kWh/year and is defined in Equation 3.11. This variable is computed by the calorific value (LHV) of the waste fraction j, multiplied with the waste flow of fraction j in waste type i entering the system (generated in process 0), $X_{0,ij}$. The total E_{fs} in kWh/year for the system is found by summarizing over alle fractions j in all i waste types.

$$E_{fs} = \sum_j \sum_i (X_{0,ij} \cdot LHV_i) \tag{3.11}$$

3.3.3 Greenhouse Gas Emissions

A SWM system represents both generation of emissions and avoided emissions. Greenhouse gas (GHG) emissions are currently not included in the model, but calculated manually for the reference and food waste scenarios[2]. All activities within the system generates emissions, but for this analysis sorting, pre-treatment, final recycling, incineration and transport emissions are included. Landfill emissions are excluded due to the lack of reliable data and assumed negligible changes in this process' emissions between the scenarios. Moreover, the system boundary of the GHG calculations encompass the biological treatment in addition to the company processes. This is of relevance to include since it is here the main changes are made between the scenarios in addition to the incinerator and transport.

Due to the limited scope and time frame of this analysis, the specific emission factors were not collected specifically for this case study.

[2]GHG calculations are calculated for System B and thus only for the RW and FW waste flows. System B will in detail be introduced in section 3.4.1.

3.3.3.1 Generated Emissions

Four types of generated emissions are distinguished between in this analysis. These types are system process emissions, transport emissions, incineration emissions and material recycling emissions. The system process and transport emissions includes the company processes and transport laps included in the model and are based on the energy layer of the system. The incineration emissions are those generated directly into the air from the combustion of waste, while the recycling emissions are the emissions related to the final recycling process of the waste. The two latter emission calculations are based on the material layer. In the following paragraphs, the generated emissions are presented.

In short, transport and process emissions are found by multiplying the annual energy consumption provided by the model and an emission factor for each energy carrier as shown in Equation 3.12 and 3.13.

$$F_{x,i} = \sum_{n} M_{x,i,n} \times e_n \tag{3.12}$$

$$F_{yz,i} = \sum_{n} M_{yz,i,n} \times e_n \tag{3.13}$$

The process emissions are calculated with the use of Equation 3.12. The emissions from process x treating waste type i are found by multiplying the energy consumption, $M_{x,i,n}$, (kWh/year) with the emission factor e_n (CO_2-eq/kWh) for energy carrier n. The total emissions for waste type i in process x, $F_{x,i}$ (CO_2-eq/year), is found by summarizing over n energy carriers. The total emissions from each process can be found by summarizing Equation 3.12 over i waste types.

Similarly, the annual emissions caused by the transport lap from a process y to a process z for waste type i is found by multiplying the energy carrier's energy consumption for each waste type, multiplied with a specific emission factor. More precisely, the transport lap's energy consumption for transporting waste type i with energy carrier n, $M_{yz,i,n}$ (kWh/year), is multiplied with the corresponding energy carrier's emission factor e_n in kg CO_2-equivalents per kWh. The total emissions, $F_{yz,i}$ (CO_2-eq/year), for the transport of waste type i from process y to z is then found by summarizing over n energy carriers, and is shown in Equation 3.13.

Table 3.5 gives an overview of the emission factors applied for the the energy carriers present in this study. The emission factor for diesel is collected from Norway's National Inventory Report for greenhouse gas emissions 1990-2015 (Miljødirektoratet, SSB, and NIBIO 2017), while the emission factors of Norwegian electricity production is collected from NVE (2016).

Table 3.5: Emission factors for the energy carriers utilized in this analysis.

Generated emissions	kg CO_2-eq/kWh
Diesel	0.265
Electricity	0.017

For the recycling and incineration processes the generated emissions are found by multiplying the material flows with a specific emission factor in kg CO_2-eq per kg recycled/incinerated material. The total emissions E_i in kg CO_2 per year for waste type i is given in Equation 3.14. The fraction j of the waste flow of waste type i, X_{ij}, in tonnes per year, is multiplied with a treatment specific emission factor e_j in kg CO_2 per tonne. The total emissions caused by treating waste type i is found by summarizing over all fractions j.

$$E_i = \sum_j X_{ij} \times e_j \tag{3.14}$$

The emission factors for waste incineration and recycling are collected from the research and literature. Raadal, Modahl, and Lyng (2009) estimated the specific GHG emissions related to treatment or recycling processes for the average Norwegian waste management system, and the emission factor for most fractions are collected from this report and given in Table 3.6. HWEEE was not included in this study, and this emission factor was therefore retrieved from Callewaert (2017a), a master thesis analyzing the urban waste management company ROAF. The emission factor for incinerating textiles was also collected from this thesis. In addition, the emission factor related to insect production (material recycling of food waste) was retrieved from Salomone et al. (2016).

Table 3.6: Specific emissions caused by incineration and material recycling of waste fractions found in this analysis.

Fraction	**Recycled**	**Incinerated**
	kg CO_2-eq/kg waste	
P&C	-	0.024
Plastics	-	2.841
Food waste	0.0175	0.031
Textiles	-	0.145
Glass	-	0.025
HWEEE	-	1.428
Garden Waste	-	0.031
Other combustible	-	0.508
Wood	-	0.012
Metals	0.051	0.019

3.3.3.2 Avoided Emissions

A waste management system can contribute to avoided emissions in several manners. With avoided emissions, it is in this study referred to the net emission reduction that the waste management system provides for by replacing emitting energy generation, fuel combustion or production processes in other systems. This can be the energy generation a waste-to-energy incinerator replaces, the avoided raw material extraction when material recycling waste, or the fossil fuel biogas production replaces.

The calculated avoided emissions are based on the material flow layer of the model and specific emission factors given in Table 3.7. For recycling, the savings are found by multiplying the total amounts of the food waste fraction recycled with a negative emission factor corresponding to the production of the material/product it replaces. In the case of insect production it is assumed that the dried larvae replaces soy fish meal, while the bioresidues from both biogas production and insect production is assumed to replace inorganic fertilizer. Furthermore, the energy produced through AD and WtE incineration is attributed a negative emissions for the avoided fossil energy consumption. E.g., it is assumed that the biogas fuel produced replaces fossil fuel (diesel) and therefore *saves* emissions related to the combustion of diesel fuels. The combustion of biogas produced from organic sources is considered climate neutral in the UNFCCC, since it does not add fossil carbon to the global carbon cycle due to its origin (Zanchi, Pena, and Bird 2011). Using this assumption, the biogas produced can be considered a net carbon saving, corresponding to the emissions of the fossil fuels it replaces.

Salomone et al. (2016) assumes a nearly 1:1 relation in tons larvae proteins produced to soy meal proteins replaced. In this analysis we therefore say that the protein outcome replaces an equal amount of soy bean meal. The avoided emissions from insect production is extracted from Pelletier (2008).

Table 3.7: Emissions avoided through waste treatment.

Avoided emissions	
	kg CO_2-eq/kg waste
Insect Production	-0.0469
Insect residues	-0.1757
Biogas Production	-3.17
Bioresidues	-3.06
Incineration	-0.283
Metal Recycling	-2.589

3.3.4 Cost Layer

The economic factors are vital for a system solution to be considered by a SWM company. A system that is not feasible, is neither a sustainable system and it is therefore natural to include a cost layer in this analysis.

The economic results of this system is calculated manually, since the model applied currently does not have a cost layer implemented. In this section, the theory behind the cost layer is introduced.

3.3.4.1 Capital and Variable Costs

The costs in a SWM system is characterized by two types of cost, fixed and variable costs. Fixed costs, or *capital* costs, is a fixed amount invested in a facility, incinerator, a collection truck or land e.g (Merrild and Christensen 2010). Included in the fixed costs are also the costs related to the design, planning and construction activities.

The variable costs are those that appear throughout the lifetime of a system. This can e.g. be the salary of the employees, unforeseen costs due to damaged components or regular maintenance costs.

3.3.4.2 Revenues

The SWM system earns revenues on both service provided and products or resources that are sold to a market. The revenues of a municipal SWM system is in Norway regulated by law to balance in the long term, i.e. there should not be a profit or adverse balance.

3.3.4.3 Annualization

The capital costs appear in the first years of a project. For instance, the planning and construction phase of a recycling facility requires large investment costs in these phases, while it only has costs related to operational and maintenance activities in the latter years. Therefore an *annualization* is done to the investment costs, to divide the total investment on the lifetime of a facility or tool.

3.3.4.4 Inflation

The data supplied is based on values and calculations for different years and this means the actual values differ due to inflation. To make the values comparable, a correctional factor is applied using the consumer price index. All costs and revenues are converted to April 2018 value with the use of Statistics Norway's (SSB) consumer price index calculator.

3.3.4.5 Discounting

Furthermore, to make a correct comparison of the costs and revenues of a SWM system, the prices should be changed to present value through *discounting*. This is done by calculating the *net present value* (NPV), with the use of Equation 3.15. Due to a limited scope and time to acquire data for this analysis, the annual costs are found for 2030 in 2018 value. The annual costs for the years between 2017 and 2030 is found by linear interpolation. This is applied in Equation 3.15 to find the correct net present value for the future scenarios. Here, the AC_t is the annual costs in year t in 2018 value. r is the discount rate, and this is set to 3.47 % in collaboration with BIR.

$$NPV = \sum_{t=0}^{t=T} \frac{AC_t}{(1+r)^t} \tag{3.15}$$

3.3.4.6 Data

This economical analysis provides annual costs containing fixed costs (distributed over estimated lifetime of the facilities/system components) and variable. For some costs, it was only possible to retrieved one sum comprising of both fixed and variable costs. These are also included in the analysis, and refered to as *unspecified* costs.

The main data supplied is based on BIR Privat's accounting figures. Where BIR could not supply cost data for the inventory necessary, estimated costs from reports regarding similar SWM systems and facilities are used as the the primary data source. The estimated costs are mainly those concerning facilities that BIR does not have today. A report from 2014 (Igesund et al. 2014) is utilized for estimating the optical sorting and pre-treatment facility costs. For the insect rearing plant, a gate fee is estimated by Invertapro.

Since BIR Privat is a branch company of the BIR corporation, they outsource all their services to other branch companies. This means that they in reality have no investment costs in the current (reference) system. Due to the legislation, it is assumed that what BIR pays is the full cost coverage for the facility utilized. This means that for some processes, the fixed costs and variable costs are embedded in one mixed sum in BIR Privat's annual accounts. With this in mind, assumptions of the costs related to the optical sorting plant are based on an expected increase in the inhabitant fee estimated by Trondheim municipality in an assessment from 2011 (ibid.). In addition, a "gate fee" based on Ecopro's pricelist for treating food waste is included in the Biogas scenarios. Similarly, Invertapro estimates a cost for BIR of 250 -500 NOK/ton delivered FW (personal communication with A.S. Ringheim in Invertapro). For the insect scenario a gate fee of 500 NOK/ton is therefore applied.

The collection fee is not included as revenues in the current cost analysis because the BIR municipal waste management system is of full cost coverage, i.e. there should be no surplus or deficit in the accounts in the long term. The findings in this analysis could however point to how the inhabitants' fee would change if the modelled system changes are implemented. On the other hand, the incinerator in the system is owned by a branch company that sells energy to the market. The amount household waste burned is directly affected by the changes implemented in the scenarios and is included as a revenue.

3.4 Case study description

The model presented previously and utilized for this report is put to use in a case study of the company BIR Privat AS, operating in the western region of Norway. This section describes the system analysed, scenarios, collected data and assumptions.

3.4.1 BIR

BIR AS (Bergensområdets interkommunale renovasjonsselskap) is one of Norway's largest waste management companies. BIR AS is owned by nine municipalities (Askøy, Bergen, Fusa, Kvam, Os, Osterøy, Samnanger, Sund and Vaksdal) and has its core business in Bergen, where the majority of inhabitants resides. The BIR corporation operate in all parts of the waste management supply chain, except for material treatment. The subsidiary company BIR Privat AS is assigned the statutory waste collection responsibility for the roughly 360 000 inhabitants in the nine municipalities. Some of the tasks within this mandate are assigned to different branch companies in the corporation. In this report two separate system models are analysed, and these are displayed in Figure 3.3 and 3.4.

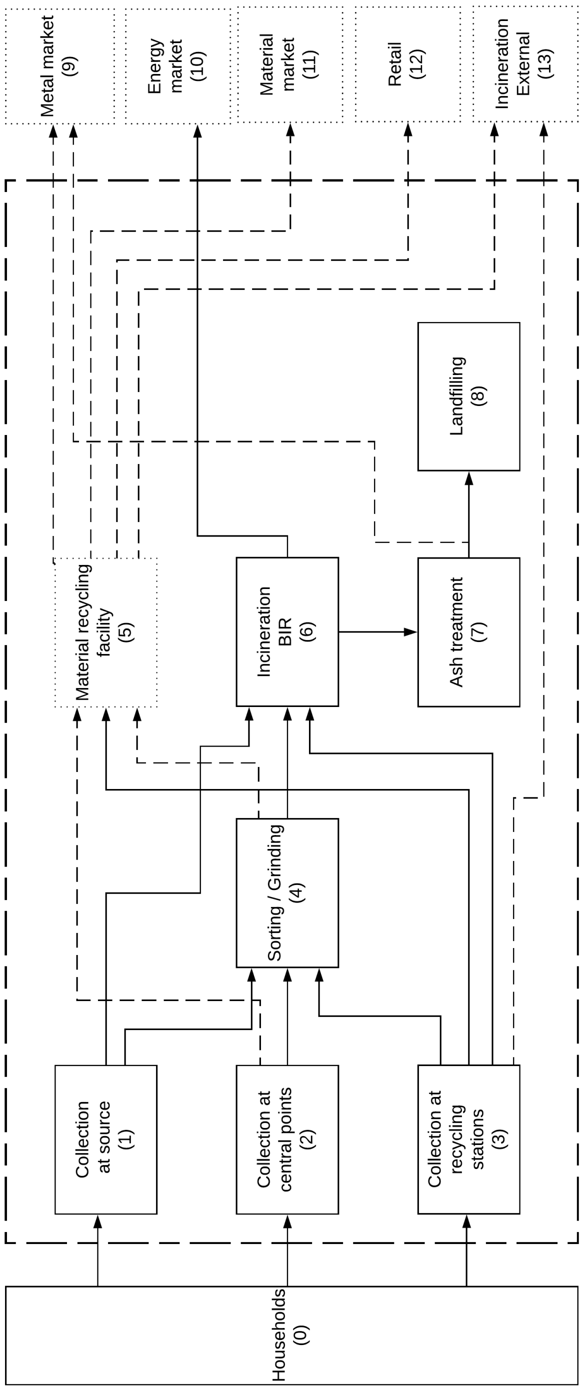

Figure 3.3: System A: Flow chart representation of the complete BIR system.

In Figure 3.3 and 3.4 the system processes are represented by boxes. The arrows connecting the processes illustrates the waste flow pathways, but also represents transport. The stippled arrows represents transport that is not included in the analysis. Detailed description of the system flows are listed in Table B.4 and 3.4 in Appendix B.

The system boundaries in system A comprehend all collection and treatment processes present in BIR's urban waste management system. The markets linked to the waste system are left outside of the system boundaries, as well as the generation itself. The data provided by BIR only comprises of what BIR collects, treats and transmits. The *process* of waste generation in the households is not part of the objective of this study (e.g. regarding prevention in the waste hierarchy), and is therefore left outside of the system boundaries. System A encompass six waste types collected by BIR; residual waste (RW), Paper and cardboard (P&C), plastics, Glass and metals (G&M), textiles and wood. These are the waste types of significant size and that are relevant for the objective of the analysis. The process with a stippled box (process (5)) is the only process that the BIR group does not operate. Process (13) represents an external incinerator to account for the losses in the downstream processes and is the final destination for the major part of the wood fraction.

The main objective of this project is related to implementing food waste (FW) sorting, and the main analysis concerning the alternatives scenarios for are therefore modelled in a limited version of the system as shown in Figure 3.4. The processes included within the system boundaries here are only those of which the RW and FW are part of. The analysis of System B is executed to provide a transparent analysis targeted at food waste solutions.

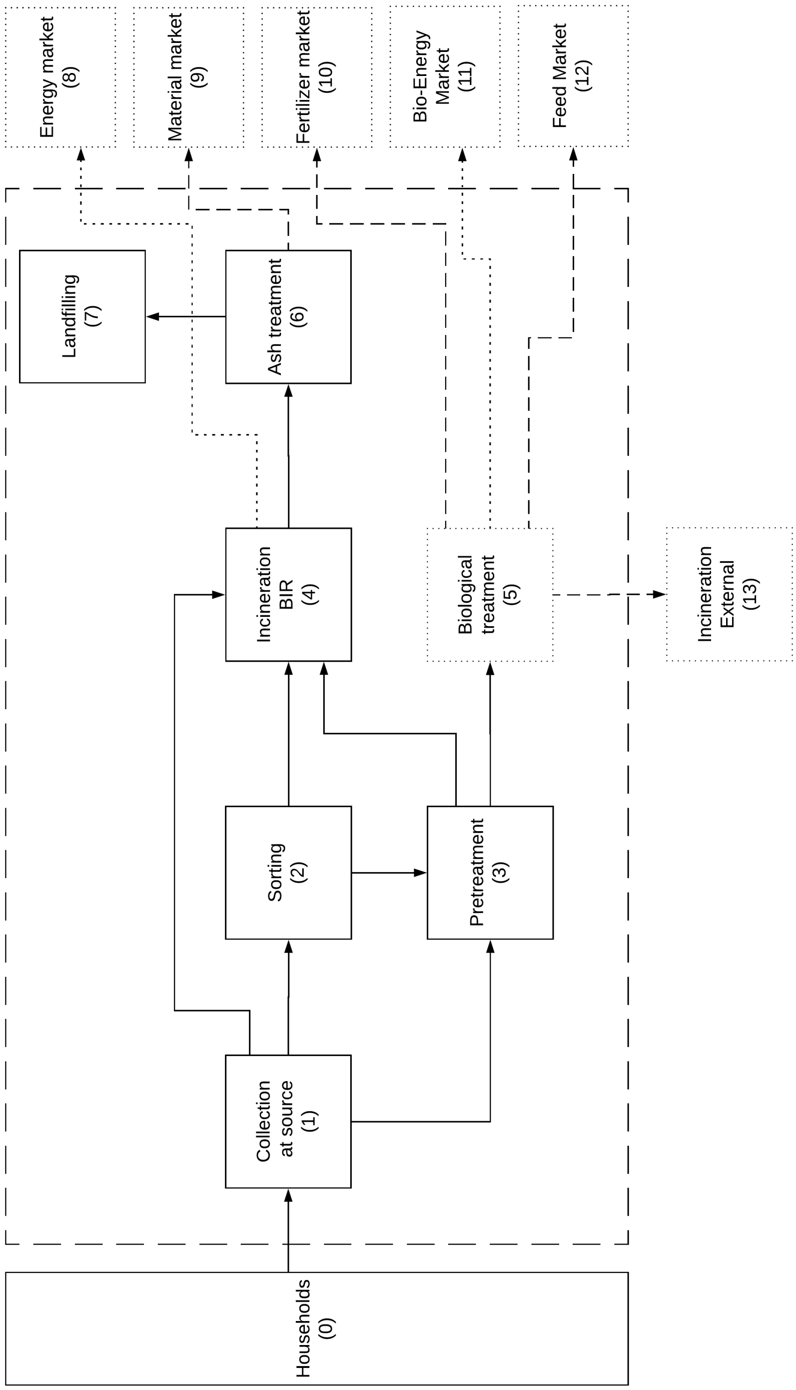

Figure 3.4: System B: Flow chart representation of the simplified BIR system.

3.4.2 Scenario Descriptions

With a basis in the system boundaries described in the previous section and illustrated in Figure 3.3 and 3.4, Table 3.8 summarizes the scenarios that are developed and analyzed in this study. This section provides a detailed description of each scenario.

Table 3.8: Scenarios modelled and analyzed in this report.

	2017	**2022**	**2030**
System A	Reference Scenario	Reference Scenario	Reference Scneario
System B	Reference Scenario	Reference Scenario Biogas Oslo Biogas Bergen	Reference Scenario Biogas Oslo Biogas Bergen Insect Scenario

The BIR household solid waste management (SWM) system as it was in 2017 is used as the base case in this analysis. Scenarios are developed and modelled to assess the environmental performance. The scenarios chosen here are meant to represent possible changes made in the system with regards to food waste treatment alternatives. The scenarios are bound for 2030, when the most important European material recycling targets are due. In addition, it is implemented intermediate effects and results for 2022 for the solutions where this is relevant.

The scenarios vary in terms of treatment solutions, while the collection method is modelled equally for all. It is assumed that source segregation for food waste is introduced. BIR has not currently made a decision for the execution of food waste sorting, but a likely option for the collection method is chosen for all modelled scenarios. The collection method is drawn up in collaboration with Barbro Relling and Thoralf Igesund at BIR. Factors such as ability to navigate, available storage area, technology maturity and economics influence the choice of differentiating the collection method between dwelling types. It was decided to differentiate between single house dwellings that today have individual collection bins, and other solutions. It is assumed that those who have their own bins today also have available space for a new separate bin for food waste. Those who are part of a collective solution such as container bin, underground or vacuum collection in a neighbourhood sort the food waste in special bags (typically *green bags*) that are put in the residual bin. The bags are separated from the RW in an optical sorting facility established in Rådal, Bergen. The remaining residual waste is sent for local incineration in all scenarios. This solution affects the dimension of a optical sorting facility, the collection rate and the transport logistics slightly.

The division of food waste collection technology is based on today's dwelling composition and is displayed in Table 3.9. It is assumed that those with individual solutions are those with bins of 60 l, 140 l or 240 l (Small bins).

Table 3.9: Distribution of subscribers per food waste collection solution.

Technology	**Percentage %**
Separate bin	58.23
Bag in RW bin	41.77

The reference scenario and selected alternative food waste scenarios are presented in the sections below.

3.4.2.1 The Reference Scenario

The reference scenario is based on BIR's reported waste flows and energy use for 2017. This scenario is projected to 2022 and 2030, without implementing any measures or process modifications. The only changes in the reference system are the increasing population and waste amounts.

Two versions of the reference scenario is modelled. The purpose of developing two reference scenarios is to maintain the extensiveness of the analysis (system A), and at the same time obtain evident results that are relevant for the objective of the project (system B). Firstly, a holistic system model as developed in the project work in the autumn 2017 is refined and re-analysed (System A). Secondly, a reference scenario only including the residual waste and food waste is developed to target the main objective related to alternative food waste treatments (System B). The reference scenario system B is illustrated in Figure A.1 in Appendix A.

The population estimations are based on the intermediate scenario[3] defined by Norwegian national statistics, Statistics Norway (SSB) (2017c), and are displayed in Table 3.10. SSB carried out a population projection in 2016 for 2040 for each municipality in Norway. From this, the population for each municipality is found by linear interpolation for 2017, 2022 and 2030. This is a simplified estimation, but assumed to be good enough for this analysis. A detailed overview of the population estimation for each of the nine BIR municipalities is displayed in Appendix A, Table A.1.

[3]Scenario "MMMM" (the main alternative), represents a medium development in the four areas of measurement; fertility, duration of life, domestic movement and migration(Leknes, Syse, and Tønnessen 2016).

Table 3.10: A projection of the population within BIR municipalities per January 1st (SSB 2017b).

	2017	**2022**	**2030**
Population	358 638	375 629	402 814

Table 3.11 shows the estimated overall growth in household waste quantities. These numbers are based on a report from 2015 by the Norwegian waste management and recycling association (Avfall Norge) describing the projected waste growth in Norway's three largest urban waste systems (Skogesal 2015). Since some fractions are left out of this analysis, the waste amount per inhabitant is adjusted downwards for 2017. The upscaling for 2022 and 2030 is done by assuming a linear growth for all fractions, so that the waste amounts included in the analysis is scaled at 81.9 % of the total projected household waste amount for all years.

Table 3.11: Estimated waste amounts within the BIR municipalities.

Waste Generation	**2017**	**2022**	**2030**
Kg per capita	363	383	411
Total tons	130 072	143 866	165 557

The population and waste amount projections presented in this section will be used in all scenarios. The increase in waste generated is assumed evenly distributed between the included waste types.

3.4.2.2 Biogas Oslo Scenario

A likely treatment for the food waste is anaerobic digestion (AD), where the organic matter is degraded and processed to become methane fuel.

One alternative for a biogas scenario is to transport food waste (FW) to the eastern region of Norway. This is a solution of current interest, mainly because the western region lacks a market for the bioresidues that is a by-product of AD as described in Section 2.3.1. In this scenario, the food waste is segregated and pre-treated to reduce the volume and mass at a facility in Bergen, before it is transported to be processed at Romerike biogas plant where the food waste from the Oslo municipality also is treated(EGE 2018).

This scenario assumes that pre-treated organic waste is transported to an anaerobic digestion plant outside Oslo. A travelling distance of 500 km between the BIR pre-treatment plant and Romerike biogas plant (retrieved from Google Maps) is accounted for in the analysis.

3.4.2.3 Biogas Bergen Scenario

The municipality of Bergen has a biogas plant that treats sewage sludge and produces upgraded methane as a fuel for the local buses.

An alternative scenario is that BIR sends the food waste to the local biogas plant. As for the biogas Oslo scenario, this solution also assumes that a pre-treatment facility is established in Bergen. This is necessary for the organic household waste to meet the quality and consistency requirements for the input to the existing biogas plant. The produced biogas is assumed used as a fuel on local buses, while the bioresidues are assumed transported to the eastern part of Norway and used as fertilizer.

3.4.2.4 Insect Scenario

A less conventional alternative is to use the food waste as nutrient input to insect rearing for protein production. Through personal contact with Alexander Solstad Ringheim (Invertapro), a planned insect production facility forms the basis of this scenario, supplied with specific data from the literature.

The insect scenario is based on the plans of Invertapro located in Voss, approximately 110 km away from Bergen. This is a research start-up that is developing a technology for large scale insect production. For now, the development is in the phase of establishing a pilot plant, and a scenario of full scale insect production is therefore modelled for 2030. The full scale plant will be able to receive 30 000 tons of organic waste and produce roughly 6600 tons of proteins (Smetana et al. 2016).

BIR is prepared to establish a pre-treatment facility, regardless of the choice of food waste treatment. Therefore, this scenario involves a pre-treatment facility in Bergen, similarly as in the other scenarios. The insect rearing plant located in Voss, is assumed to receive pre-treated food waste suitable for insect feed. The process at the insect rearing plant is based on the technology described in a study conducted by Salomone et al. (2016) in an existing pilot plant in Southern Italy and the current analysis utilizes specific data from this case study.

In this scenario it is assumed that there are three output products that can be sold: oil, protein and organic residues. The protein is used as feed in the aquaculture industry and pet feed (which is in accordance with the current legislation), while the residue product (excrements and food waste remains) is in the literature found to have adequate properties as fertilizer (Salomone et al. 2016; Thévenot et al. 2018).

3.4.3 System Description

The system structure vary for the different scenarios. In this section, all processes and technologies present in the scenarios are described.

3.4.3.1 Collection At The Household Address

BIR Privat AS collects waste from private households in several manners. A main target is that all customers are offered the same level of service at any address, independent of the dwelling type and collection technology (BIR Privat AS 2016). Today, BIR Privat AS offers collection of Paper and cardboard (P&C) in addition to residual waste on regular routes at all household addresses. Since 2008 BIR have also offered plastic collection in separate bags for single dwellings, and in bins for shared solutions. Not all housing cooperatives have area available and therefore do not have plastic collection, but in this study the collected plastic is assumed shared among all inhabitants due to lack of data for the exact collection addresses.

The most common solution for home collection today is the traditional bins with wheels, from 60 litres capacity to larger bins for housing cooperatives that holds 660 litres. In newer developed residential areas, modern underground containers are the preferred solution. In addition, pneumatic collection technology is making it's entry in the BIR municipalities. In central parts of Bergen, a central vacuum system is installed and put to use. The waste is collected in tanks below the ground. A control centre monitor the percentage of filling in the tanks and the waste is in turn transported in underground pipes to the central and into compressing containers. These containers are regularly transported to the incineration plant or sorting facility.

A pneumatic technology is developed by the branch company BIR Bossug AS and adapted to fit the needs of housing cooperatives' waste collection. The solutions offered for e.g. apartment blocks are mainly mobile vacuum systems, as this technology is more appropriate for smaller installations (Nilsson 2010), though BIR also offers stationary vacuum systems.

The collection technologies implemented in the model are those listed in Table 3.12. The aggregation of 17 different collection solutions is done based on the cars that collect the waste. The cars are described in Table A.2 in Appendix A. Due to lack of exact number of customers per collection technology, the distribution is estimated with the use of average inhabitants per housing unit for some cooperative housing areas. This results in an overestimation of 3.69 % housing units in the analysed area, but it is considered good enough for this analysis. The current distribution shows an overwhelming share being the traditional bins, and this dominates the results. As a simplification it is assumed that this composition is constant until 2030, but in reality BIR targets an increased implementation of modern technologies (underground and vacuum technologies) (BIR Privat AS 2016).

Table 3.12: The distribution of collection technologies offered by BIR Privat AS.

Technology	**%**
Small Bins (60-240 l)	58.23
Large Bins (400-660 l)	22.52
Containers	9.42
Underground	1.03
Mobile Vacuum	5.60
Stationary Vacuum	3.20

3.4.3.2 Central Collection Points

Glass and metals (G&M) are collected in containers located on central points. The ideal density of these collection points are about 1000 inhabitants per point (Dalen et al. 2017). These are usually located on parking lots. Collection towers for textiles, distributed by ideal organizations (mainly Fretex and UFF) are often co-located at BIR's collection points.

3.4.3.3 Recycling Stations

BIR operates eleven recycling stations in nine municipalities. Three of these are within Bergen municipality. In addition they have a mobile recycling station, i.e. a truck with fixed, longer stops in central Bergen. Some of the fractions are free for residents to deliver, while for other fractions the inhabitants pay by volume. The recycling stations received 46 % of the generated household waste in 2017 (Relling and Grevskott 2017).

The recycling stations also receive some commercial waste. Based on the number of customers, BIR estimates the total amounts to be approximately 5-10 %, but with some variations between the fractions sorted for. In this model however, we subtract an overall 7.5 w% from the reported amounts delivered at the RS to estimate the household amounts, due to a lack of basis to assume more exact distribution.

3.4.3.4 Sorting Facility

Today, BIR AS operates a paper sorting facility in Bergen. Specific energy data for this facility was hard to acquire, since there are several activities at the same address. However, it is assumed that most of the reported energy used in the sorting facility is attributed to the paper sorting, since this is the main activity and major plant on the site.

Furthermore, the chosen collection method for the organic waste in the alternative scenarios implies specific bags delivered in the same bin as the residual waste. This requires an optic sorting facility where the bags with organic waste are recognized and separated from the residual waste (RW). In the scenarios this is relevant for, it is assumed that such a optical sorting facility is placed next to BIR's incineration plant. It is assumed that some RW goes with the flow to the pre-treament facility, and some food waste (FW) goes to incineration. In a study performed by Avfall Norge (Syversen and Schefte 2007), large variations in the sorting performance for several sorting facilities was found, depending on camera adjustments and quality follow-up. When the facility operates under correct conditions, the suppliers can guarantee 95 % sorting rate. However, a picking analysis at Interkommunalt Renovasjonsselskap i Salten (IRIS) composting facility showed a variation of 2-23 % sorting error for received food waste from four different optical sorting facilities. The found average of 11.4 % bags incorrectly sent to biological treatment is applied in the scenarios. Similarly it is assumed that 22 % of the collected organic waste is sent to incineration, due to sorting errors and torn and damaged bags (Callewaert 2017a).

3.4.3.5 Material recycling facility

Several of BIR's collected fractions are material recycled. This segment is not operated by BIR[4], but sold to external companies. Therefore, an estimation of the average material recovery is used, based on Tchobanoglous and Kreith (2002) as recited by Christensen and Bilitewski (2010). The assumed recovery rate for different materials are listed in Table 3.13.

[4] when considering treatment of organic waste in a separate process

Table 3.13: Typical recovery rate in MRFs as given by Christensen and Bilitewski (2010).

Material	Recovery Rate [%]
Mixed P&C	90
Mixed Plastics	90
Mixed Glass	90
Tin cans	90
Aluminum cans	90

3.4.3.6 Pre-treatment for food waste

In all scenarios with food waste sorting, it is assumed that a pre-treatment process is necessary. It is therefore assumed that BIR establishes a pre-treatment plant in Bergen. The purpose of such a plant is to prepare the wet-organic waste for further biological processing. The objective is to obtain a desired consistency, remove undesirable elements and/or sterilize the substrate (Khoshnevisan et al. 2018).

The pre-treatment process is a critical phase in the chain of biological treatment processes. The performance of a biogas plant is highly dependent on the input quality. Research shows that current pre-treatment technologies causes an average of 24 % reject from the Norwegian biogas plants (Marthinsen 2017). BIR is therefore willing to invest in robust technology that is widely tested and acknowledged. In Northern Europe, Sweden is perhaps the country with the longest experience with separation and treatment of food waste in biogas plants (Igesund et al. 2014). Cellwood Machinery is a Swedish company that offers reject removal machines for organic waste streams. As for several other pre-treatment technologies (ibid.), the process is made up of several sub-processes, but in this simplified model, the process is modelled as one component with a total input/ output- and dry matter ratio. The technology described in Cellwood's websites (*Bioenergy* 2018) is found to be of similar entity[5] to the system described by Naroznova et al. (2016) and included in a comparative LCA study by Khoshnevisan et al. (2018). The "Biopulp technology" is based on Gemidan Ecogi AS (ibid.) and includes a pulper and separator similar to those presented on Cellwood's websites. Furthermore, the

[5] Cellwood offers further components than those deemed necessary for pre-treatment of FW. The two final components of the Cellwood technology are not found in these studies, but Cellwood emphasizes that each component can be added separately to an existing system.

following steps are dewatering, reject washing and storage. The output of the process is a raw pulp of organic compounds, reject and water. The water is assumed recirculated and the reject is incinerated in BIR's incinerator.

Since Cellwood does not supply accurate data on performance, the data used for this process is based on the case study by Naroznova et al. (2016) and applied in a study by (Khoshnevisan et al. 2018). The data applied is listed in Table 3.14. Møller et al. (2012) reports of an energy use in a pre-treatment facility in of 15.8 kWh/ton FW in Norway. This is a simpler technology with fewer stages, and hence the lower energy use. Khoshnevisan et al. (2018) found that the Biopulp technology used more energy than the compared technologies, but resulted in a better separation of reject and substrate. BIR has expressed a desire to acquire advanced and reliable technology for the pre-treatment, since it is a crucial step in biogas treatment.

Table 3.14: Specific data per ton MSW input for a pre-treatment facility (Naroznova et al. 2016; Khoshnevisan et al. 2018).

Data	
Energy [kWh]	26.4
Input TS MSW [%]	33.7
Output TS MSW [%]	14

It is acknowledged that this process always will need to vary to fit the next treatment process. These variations are considered beyond the scope of this report, and therefore a simplification is done where the same pre-treatment technology is adopted for all scenarios. It is in any case a desire from BIR to establish a pre-treatment plant that can accommodate more than one treatment solution for future flexibility and investment security.

3.4.3.7 Biological treatment for food waste

As introduced in Section 2.4, there exists several alternative technologies for biological treatment of organic waste. In BIR's current system, only garden waste is treated biologically in an aerobic process to produce compost. FW is source segregated in the modelled scenarios, in contrast to the current system where FW is collected with the RW. The introduction of FW sorting must also mean the introduction of a treatment of the fraction, and options for biological treatment are added to the system. In this analysis AD and insect rearing represents the alternatives for biological treatment of FW.

Biogas plant
Two scenarios include treatment in a biogas production facility. These are in reality different plants (in Oslo and Bergen), but in this analysis they are modelled similarly. The Romerike Biogas plant (RBA), supplies an energy use for the complete facility - including the process of pre-treatment which is co-located. The pre-treated FW is assumed entering directly into the thermal hydrolysis process, the main process of the biogas plant, together with FW from the local pre-treatment. The applied energy assumptions in this assessment are based on the upper values of the literature. The measured energy use in RBA is significantly larger than what is given in the literature studied. van Zanten et al. (2015) found an AD facility (excluding pre-treatment) to be using 49.2 kWh per ton treated organic mass (DM content of 10 %), while Bernstad and la Cour Jansen (2011) found 50 - 75 kWh/ton FW. It is however reasons to believe the included energy consumption in these studies are limited to the operating machinery, while the facility in Oslo reports of the total energy consumption, from offices to digestion machines, and therefore the literature is placed emphasis on to make it comparable to the literature-based modelling of the insect facility.

Similarly, the reject amounts reported from RBA includes reject from the pre-treatment plant. It is assumed that a reject of 17.6 % includes reject from the pre-treatment. Therefore it is applied a reject rate of 7.5 % based on Ecopro's statements instead (Bjørndal 2013). This however, turns out to be close to the findings of (Møller et al. 2012) for only the biogas production and is therefore assumed a valid assumption. Remember that there is modelled a reject rate in the pre-treatment facility also. Table 3.15 shows the AD facility input data applied in this analysis.

Table 3.15: Specific data per ton MSW input for a biogas plant (ibid.).

Data	
Energy [kWh]	75
Input TS MSW [%]	14
Biogas output	57.75
Bioresidues output	34.75
Reject/loss [%]	7.5

Insect rearing plant
The insect rearing plant is based on the plans of Invertapro, based in Voss, Hordaland, Norway and supplied with specific data from the literature (Salomone et

al. 2016). The insect production is based on the breeding of the Black Soldier Fly (BSF), a highly resilient insect, with an adequate behaviour and appropriate properties for protein production (Čičková et al. 2015) .

Invertapro is planning a stepwise feeding process over five weeks, where insects are fed with organic waste matter. In this specific analysis, it is assumed that the waste stream originates from BIR's household waste. The pre-treated substrate is fed to the insects (Black Soldier Fly larvaes) as nutrition. After five weeks 99 % of the larvaes are harvested, while the last 1 % becomes flies and lies eggs. The organic input matter together with the harvested larvaes, after drying and milling, results in four outputs as displayed in Table 3.16.

Table 3.16: Distribution of output from BSF rearing (ibid.).

Output	[%]
Water	62.35
Proteins	15.8
Oil (fat)	13.17
Residues [kg/ton FW]	334.6

Assumptions of energy use and greenhouse gas emissions are based on a study conducted by Salomone et al. (ibid.).

3.4.4 Data and Assumptions

The data collection carried out in this project is to a large degree based on available statistics and reports from BIR Privat AS. While the scenarios are partly based on literature, data of the waste stream masses and compositions are supplied by BIR's internal picking analyses and reported quantities. For the information provided by BIR Privat AS to fit the scope and model of this analysis, some adjustments and assumptions are made for this data. These are described in the following sections.

3.4.4.1 Waste Types

In this analysis, the emphasis is put on the waste types influenced by the measures introduced in the scenarios. The main work of this project is centred around the residual waste (RW) and food waste (FW) types. To achieve transparent and specific results, System B models all scenarios for only these two waste types. Furthermore, these scenarios only includes collection at source, and the RW from recycling stations is not included. In addition, a holistic assessment of the reference scenario is performed to provide general and comparative results (System A). For the holistic analysis, a total of six[6] waste types in BIR's waste system is accounted for and all are listed in Table 3.17.

Table 3.17: Complete list of waste types included in the analysis.

Collection Method	Waste types
Home and at collection points	Residual waste*
	Paper and cardboard (P&C)
	Plastics
	Glass and metals (G&M)
	Textiles
	Food waste*
Recycling stations (only)	Wood
*Waste types included in System B.	

Waste types at Recycling Stations
The recycling station (RS) is included in the System A analysis. BIR reports of 12 waste types sorted at the recycling stations (RS), in addition to several categories of hazardous waste. At the RS it is segregated between bulky and small units residual waste. This is included in the residual waste type, but modelled with two different pathways from the RS since the bulky waste is grinded before incineration. Garden waste types are left out of the model, due to little relevance and recycling potential other than the existing composting process. Hazardous waste is treated in a system separate from the rest of BIR's operation, and not considered in this analysis. Cables are excluded, due to being a minor fraction with little influence in the big picture. For similar reasons, plaster, tires and LCM are not included in the model. Wood is considered a significant flow with considerable recycling potential in the waste handling system and is included in the model. Currently, BIR's incinerator utilizes the amounts they need to achieve the desired LHV, while the major part of this fraction is sent for incineration in Sweden.

The Composition of Residual Waste (RW)
A picking analysis is a method to characterize the composition of a waste flow. Such an analysis is performed to understand how a recycling scheme is working,

[6]Food waste (FW) is not included in the System A analysis, since it is not collected in the current BIR system.

or where the potential for a new recycling scheme lies (Lagerkvist, Ecke, and Christensen 2010). A picking analysis is carried out by sampling large batches of waste from a specific waste type. The masses are spread out on large tables where fractions and subfractions of materials are sorted and weighed. These fractions and subfractions are defined by the waste management company. This analysis is a costly method as it is time consuming and labour intensive. It is however a useful tool to map a waste flow; trends, potential and problems.

The RW composition data used for this case study is mainly based on a picking analysis of collected household waste done by BIR Privat AS in the autumn of 2017. The picking analysis from 2017 was performed with the use of the Norwegian waste management and recycling association, Avfall Norge's guide for picking analysis of household waste (Syversen, Bjørnerud, et al. 2015). Avfall Norge recommends a sample of at least 3000 kg to sufficiently reduce uncertainty in the analysis, and BIR therefore analysed 3400 kg during this analysis. The samples were gathered from incoming trucks separated by dwelling type.

BIR Privat AS identified 10 fractions in their RW picking analysis of collected waste. Minor adjustments are implemented to fit the remaining parts of the system and the weighted composition used in this case study is listed in Table 3.18 and defined in Table B.1 in appendix B. By *weighted composition* it is meant that the results are weighted by the amounts collected and analyzed from each dwelling type as well as the dwelling structure in the area where BIR collects household waste. In the model, the weighted composition of the residual waste is applied to all the collection technologies, all though it is acknowledged that this is a source of error.

Table 3.18: The composition of the residual waste collected from households by BIR.

Name	%
P&C	7.1
Plastics	12.9
Glass	3.0
Food waste	40.7
Garden waste	4.2
Textiles	4.6
HWEEE	1.2
Other combustibles	20.6
Other non-combustible	2.5
Metals	3.2

Because the composition of the residual waste flows delivered to recycling stations differ from those collected at routes, these are distinguished between in the model of system A. Food waste is for instance a significant fraction in the residual waste collected at home, while it is not accepted at the RS. BIR separates bulk and small units residual waste at RS. In 2014 and 2015 there was carried out picking analysis for the small unit RW type at recycling stations (RS) BIR operates. The stations analyzed are placed in different types of demographic areas, and the analysis are done at different times of the year to better represent the variation within the BIR municipalities and by season. The sets of results from the RS deviates significantly for some of the fractions, but this is expected to be due to random pick, and not differences at the station. This is also the case in picking analysis done in other areas, such as a study performed by Rasmussen and Aschjem (2017) in Grenland (RiG) in Eastern Norway. Due to large incoming bulks from the visitors, the variations are larger at RS than in the collected household residual waste. The presented numbers are therefore a sum of six picking analysis done at four different RS. Because of the variation in fractions sorted for, some aggregation of smaller fractions is done to fit this analysis. A final composition is found from the summarized amounts in all picking analysis carried out and shown in Table 3.19. The rightmost column shows the estimation of composition in the bulk residual waste delivered at RS. This is a rough estimation done by employee Kirsten A. Grevskott in BIR Privat AS.

Table 3.19: The composition of bulky and small units residual waste types at BIR's recycling stations.

Fraction	**%** Small units	**%** Bulk
P&C	9.56	0
Plastics	8.51	0
Garden waste	4.41	0
HWEEE	3.74	0
Textiles	10.56	0
Other combustibles	41.22	20
Other non-combustible	9.64	10
Wood*	8.50	50
Metals	4.15	20

*Fractions only found at the RS, and not included in System B.

When combining Table 3.18 and Table 3.19, the total becomes eleven fractions. Furthermore, the assumed composition of the remaining waste types are listed in Table B.2 in Appendix B.

Sorting Rate and Composition of Food Waste

Currently, BIR does not offer food waste sorting and does not have data on the inhabitants' response to such a service, i.e. the degree of sorting is not is given. Therefore, assumptions of the sorting degree of food waste in the BIR area are based on experience data from other urban SWM systems in Norway that provides this service today. The sorting degree from inhabitants is found to vary between the offered services (Syversen and Schefte 2007). Historically, the sorting degree is larger in the households with a separate bin for FW than in the households with bags in RW bin, and this was also found by Hanssen et al. (2013) when comparing findings in 29 systems with FW sorting in 2013. The assumed collected amounts for the two sorting and collection solutions are therefore different.

Renovasjon i Grenland (RiG) has a bag-sorting system for FW and plastic together with RW. Their picking analyses found a sorting degree of nearly 60 % for FW. That is, 60 % of the generated FW from the household is sorted, while 40 % ends up in the RW (Bjørnerud 2017b). Romerrike Avfallsforedling (ROAF) operates a similar system, and reports of a sorting degree of 49 % (RoAF 2016). Oslo REN found the sorting degree to be 46 % in 2017 (Hjellnes Consult 2017). BIR will need a facility in similar scale as RoAF, with the described collection and sorting scheme for food waste (Section 3.4.2). It is natural to assume a slightly lower sorting degree in the larger urban area of BIR compared to RoAF, but with Oslo with so close to RoAF in sorting degree, it was decided to model a sorting degree of 49 % in the BIR area also. The increasing public awareness may increase the sorting degree over time, but the effects are hard to estimate, and the sorting degree therefore remains constant for all scenarios in this analysis.

The incoming FW is composed of organic waste, the (plastic) bags they are sorted in and some undesirable fractions. The assumed composition of the FW collected in bags in the RW bin is based on a picking analysis from RoAF by Bjørnerud (2017a) and listed in Table B.3.

The households with a separate FW bin is assumed to collect more kg FW per inhabitant. While collection with optic sorting receives 30-55 kg per inhabitant, a separate bin can provide 60-90 kg FW per inhabitant (Syversen and Schefte 2007). The environmental report from Renovasjonsselskapet for Drammensregionen (RfD) for 2016, shows a sorting degree of 57 % for FW in separat bins, and is used as the assumed sorting degree for single dwellings in the scenarios (Renovasjonsselskapet for Drammensregionen 2018). In this analysis it is also accounted for a different composition for the different collection solutions, and this is shown in Table B.3. The composition of the FW from separate bins is based on the case of RfD, while the FW in RW bin is based on RoAF. The main difference is the received garden waste in the separate bins, which more rarely occurs in the case of bags-in-RW solutions.

3.4.4.2 Transport

The transport in the system is represented by the arrows connecting the processes in Figure 3.3 and 3.4. The specific transport distances within the system is estimated based on overall reported numbers from BIR. The kilometers travelled for collection at home and at central points is based on available data and estimated with Equation 3.16.

$$\frac{km}{trip} = \frac{\frac{km}{y}}{\frac{tons}{y}} \frac{tons}{trip} \tag{3.16}$$

The allocation of transport to waste types is a subject to uncertainty due to the limited data available. A total collection distance reported by BIR Transport AS is split between residual waste, paper and plastics based on the collection frequency. For every time paper and plastics is collected (assumed to always be done in separate trucks), the residual waste is collected four times. However, when FW sorting is introduced, the distance travelled for RW changes. In the case of FW in separate bins, it is assumed that the the FW and RW is collected at alternating weeks and the travelled distance remains the same in total for these dwelling types. In the case of green bags (GB), the total distance remains the same since FW and RW is collected together and the total amounts are not assumed to change. However, since the compacting cars must compact approximately 30 % less (Syversen and Schefte 2007), the number of loads are increased as the load capacity is reduced compared to mixed RW. The load assumptions are listed in Table C.3. This leads to a total of more tonnekilometres.

There are large distances within BIR's municipalities. Some collection points are for instance over 100 km away from the storage area. The residual waste is therefore attributed 72 % of the distance while paper and plastic is assigned 12.5 % each. This causes an insecurity due to the fact that not all addresses have plastic collection. In addition, the G&M is attributed 3 % of this travelling distance. The system specific transport assumptions are based on the 2017 reference scenario and are also adopted to the modelled scenarios.

There is a change in the collection scheme and logistics for the scenarios with FW collection. The average kilometer per trip on the collection routes is assumed constant for all scenarios. This is because the routes are assumed unchanged. The inhabitants with GB in their RW bin will not notice any difference in the collection frequency. At the same time, the inhabitants with a separate bin for food waste is offered collection every week (as before) but with alternating collection of FW and RW. The assumptions are listed in Appendix C. The assumed utilized fuel is diesel for all scenarios, since this is the current fuel in BIR Transport.

The transport laps that are not operated by the BIR corporation are excluded from the analysis. This is due to too much uncertainty related to the distances travelled and vehicles used[7]. The exception is the transport of food waste for treatment. The transport energy vary much for the FW fraction between the scenarios, and is therefore necessary to include. This means that the added, removed or re-allocated tonnekilometres of transported food waste to treatment are included even though BIR does not operate the laps themselves. The said laps are known for all scenarios.

The collection trucks carry loads of various weight depending on the waste type.

[7]The travel distances left out are not affected by the changes implemented in the scenarios, and do not represent a loss of data related to the objective of this analysis.

Ton/trip is given in Table C.3 and is based on average truck load for a given time series for RW in BIR's household waste collection. Furthermore, the densities of P&C and Plastics are used as the baseline for estimating the ton/trip for these fractions. The G&M and FW use specific sources (Dalen et al. 2017; Seldal 2014). In addition, when the FW is collected in bags (GB) together with the RW, it requires a reduced compression to avoid damaging the bags and mixing the two fractions. A typical estimation is a reduction of 30 % in the truck load compared to only RW (Syversen and Schefte 2007), and is used as the basis in this analysis. This causes the tonne-kilometres (tkm) for RW and FW collection from households to become slightly larger in the FW scenarios. An overview of the tkm for each waste type and scenario is given in Table C.5.

4 Results

In this section, the results of most significant interest are presented. The findings of System A and System B are displayed in separate sections. With respect to the objective of this analysis, the results from System B are placed emphasis on.

4.1 System A

System A is designed to represent the BIR system in a holistic fashion. This is reflected in the inclusion of six waste types and all the current existing collection processes (three) in the analysis. The main objective of including the System A results in this study is to provide a data set that can obtain overall results for BIR's current household waste and treatment system. This is not only useful for the specific System B analysis, but it can also be used for comparison of similar systems in the future. The results for System A are utilized in Section 4.2.1 to examine the effects of introducing FW treatment on BIR's total household waste collection system. The reference system A is modelled for 2017, 2022 and 2030, but the only changes made are increasing waste amounts and population in the area analysed. This means that the performance of the system remains at the same level throughout the period. The results presented here are therefore valid for all studied years.

As expected, the performance changes are not large from 2016. A representation of the system for 2016 was modelled in a project work in the fall of 2017. Both the conducted studies are based on picking analyses reported by BIR in 2016 and 2017, respectively, and should be compared carefully. The main objective of the picking analyses are to show the trends over a longer time span. More specific quantification can be under the influence of the random selection from year to year and should not be used to draw categorical conclusions. Therefore, only limited comparisons between 2016 and 2017 are made, and the emphasis is on the results for 2017.

The findings show that the analyzed BIR system has a collection efficiency of 36.9 % in 2017, when defined as in Equation 3.4[8]. This means that there is a significant potential that could be released by the population if the source segregation services were fully utilized. In other words, there is still much materials in the RW bin that could be recycled. The collection efficiency results for the collected fractions are displayed in Figure 4.1 together with the material recycling results, to illustrate

[8]When only including those fractions that are collected to be recycled in the BIR system. food waste (FW), is among the significant fractions that is currently not collected to be recycled at BIR.

the different results for the two indicators. Firstly, for the collection efficiency, the main drawbacks are the poor results in plastic sorting (17.3 %[9]) and textiles (25.2 %). There is also found much packaging glass and metal in the RW, but this is slightly reduced since 2016, and this reduction is most likely induced by the introduction of a flexible waste collection fee[10].

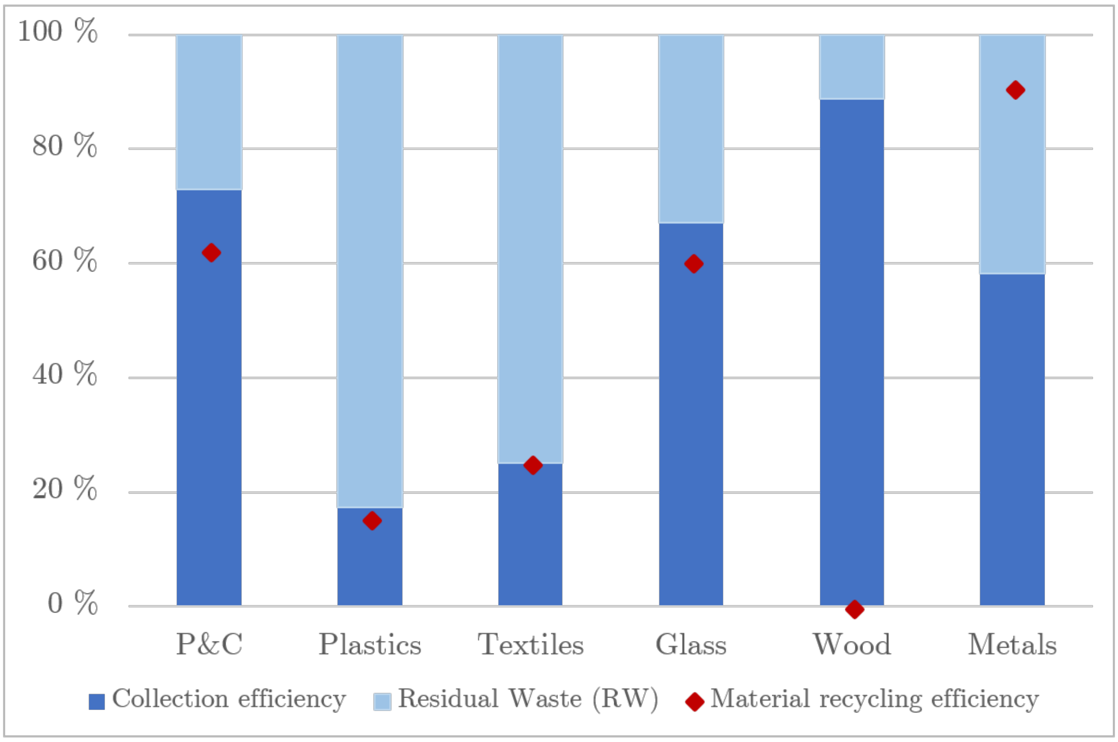

Figure 4.1: The collection and material recycling efficiency for the fractions in the reference scenario (System A).

Secondly, the material recycling efficiency (MRE) for the collected fractions is represented by black dots in Figure 4.1. The dots illustrated the losses throughout the system, from the sorting facility to the final recycling process[11]. In regards to metal sorting, the figure shows that only 58 % of the metals are sorted correctly.

[9]The picking analysis and source segregated quantities should be carefully compared for some waste types such as plastic. The plastic fraction found in the picking analysis often contains water and other attached substances which is not eliminated in BIR's reports.

[10]In 2016 BIR introduced a new tax scheme where the collection of RW has a basic fee rate that includes one collection each month. Each additional collection increases the fee. The scheme is designed to motivate increased material recycling.

[11]The exception is the wood fraction, where it is assumed that all is incinerated and nothing is material recycled.

However, as the only fraction, the material recycling is larger than the collection efficiency with its 90.8 %. This is due to the ash treatment where metals are recovered from the incinerated RW ashes.

The material recycling efficiency is found to be 20.9 %. This is lower than what BIR states (25.7 % (BIR AS 2017)), and that is due to different system boundaries applied. Some recyclable waste types received at the BIR recycling stations are not included in this analysis, and this affects the company's recycling rate. Either way, the result shows that the current BIR system is far away from achieving a material recycling of 65 %, and significant measures must be taken. Further comparative results are shown in the next section when viewing the scenarios modelled with System B.

In the reference 2017 scenario, the findings show that the RW (collected and from recycling stations) stands for the production of 145 GWh in BIR's incinerator, where most of the energy is recovered and utilized in the district heating system. In 2030 this is increased to 184 GWh. The resulting lower heating value (LHV), is found to be 6.93 MJ/kg, and this is lower than what BIR states themselves (10-11 MJ/kg). However, the LHV BIR operates with is for the overall system, and industrial waste also is included, while the LHV calculated in the model is based on the composition of the household waste.

The energy recovery rate (ERR) of the system is found to be 53.8 %. This is an increase compared to the findings of the modelled 2016 system (51.9 %), and can be attributed the increased amount of energy generated by the BIR incinerator, due to an overall increase in collected RW and a different composition of the incoming waste that in turn increases the LHV. The main contributor to the ERR indicator[12] is the feedstock energy (E_{fs}). The large energy potential in the collected waste is not fully exploited due to the system material recycling, in addition to losses in the system processes. The ERR could also be further increased if the transport energy was reduced, but this is only 6 % of the E_{in}.

4.2 System B

The main emphasis is in this report placed on the results from the assessment of System B. System B only includes the residual waste (RW) and food waste (FW) types. Different food waste treatment solutions are modelled to evaluate the resulting performance of BIR's urban municipal waste management system. Two treatment technologies in three different locations are modelled and compared to the current system where food waste sorting and collection is not a service

[12] Defined in Section 3.3.2.

provided by BIR. The results are described for each area of interest in the following subsections.

4.2.1 Materials

The material data lays the foundation for this analysis, and also shows effects of interest. The results of most interest is the changes in material recycling efficiency (MRE), an indicator defined in Section 3.3.1.2. These are displayed for System A and System B in Figure 4.2. Note that the system boundaries are defined differently for the two systems and are therefore not comparable.

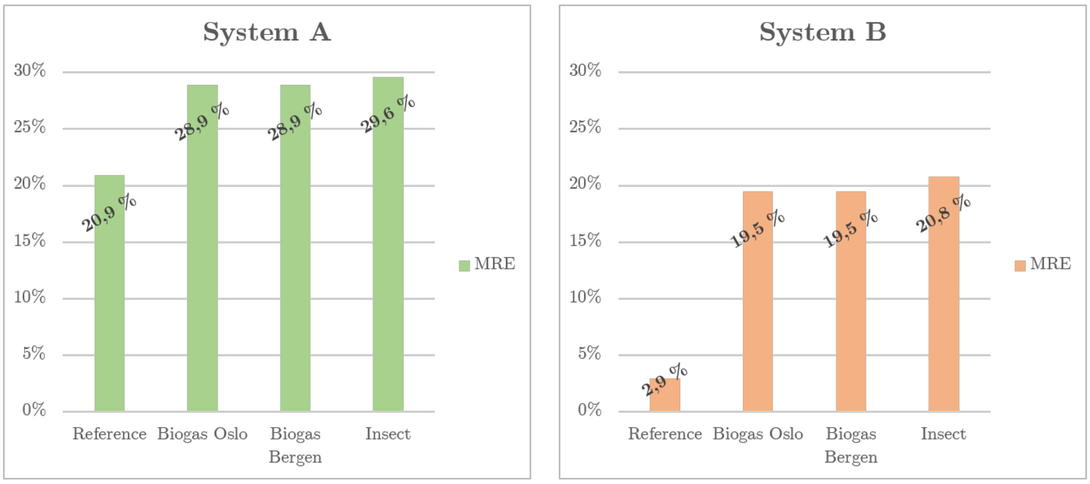

Figure 4.2: The material recycling efficiency (MRE) displayed for System A and B.

The results for System B are displayed to the right in Figure 4.2, and represents the material results for the RW and FW, making the significance of FW in the RW visible. System B has a MRE of 2.9 % in the reference scenario, due to ash treatment and metal recycling of the RW. For the scenarios with biogas treatment, the material recycling efficiency of the company is increased to 19.5 % when including both the quantities of FW that becomes biogas and fertilizer in accordance with the EU definition (Potocnik 2011). In the insect scenario, the FW collection rate is modelled similar to the biogas scenarios, but here the material recycling efficiency of the system becomes 20.9 %[13]. The positive results for the insect scenario is due to the recycling of all resources entering the biological process of insect rearing. Salomone et al. (2016) reports of residues with "physical quality (...) that could

[13]Note that this is when an average of 53.9 % of the food waste fraction is collected correctly with the food waste.

allow easy storage, packing and transport without any further transformation or stabilization process" (p.896). Therefore the assumption of no loss in this stage of insect rearing is applied, whilst in the biogas plants it it assumed that some organic waste is lost in the process of digestion and upgrading.

Food waste scenarios are not modelled in System A, but the corresponding MRE is calculated manually to see the effects of introducing food waste sorting and treatment in BIR's household waste collection system. The results are illustrated to the left in Figure 4.2. The calculations are done because System A represents the material performance indicators most correctly for BIR's complete system, and therefore supplies valuable holistic results.

As described in Section 4.1, the material recycling efficiency (MRE) for the reference System A is 20.9 %. However, when introducing FW sorting and treatment, this performance indicator increases. For the scenarios with biogas treatment, the material recycling efficiency of the company is increased to 28.9 %, while the insect scenario results in an MRE of 29.6 % for similar reasons as those listed above.

4.2.2 Energy

The energy consumption of the scenarios are larger than those of the reference system. This is due to increased transport and the added processes to the treatment system when introducing sorting of food waste. With the system boundaries applied in this analysis, the energy produced is of a higher order of magnitude than that consumed. The energy balance of the system is illustrated in separate figures due to different magnitudes of order for production and consumption.

In terms of collection transport energy, Figure 4.3 shows the results for the 2030 scenarios and the contributions from each waste type collected. The residual waste (RW) energy consumption is reduced from the reference scenario, because a share of the currently collected RW is sorted and collected as food waste (FW) in the alternative scenarios. The total transport energy is increased for all scenarios compared to the reference. Transporting the pre-treated waste to Oslo is evidently the most energy consuming solution. These results are reasonable, given that biogas treatment in Oslo means more tonnekilometers travelled, due to the large quantities of pre-treated FW transported between Bergen and Oslo (500 km). However, the transport energy consumption would be even larger if untreated FW was sent to Oslo. The biogas Bergen scenario only transports the bioresidues between Bergen and Oslo and this results in an energy consumption on this particular lap that only corresponds to 26 % of the energy consumption in the biogas Oslo scenario. Similarly, the insect scenario is modelled with the insect rearing facility at Voss, 110 km away from the pre-treatment plant in Bergen. Not surprisingly, this scenario consumes less transport energy than both Biogas Bergen and Biogas Oslo. What is more interesting of the findings is that the transport energy consumption of the insect scenario and the reference scenario is close to equal. The collection and transport of the FW to Voss only marginally surpasses the energy consumption reduction in the RW collection from the reference scenario.

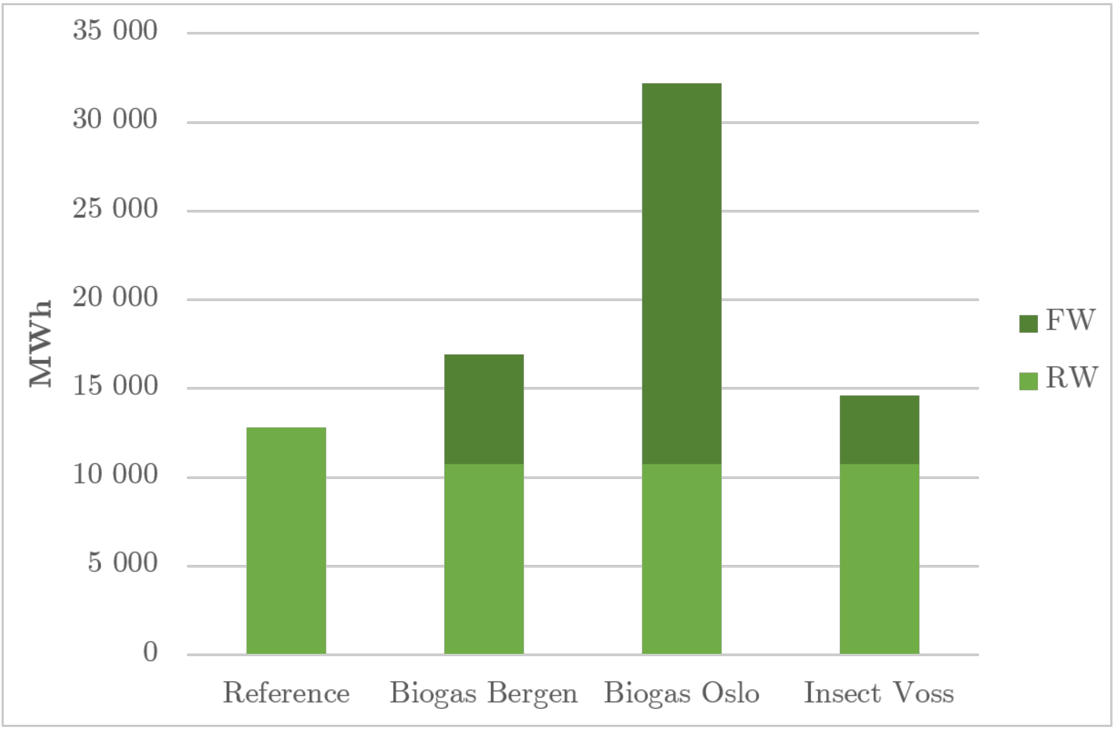

Figure 4.3: The transport energy consumption in all scenarios for food waste (FW) and residual waste (RW) in 2030.

The consumed process energy is displayed in Figure 4.4. The introduction of FW sorting means that processes are added to the system and therefore the energy consumption increases in all scenarios relative to the reference system. The biogas scenarios are modelled with similar technology assumptions for all treatment processes, and have similar process energy results. These scenarios are also those with the largest process energy consumption. The findings show that they have nearly four times larger process energy consumption than the reference scenario. The biogas production is the process with the largest singular contribution to the process energy consumption. Figure 4.4 shows that the energy consumption of the insect rearing facility is the lowest of the final treatment alternatives for FW.

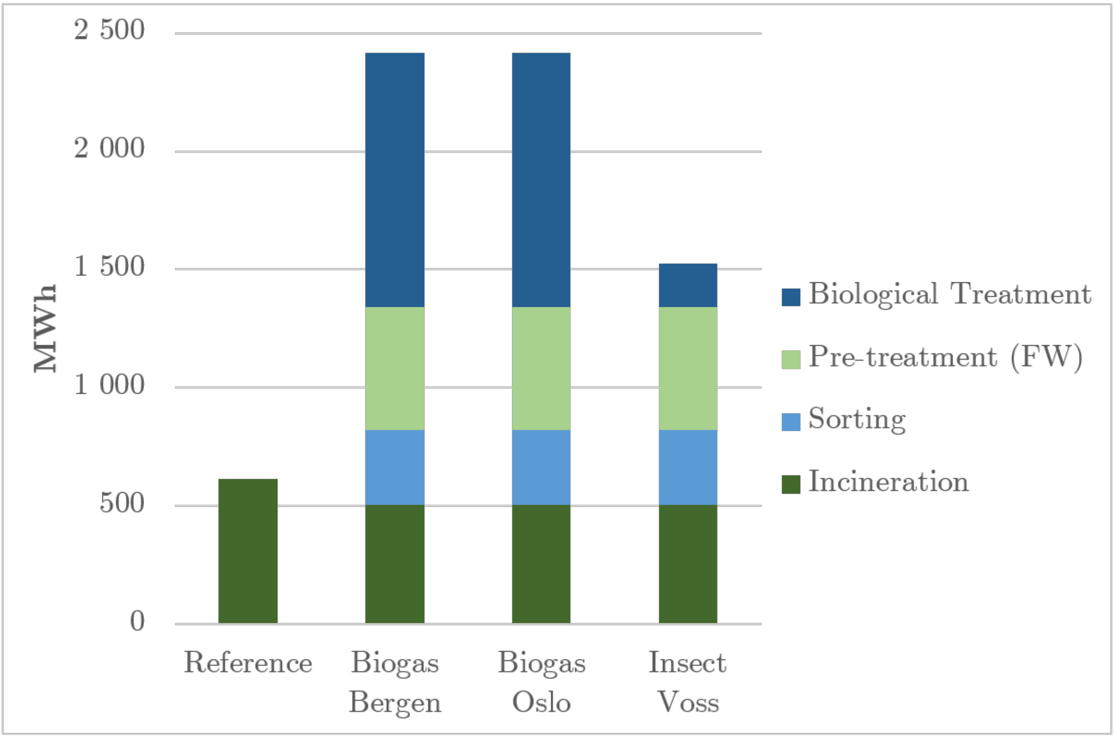

Figure 4.4: The consumed process energy for each process and scenario in 2030.

Overall, the energy consumption of the insect scenario with a treatment facility in Voss is the lowest of the food waste sorting scenarios. The biogas Bergen scenario has the second lowest energy consumption with a larger transport energy consumption than the insect scenario due to the transport of bioresidues to Oslo. In addition, the process energy for the biogas production is larger than the insect rearing.

All systems modelled produce energy through incineration. In the biogas scenarios, biogas fuel is produced through anaerobic digestion of the sorted food waste, while the insect scenario replaces energy production from the BIR incinerator with material/ product production. It is assumed that the reduction in energy generation from the BIR incinerator in the insect scenario is similar to that of the biogas scenarios since similar amounts of FW is removed. The two modelled biogas scenarios assumes a similar technology (in different locations) and therefore also similar energy production is assumed.

The energy production of the systems are shown in Figure 4.5. While removing almost 50 % of the FW from the BIR incinerator means a reduction of 6.5 GWh in yearly energy produced, the biogas production contributes with 8.6 GWh of upgraded methane fuel from the organic waste. This indicates that anaerobic digestion recovers the food waste at a better rate than incineration does.

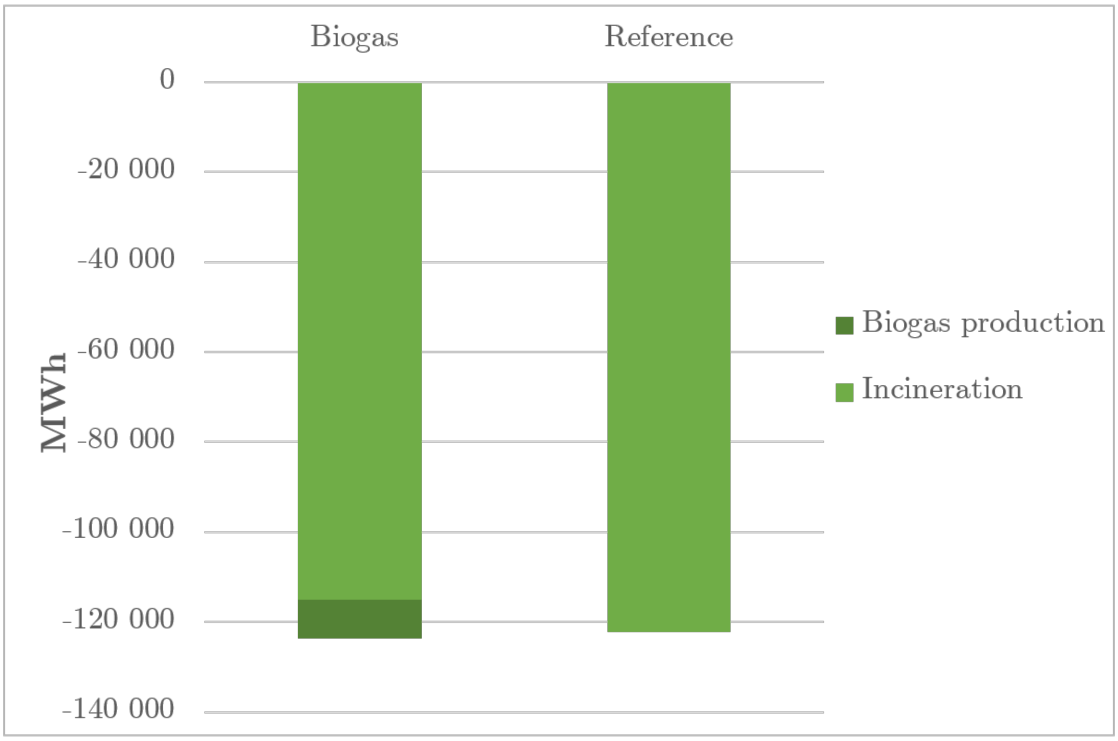

Figure 4.5: The produced energy shown for the biogas and reference scenario in 2030.

In all the modelled systems, the main contributor to the energy production is the waste-to-energy incinerator. In the reference scenarios, BIR incinerates all the waste generated in System B. When removing a share of the food waste (FW) from the incinerator, the total energy generation from household waste is reduced. The LHV is however increased, because the organic waste has a low LHV due to much water content. This means that the energy production per tonne waste is increased, and benefits the incinerator[14]. In the biogas scenarios the removed FW is recovered to methane fuel and the net production of energy in the biogas scenario increases. In the insect scenario, no new energy is recovered, and there is a net decrease in energy production in the system.

The overall energy results for the scenarios of system B are best illustrated by the energy recovery rate (ERR) indicator, introduced in Section 3.3.2. The ERR

[14]This only holds to a certain degree, since the incinerator is designed for a given LHV, and will be damaged by a too high LHV (Igesund et al. 2014).

is reduced from the reference system (B) to the scenarios. This is due to the increased transport work for the waste types studied, in addition to a lower energy production. The results are summarized in Table 4.1. The lowest ERR is found for the biogas Oslo scenario and is owing to the significant increase in transport energy compared to the other scenarios. The insect scenario comes out second lowest, due to that this scenario does not provide energy from the biological treatment of food waste. Biogas Bergen represents the least reduction, and deviates from Biogas Oslo due to the large difference in transport energy seen in Figure 4.3.

Table 4.1: The energy recovery rate (ERR) of the analyzed scenarios.

Scenario	ERR [%]
Reference B	78.1
Biogas Oslo	70.2
Biogas Bergen	76.8
Insect scenario	72.6

Figure 4.6 summarizes the model findings for System B in 2030, and illustrates the trade-offs between the energy efficiency and material recycling for all scenarios. The resulting energy recovery rate (ERR) is high for all scenarios due to the significant energy production, where the incinerator is the largest contributor. However, the ERR is reduced for all scenarios with FW sorting due to increased process energy and transport energy consumption. Simultaneously the system achieves a significant increase in the material recycling efficiency (MRE) when a share of the food waste is treated separately. The biogas Bergen and insect scenarios shows the best summarized results for the ERR and MRR.

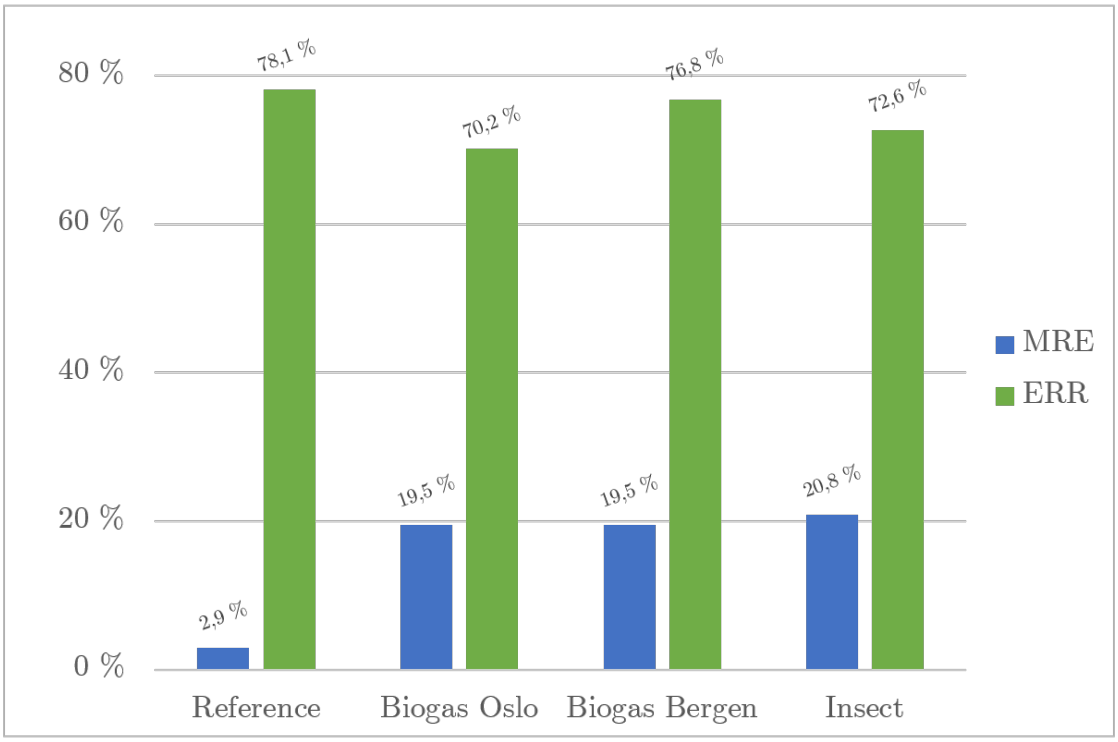

Figure 4.6: The material recycling efficiency (MRE) and energy recovery rate (ERR) for System B in 2030.

4.2.3 Greenhouse Gas Emissions

In this analysis the greenhouse gas emissions are estimated based on both the energy layer and material layer of the system. The greenhouse gas emission contributions included in this analysis stem from electricity and fuel consumption in the processes and transport in the system, in addition to incineration emissions to air. At the same time, a SWM system contributes to avoided emissions through recycling and with that avoidance of raw material production.

Figure 4.7 shows the emissions caused by the waste transport within the BIR system studied. As expected, the biogas Oslo scenario stands for the largest emissions related to transport. The results makes it evident that the biogas Oslo scenario is the only scenario where the food waste (FW) stands for the largest transport emissions. The emissions from the transport of the residual waste varies between 6.6 kg and 7.9 CO_2 per inhabitant for the different scenarios.

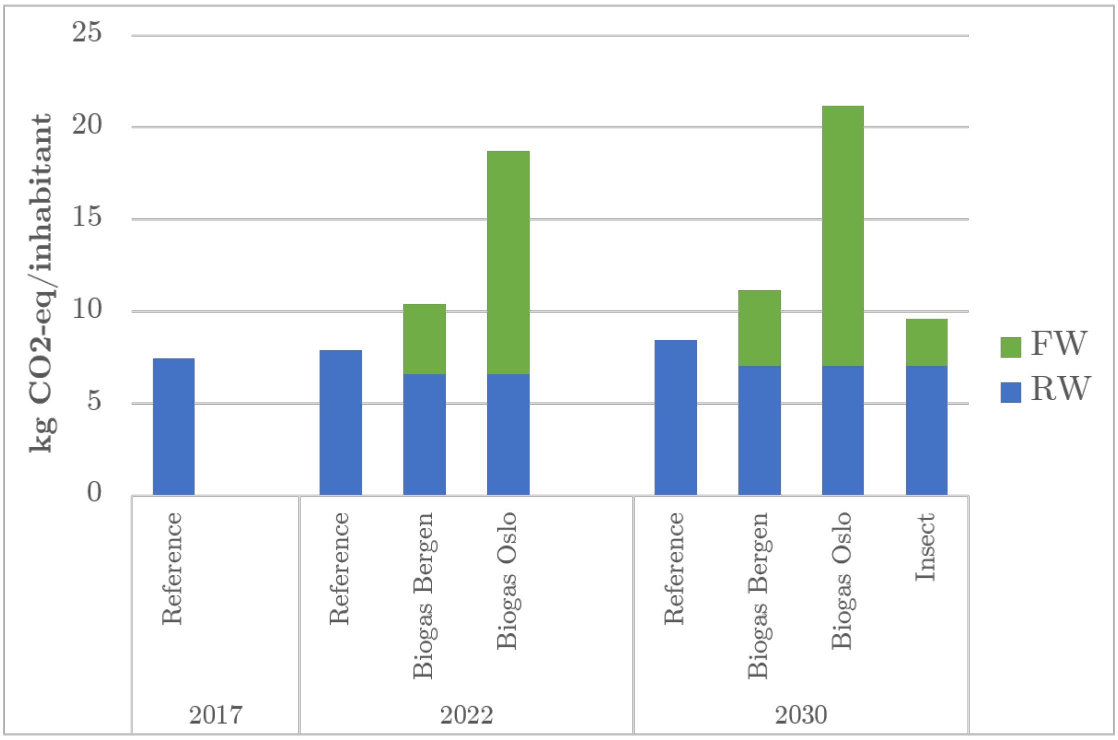

Figure 4.7: The annual transport emissions represented for residual waste (RW) and food waste (FW) in all modelled scenarios.

We can study the process emissions the same way, and Figure 4.8 shows the results. When compared to the transport emissions, the process emissions are comparatively small. There are two reasons for this. Firstly, the total energy consumption related to the transport of waste is of a much larger magnitude than the processes included. Secondly, the included process emissions stem mainly from electricity consumption, and this energy carrier has a very low emission factor when applying the Norwegian energy mix (NVE 2016). Unlike the transport emissions, the process emissions are the largest for the treated RW. The largest contribution is from the diesel consumption in BIR's incinerator, while the FW processes are modelled to operate with electricity and therefore result in less associated emissions[15]. Figure 4.7 and 4.8 also illustrates that the emissions per inhabitant gradually increases in the reference scenario due to the expectation that the waste amounts are growing faster than the population.

[15]It is likely that the biogas facilities utilize an engine fuel for certain operations, but this is assumed to be the produced biogas and thus does not contribute to net emissions.

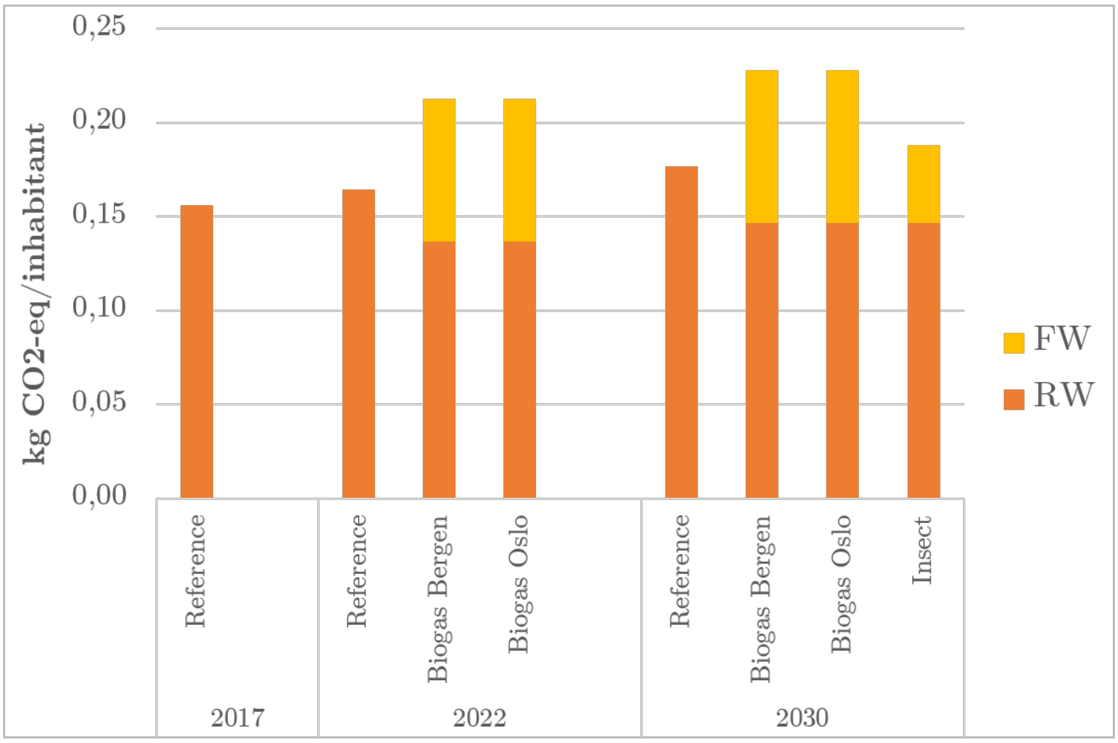

Figure 4.8: The annual process emissions represented for residual waste (RW) and food waste (FW) in all modelled scenarios.

Figure 4.9 shows the emission balance for the different scenarios in 2030, when including the transport and incineration emissions in addition to the avoided emissions. Avoided emissions represent the emission reduction caused by metal recycling (avoided exploitation), biogas production (avoided combustion), insect rearing(avoided soy production) and utilization of bioresidues (avoided fertilizer production). The emissions related to the recycling processes[16], are not included in figure 4.9, due to its negligible contribution compared to the other categories included. The graph illustrates that the net emissions are positive, i.e. the emissions are larger than the avoided emissions. This is caused by the singular contribution from the emissions related to the incineration of RW. At the same time, Figure 4.9 illustrates the difference in scale for transport emissions and the avoided emissions as a positive result.

The net emissions estimated in this analysis are low, but note that the emission calculations done here represents system B, and thus only includes the collection and treatment of RW and FW. This is in other words only part of the waste

[16]Including process 3 and 5 in system B, in addition to savings due to metal recycling.

treatment emissions each resident is causing.

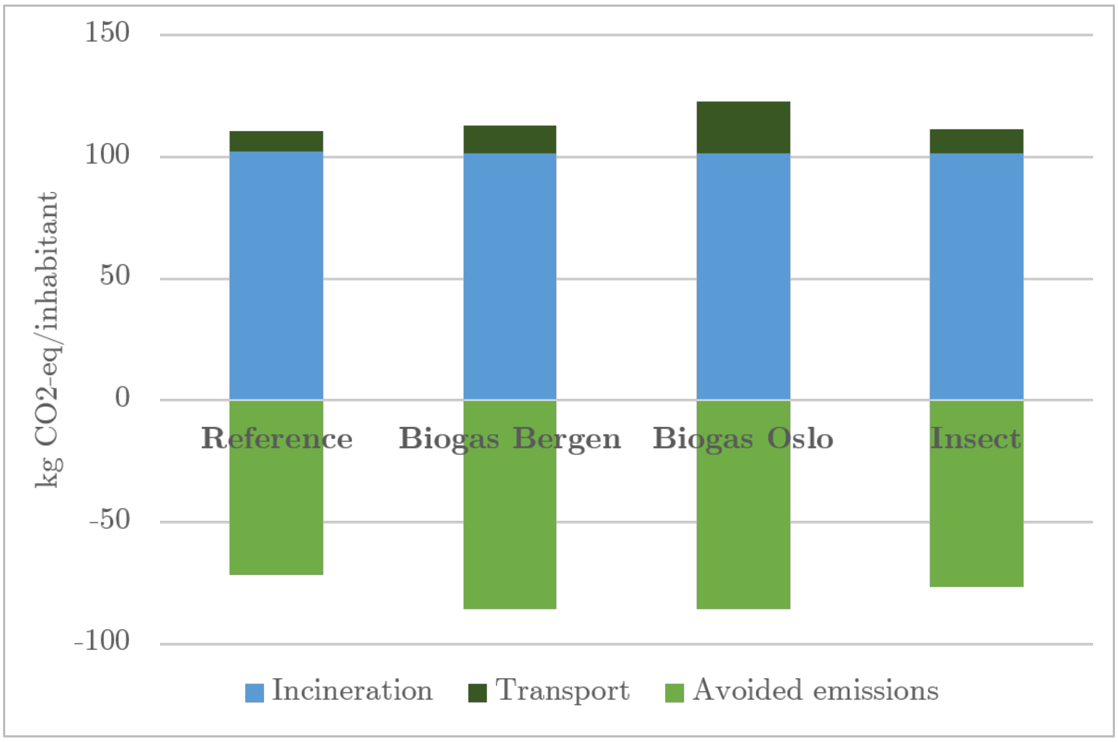

Figure 4.9: The annual emissions for all modelled scenarios in 2030.

The changes in incineration emissions between the scenarios are marginal, and are barely visible in Figure 4.9. Removing approximately half of the food waste from the residual waste will reduce the emissions from the BIR incinerator with 1.04 kg CO_2-eq per inhabitant, i.e. it goes from 102 to 101 kg CO_2-eq per inhabitant[17]. A share of the collected FW ends up in the incinerator after the sorting and pre-treatment, and is part of the explanation for the small changes. The rest of the incinerated amounts are unchanged and the incineration of FW does not stand for a significant contribution to the total emissions. The main contributing fractions to the incinerator emissions are *plastics* and *Other combustibles*.

The results presented Figure 4.9 favours the biogas Bergen scenario, which stands for the largest avoided emissions in addition to have lower transport emissions than the biogas Oslo scenario. The difference from biogas Bergen to the insect scenario

[17]The measured CO2-emissions in 2016 were 243 kg per inhabitant (MD 2018), so the findings here might be an underestimation. However, the measurements are based on the composition of the total received waste quantities, including industrial waste, and have an different overall composition than what is analyzed here.

is not large, but the insect scenario has a lower production output and therefore less avoided emissions.

4.2.4 Costs and Revenues

The costs and revenues analyzed in this study are those related to the investments and operation of the reference and alternative systems for food waste treatment. The analysis is performed to compare the current and alternative treatment solutions for food waste and therefore the cost analysis is executed for System B in 2030 only.

Figure 4.10 shows the composition of costs and revenues for all scenarios for the system in view (System B). The unspecified costs are those where the only available cost data was a single total amount in NOK/year. These include the costs related to collection transport, waste incineration, optical sorting and ash treatment. It is assumed that this is a combination of variable and fixed costs. In the reference scenario, all costs are modelled as unspecified costs. This is because BIR Privat today buys all their treatment services from other BIR branch companies, and in reality has had no direct investment costs related to the establishments of these facilities. It is however assumed that the investment costs are accounted for in the fee BIR Privat pays to send their waste for treatment, since it is sent to another branch company of the corporation. The reduction in incineration revenues are also included, since the effects of sorting FW are significant for the incineration.

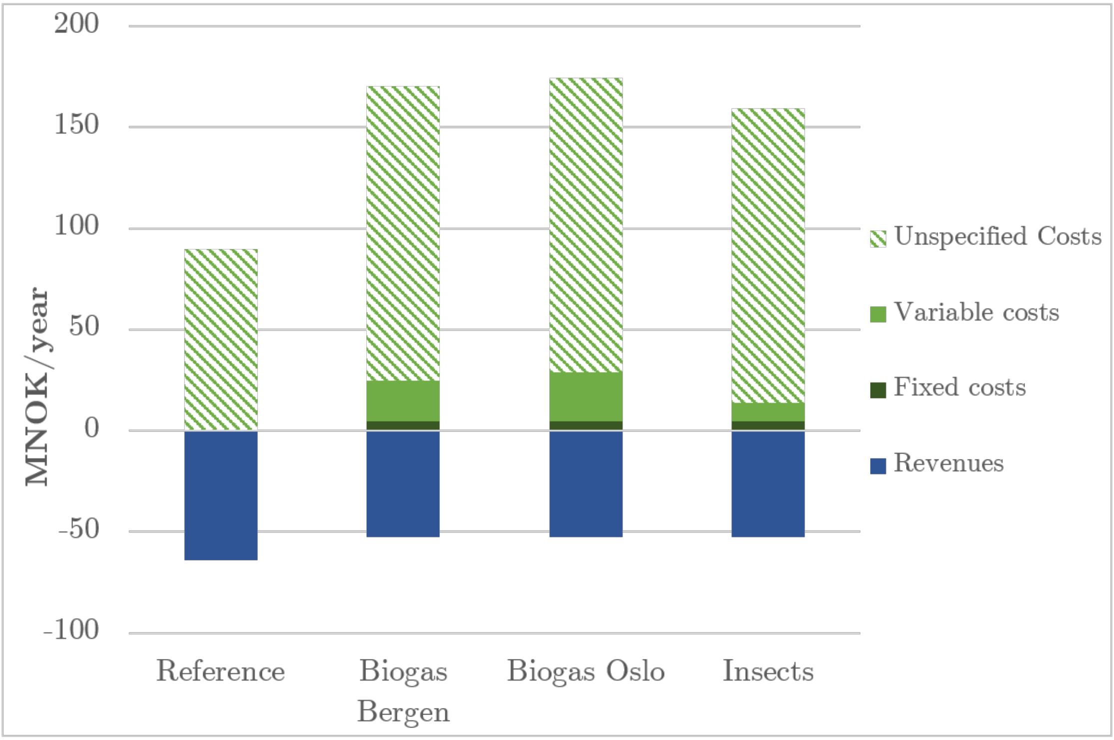

Figure 4.10: The costs and revenues of the 2030 scenarios by type.

Figure 4.10 illustrates the main economic features of introducing food waste sorting. With the system boundaries applied, the costs increase and the revenues decrease when food waste sorting and treatment is introduced. Firstly, the costs are significantly increased for all alternative scenarios. This is not a surprising result, since all scenarios means both investments in facilities (minimum optical sorting and pre-treatment facilities), added treatment costs for the FW and increased transport. The fixed costs for BIR are similar for all scenarios, since this includes the investment in an optical sorting facility and a pre-treatment plant. The variable costs are the largest for the biogas Oslo scenario because it has has the largest contribution of transport costs for the pre-treated food waste.

Secondly, the revenues are found to be reduced from the reference scenario. This is due to the reduced revenues when less household waste is incinerated. With the applied system boundaries that only encompass BIR, there are no new revenues acquired. However, there are in all FW sorting scenarios accounted for a gate fee - an expense for BIR to pay for outsourcing the treatment of FW. The insect scenario has the lowest variable costs because of the lower gate fee. The cost of transporting pre-treated FW in the insect scenario is also the lowest of the scenarios.

Furthermore, Figure 4.11 shows a comparison of the expense contributions from the different system processes for all scenarios in 2030.

The scenarios with food waste sorting means that additional processes are introduced in the system compared to the reference scenario, increasing the costs. For all scenarios, the costs of collecting waste at home addresses stands for the most significant contribution to the total costs. The collection transport and the corresponding costs are increased from the reference scenario to the FW scenarios as a consequence of the new collection scheme.

Of significant costs only appearing in the FW scenarios, the optical sorting facility stands out. This is based on estimations from Trondheim municipality, and assumed valid due to the similar characteristics with the BIR urban waste management system. The incineration costs are slightly reduced from the reference scenario, inline with the reduced amount of waste sent for incineration.

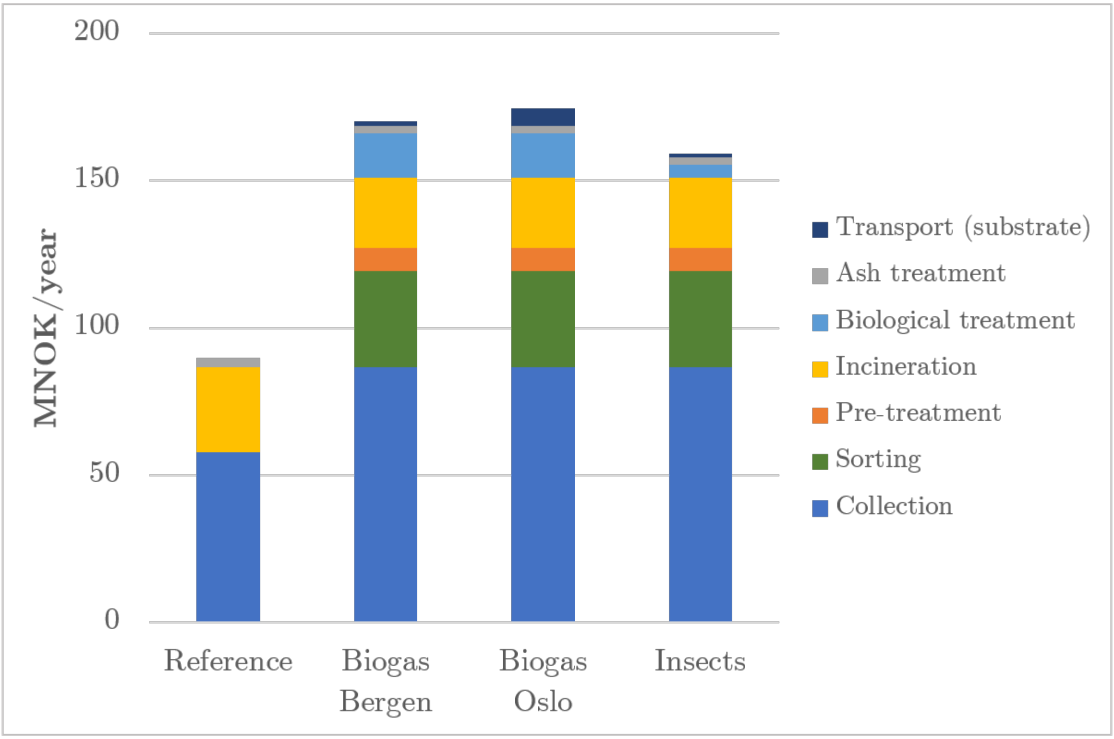

Figure 4.11: The process costs for each scenario in 2030.

The two biogas scenarios are modelled with similar costs for all components except for the transport to Oslo. The biogas Oslo scenarios transports pre-treated food waste to Oslo, while biogas Bergen only transports bioresidues to Oslo. This results in less tonnekilometres for the biogas Bergen scenario and thus a lower cost. However, the transport (substrate) costs are the lowest for the insect scenario. Here, the pre-treated food waste is transported between Bergen and Voss, which is a significantly shorter distance than Bergen - Oslo.

Based on the sources of this report (Ecopro and Invertapro), the price for treating the food waste differs significantly for the two different treatment technologies. The lower gate fee and less transport makes the insect scenario of lower costs than both biogas scenarios.

4.2.5 Sensitivity Analysis

The model offers the possibility of a sensitivity analysis where the user can adjust the transfer coefficients (TC) to see the effects this has on the system performance.

A sensitivity analysis was executed for System B with the use of the applied model. However, since system B is a limited model of the system, the indicators in the material layer are not representative, and thus the sensitivity analysis in the model did not give meaningful results. These results are therefore not included here. However, the energy indicators are consistent for system B and this sensitivity analysis of the biogas scenarios are displayed in Table 4.2 and 4.3 for the two biogas scenarios, respectively. The transfer coefficients TC are defined in Appendix B.

Table 4.2 sums up the findings for the biogas Bergen Scenario. The process energy fluctuates significantly (in percentage) due to the little order of magnitude of the included process energy. When TC24 is increased for residual waste (RW), and correspondingly reduced for TC23, there is a net saving in process energy since the pre-treatment process (3) is assumed to have a larger energy consumption than the incinerator (4). This is a desirable outcome, since this change in the sorting facility will send less RW to pre-treatment, and more to incineration. Similarly, when increasing the food waste (FW) sorted correctly and sent to pre-treatment (i.e. increasing FW in TC23), the process energy increases since there is more organic matter going to the energy demanding biological treatment. Furthermore, there is a large increase in consumed process energy, when the food waste (FW) in TC35 is increased with 10 %, and thus increasing the quantity organic matter sent for biological treatment (reducing the organic reject from the pre-treatment). The sensitivity results reflects the prominent specific energy consumption for the biogas production. The transport energy is also significantly increased, due to the increase in fertilizer transport between Bergen and Oslo. All sensitivity changes increases the generated energy (also TC24, but only marginally). The maximum increase is roughly 2.11 %, but remember that the originally generated energy is a large quantity. When TC24 is changed, the transport only marginally changes since process 2 (sorting facility), 3 (pre-treatment) and 4 (incinerator) are modelled at the same location. However, when increasing the amount of FW correctly sorted at the optical sorting facility (2) and sent to pre-treatment (3), there is an increase in transport energy consumption due to the effects further down the process chain.

The resulting energy recovery rate (ERR) changes in the biogas Bergen scenario are small. This illustrates how small contribution the fluctutation process energy is in the modelled system.

Table 4.2: Transfer coefficient (TC) sensitivity results in percentage change for the energy layer of the biogas Bergen scenario (System B).

TC	Fraction		ERR	Transport Energy	Process Energy	Generated Energy
TC24	RW	+10	+0.33	0	-22.50	0
TC23	FW	+10	-0.31	+8.4	+26.2	+0.96
TC35	FW	+10	-0.43	+18.5	+40.3	+2.11

The only differences between the biogas scenarios in the sensitivity analysis of the energy layer is the energy recovery rate (ERR) and the transport energy. The processes are modelled equally for the two scenarios, while the transport energy consumption originally is larger for the Oslo scenario. For instance, when increasing the organic matter sent from pre-treatment to biogas production, the biogas Oslo scenario has an increase in transport energy of 57 %. The large quantity of transport energy requirements in the biogas Oslo also leads to larger effects on the system ERR. E.g., sending more pre-treated FW from Bergen to Oslo reduces the ERR with 8.74 %, mainly due to the increase in transport energy.

The findings from this sensitivity shows that the model makes large fluctuations for changes in the TCs.

Table 4.3: Transfer coefficient (TC) sensitivity results in percentage change for the energy indicators of the biogas Oslo scenario (System B).

TC	Fraction		ERR	Transport Energy	Process Energy	Generated Energy
TC24	RW	+10	+0.31	0	-22.50	0
TC23	FW	+10	-4.10	+25.9	+26.2	+0.96
TC35	FW	+10	-8.74	+57.1	+40.3	+2.11

Furthermore, the model does not offer sensitivity on flow composition, i.e. re-allocating a fraction from one waste type to another. Instead, a simplified manual analysis was performed with a *perfect collection* assumption for the food waste. Assuming that there currently is no other waste type than residual waste (RW) receiving food waste (FW), a simple sensitivity analysis is performed where *all* FW is allocated to the FW sorting (no FW in the RW bin) and the corresponding material recycling efficiency is estimated. Applying the defined transfer coefficients throughout the pathway for the food waste gives the results as shown in Table 4.4.

Table 4.4: The material recyling rates for the scenarios and a *perfect collection* situation for food waste(FW).

	Model	Perfect FW
Reference A	20.9	
Biogas Scenarios	28.9	35.8
Insect Scenario	29.6	37.0

Perfect Collection is not a realistic assumption for any system, but the intention was to uncover what the maximum potential of introducing food waste sorting is in the BIR system, with a basis in the current situation. The results shows that the maximum achieved material recycling efficiency when introducing food waste treatment is 37 % for the BIR system.

5 Discussion

In this study, an analysis of the environmental performance of an urban waste management system has been performed. The analysis is holistic on several levels. Firstly, a comprehensive analysis was performed for material and energy flows, including all types of household waste collection for six different waste types. Secondly, a specific analysis of the system including only residual waste (RW) and food waste (FW) is executed for materials, energy, greenhouse gas emissions and costs. In this chapter, the main results presented previously are further evaluated and highlighted with the research questions in mind. The findings for System A are discussed in short, before the four layers of the main analysis (System B) are discussed in-depth. Then, strengths and weaknesses of the analysis are presented in Section 5.2. Furthermore, implications of the findings and suggestions are drawn up for subsequent work in the field of study in the final section of this chapter.

5.1 Main Findings

A comprehensive assessment of the BIR household waste collection and treatment system has been carried out for the current situation (System A). The results show that the current system has a material recycling rate of 20.9 %. The reason for the relatively low material recycling result is a combination of the current collection scheme and low collection efficiency for some fractions. The residual waste (RW) is the largest waste flow included in this study, and this is today incinerated and energy recovered locally. Of the fractions present in the RW, the major fraction is food waste (FW) (40.6 %). The large fraction is firstly owing to that BIR does not offer a service for source segregation and collection of FW. However, nearly 60 % of the FW found in BIR's most recent picking analysis was edible and could have been avoided. In addition, the collection efficiency is low for fractions such as plastics. Based on the supplied data from BIR, they currently collect 6.13 kg plastic per inhabitant in the plastic bin, and is slightly below the national average collected plastics[18]. In other words, a part of the potential lies in both the residents' consumption and utilization of the collection system.

Paper and cardboard (P&C) is the fraction that represents the best collection efficiency of the waste types assessed in System A. Factors affecting the good performance can be a well-established practice of P&C recycling or the low threshold for paper recycling. P&C recycling demands a low degree of effort from the inhabitants since it requires no cleaning and is collected in a separate P&C bin at household addresses in BIR's system. In regards to the material recycling, the metal recycling is found to be 90.8 %. This is the highest recycling rate for a

[18] 7.68 kg per inhabitant (SSB 2018)

single fraction in the studied system, and is a result of the combination of a collection efficiency of 53.8 % and urban mining ash treatment of the residues from the incinerator. This way, a significant amount of metals are recovered from the RW. However, the desire is still for there to be less metals in the incinerator, because it reduces the LHV and heavy metal formations is released with the flue gases as emissions when combusted (Wu, Lin, and Zeng 2014). A reduction of metals in the RW will also contribute positively to the workings of the collection scheme, as the energy and environmental effects are improved when the metals goes directly through its designated value chain and is recycled without going via incineration and ash treatment.

The results for the energy layer shows the significant energy generation the RW stands for in BIR's incinerator. In reality, the contribution is likely to be larger, since the LHV is modelled lower than expected. The energy recovery rate (ERR) is found to be 53.8 % for the BIR system. The transport energy requirements are only at a magnitude of 10 % of the generated energy in the incinerator, and the process energy is even less. The system boundaries and limited obtained data for the process and transport energy are however likely to have affected the ERR positively. It is expected that the ERR would be lower if more specific data was acquired and included for the energy input, E_{in}, especially when comparing to similar case studies carried out in 2017 (Schjoldager 2017; Callewaert 2017a).

The main activity of this assessment has been to study the effects of implementing FW sorting in the BIR system. It was therefore modelled a system only considering the collected RW and FW flows (System B). The following discussion on introducing food waste sorting and treatment put an emphasis on the results from System B, but results obtained for System A is included where it is relevant for the objective of this analysis.

Firstly, the material recycling efficiency (MRE) results are calculated in two ways and the results illustrated in the bar charts of Figure 4.2. When including FW sorting in system A, the overall effect on BIR's household waste collection system is estimated. Here, the MRE effects of introducing food waste sorting is an increase at minimum 8.0 % (biogas) and maximum 8.7 % (insect rearing) with the applied assumptions. The best overall results are found for the insect scenario, with 29.6 % material recycling. The FW scenarios are modelled with an average collection rate of 52.3 % of the generated FW, in accordance with corresponding collection schemes in other Norwegian urban waste management systems. This means that there currently is much unreleased potential in the food waste fraction in Norwegian solid waste management (SWM) systems. At the same time, findings show that almost 60 % of the FW in the RW in BIR's system (Relling and Grevskott 2017) is edible, and should never have entered the waste management system in the first place. If, as a thought experiment, it is assumed that the edible FW is eliminated from the waste management systems in the future, the maximum potential for FW recycling in the future is not underestimated in this study. This highlights a re-occurring dilemma for the waste management companies; should one pursue the growing awareness and expect a reduced waste generation or dimension expensive high-tech facilities for the current (and increasing) waste quantities?

The MRE calculated for System B, shows the same trends as those found in System A, but, more importantly, it illustrates the significance of FW in the RW. Remember that the current system boundaries (reference system) only comprise this waste type. Currently, only the metals are recovered and recycled from the RW, but if approximately half of the FW is treated biologically the MRE for what is currently sorted as RW is raised with up to 17.9 %.

The insect scenario represents the modelling with the best material recycling efficiency, due to the simple technology and corresponding low losses during treatment. The difference to the biogas scenarios are less than 1 %. However, it can be argued that the differences between the biogas scenario and the insect scenario, in terms of material recycling, are larger than what the model measures.

Moreover, following the EU waste legislation, the output from the biogas scenario and the insect scenario is all accounted for as material recycled. The difference between the products are however more significant than this definition illustrates. The fuel share of the biogas scenario is in reality energy recovered, but with the legislation definition (Potocnik 2011), *all* output, both fertilizer and fuel, is accounted for as material recycled. This conceals the difference between the scenarios. The insect scenario in reality conserves the resources at a higher level in the waste hierarchy, by less degradation of the output, in addition to resource quality and area of application, and can be said to represent a higher level of *upcycling* compared to the biogas scenario. With the perspective of circular economy (CE), the dried larvaes represent a more circular system than a biogas scenario, where a large share of the output (the biogas itself) is combusted at first utilization. The output of the insect scenario is 100 % material input to agriculture or as feed.

Regardless of the definition of material recycling, the best result is still far away from reaching the EU target of a municipal solid waste (MSW) material recycling rate of 65 %. The material sensitivity performed in section 4.2.5, illustrates the maximum recycling potential of the current FW amounts generated (System A). The insect scenario has the largest potential, with its 37 % MRE if all FW was source segregated. In other words, this is a maximum increase of 16.1 % from the current system. These findings emphasis that further measures needs to be taken to close in on the EU targets. With that said, it is evident that a complex system also triggers complex solutions, meaning that the approach to achieve significant improvement must be diversified and target several waste fractions. A start is to target the main problem fractions, and here, the organic food waste is a significant share with much potential, and a natural starting point. Furthermore, other problem fractions in BIR's system are plastics and textiles, and these finidings are substantiated in analyses of similar urban waste management systems around Norway (Schjoldager 2017; Callewaert 2017a).

The energy layer shows a large energy consumption and indicates that the introduction of FW sorting significantly increases the energy consumption when compared to the reference scenario. This is not surprising when considering that these scenarios mean an addition of three treatment processes as well as increased transport. The insect scenario has the lowest energy consumption of the FW scenarios, due to fewer food waste tonnekilometres, in addition to a low specific energy for the biological treatment. It can be argued that the applied specific energy consumption for the insect rearing facility most likely is an underestimation since it is based on South-European conditions and climate. However, the insect production process is based on a simpler concept and technology than biogas production, making it likely that the energy consumption is lower than for biogas production, and the results are assumed valid. Moreover, the insect technology is currently an immature technology, and it is therefore reason to believe the energy requirements has potential to be reduced in the future when the technology is further developed and made more efficient. Furthermore, the insect scenario performs worst of all scenarios in terms of energy production, since what is energy recovery of the FW in the other scenarios becomes material recycling in the insect scenario. The results demonstrate an increase in energy production when removing a share of the FW fraction from the incinerated RW and utilizing it in biogas production instead. Hence, the biogas scenarios perform best on energy production, but the trade-offs are the resulting increase in energy consumption in transport and processes that reduces the benefits of the increased energy production. The benefits are further reduced due to the increased LHV in the incinerator when a share of the FW is removed.

Furthermore, the energy recovery rate (ERR) is reduced in all scenarios, but neither here are the differences large. The sensitivity analysis shows that the ERR would be further reduced with 8.7 % if increased amounts of food waste (+10 %) is sorted, pre-treated and transported to Oslo for biogas production. This is rooted in the transport energy being a much larger contribution to the indicator than the biogas generation is for this scenario. In comparison, the ERR only decrease by 0.43 % when similar changes are applied to the biogas Bergen scenario. Here, the transport represents a smaller share of the ERR nominator, and thus smaller fluctuations are induced. However, in contrast to Oslo, the limited changes represents a positive change in the ERR, and thus there is reason to believe that increased FW to biogas production in Bergen would benefit the energy balance, while in Oslo the gains will be reduced due to the significant transport requirements for this scenario.

Despite the reduced energy production, the insect scenario replaces energy recovery from the incinerator with material recycling, and thus moves the treatment of the food waste upwards in the waste hierarchy compared to the other scenarios in this study. This makes the insect scenario a more desirable treatment method for the food waste viewed in the light of EU's circular economy action plan. Material recycling is preferred over energy recovery due to the lower level of processing, and hence less energy input requirements and degradation of resources.

Moreover, the greenhouse gas (GHG) emissions are calculated for the RW and FW treatment in all scenarios (System B). The relatively low CO_2-eq net emissions are signified by the limited system boundaries that only comprise collection and treatment of two waste types.

The transport and incineration stands for the two main emission contributions in the studied system. Of the two, the incineration is the major contributor with over 100 kg CO_2-eq per inhabitant. In contrast to these two contributors, the process emissions are negligible. The modelled process emissions are signified by the low emissions from the Norwegian electricity mix, mainly based on hydro power. The transport sector has considerably higher emission levels due to the dominating fossil fuels. However, if biogas was utilized as the transport fuel, the transport would have zero emissions because of the non-fossil origins of the organic matter. The increased supply of biogas from food waste treatment could in turn benefit the transport of the food waste itself when considering the related emissions.

The biogas scenario contribute with -5.6 kg CO_2-eq per inhabitant for avoided diesel fuel combustion. These avoided emissions only include the direct avoided emissions, i.e. those caused by combustion of diesel in the vehicle motor. This means that the emissions related to the extraction of fossil energy and the production of diesel fuel are omitted. If these were included, the difference between the biogas and insect scenario would increase. However, a limited scope curtails the comprehensiveness of the emission layer of this analysis. The emission indicators applied are also considered the reason for the fertilizer having a very significant contribution to the avoided emissions, since this applies to the production process itself. However, fertilizer, in contrast to diesel, does not account for emissions in the use-phase of its lifetime. A more extensive analysis of this layer would mean a complete life cycle assessment (LCA), and are considered beyond the scope of this analysis. Nevertheless, the results are an indicator of the contributions found in the different systems and considered an important factor when assessing and comparing the environmental performance of the scenarios.

The fluctuations in net emissions are not large between the scenarios (27-39 kg CO_2-eq/inhabitant), but there is a reduction in all the FW sorting scenarios compared to the reference scenario(39 kg CO_2-eq/inhabitant). The best net result is found for the biogas Bergen scenario, due to a significant share of avoided emissions (equal to the biogas Oslo scenario), and less than 50 % of transport emissions compared to biogas Oslo.

Moreover, it can be argued that the biogas replacing fossil fuel has a better environmental effect than energy recovery replacing hydro power generated electricity, such as in the case of the Norwegian electricity mix. This effect is reflected in the GHG layer, where the incinerator contributes with significant emissions to air and limited avoided emissions, while the biogas replacing fossil diesel noticeably contributes to avoided emissions. The insect scenario does not contribute to avoided emissions from avoided fossil energy production, but has a strength in that it replaces production of feed, while maintaining the organic matter's quality throughout the treatment. This is valued in the EU's circular economy (CE) Action Plan, but not valued more than biogas production in the legislation when evaluating the material recycling.

Furthermore, it is worth noting that the avoided emissions are much larger than the generated transport emissions. In the energy section, the transport was appointed as the main contributor to the energy consumption, but if the transport emission results are added to the equation, they only correspond to a fourth of the avoided emissions in the biogas Oslo scenario, and 12-13 % in the other scenarios. Nevertheless, there is a positive emission balance for the RW and FW waste treatment in all scenarios caused by the incinerator.

The final layer of the conducted analysis is the cost layer. A limited cost analysis was performed to illustrate the effects of incorporating food waste sorting in the BIR municipal waste management system. The main features of the findings in all scenarios show, not surprisingly, that the cost increases and the revenues decreases when implementing FW sorting. The investments in facilities or costs of outsourcing the treatment in addition to a more comprehensive and expensive transport infrastructure adds up to a nearly doubled annual cost compared to the current system. Simultaneously, the BIR corporation's revenues are decreased when the amount of combusted household waste is decreased. None of the scenarios involves that BIR establishes a final recycling facility, and thus there is no addition to the revenues from food waste treatment in the current scenarios. All scenarios are instead modelled with a gate fee BIR needs to pay to other companies for treating the waste they are responsible for. An interesting analysis that was not conducted here would be to estimate the costs and revenues of establishing a treatment facility within the BIR corporation. What would the gains be for BIR? Either way it is executed, it is safe to say that implementing FW sorting will be more expensive for the residents, who in turn will be the ones paying for a new collection and treatment scheme through the waste fee.

Most importantly, this assessment report can contribute with a holistic comparison of alternatives for food waste treatment. Table 5.1 summarizes the main findings of this analysis.

Table 5.1: A summary of the main results for materials, energy, (net) greenhouse gas (GHG) and (net) costs.

	Scenario	η_{coll}	**MRE**	**ERR**	**GHG**	**Costs**
			%		kg CO_2-eq/inhab	MNOK/y
System A	Reference	36.9	20.9	53.8		
	Bio Bergen	47.8	28.9	41.7		
	Bio Oslo	47.8	28.9	39.9		
	Insects	47.8	29.6	39.5		
System B	Reference	-	2.9	78.4	39.25	25.9
	Bio Bergen	22.5	19.5	76.8	27.42	117.8
	Bio Oslo	22.5	19.5	70.2	37.46	121.9
	Insects	22.5	20.8	72.6	35.27	106.8

In regards to the main analysis, the trade-offs and benefits are illustrated for system B in the lower half of Table 5.1. Here, the differences between the reference scenario and the FW scenarios become apparent. Both the material recycling efficiency (MRE) and GHG emission performances are positively affected from the current system to the FW scenarios. However, the energy recovery rate (ERR) is negatively affected, caused by increased energy consumption and less energy production. However, as pointed out before, this indicator does not reflect the type of energy generated. For instance, the benefits of replacing fossil fuels compared to electricity in the Norwegian electricity mix. This shift in energy carrier becomes apparent in the GHG-indicator. Here, there is a reduction in net emissions in all scenarios, and the biogas Bergen stands for the best result. Biogas Bergen has the best performance due to the combination of low transport energy (though an increase from the reference system) and a significant supply of biogas.

An additional benefit of biogas is that it easily allows storage, in contrast to the electricity and heat produced by incineration.

The indicators for the scenarios in System A are computed manually by combining the results from System B with the reference scenario of System A, and is done in the final stages of this thesis work to supply the overall results. Greenhouse gas (GHG) and cost computations for system A were considered beyond the scope of this analysis, and are not included. However, the GHG emissions and costs seen in the results for System B are valid trend lines for the effects on the complete system. Firstly, the collection efficiency in the scenarios is improved, due to the addition of food waste to the recycling scheme. This is similar for all scenarios due to the similar collection assumptions. Secondly, the MRE is improved, thanks to an increased share of collected waste being material recycled when FW is treated biologically. The trade-offs are the increased energy consumption, and this causes a drop in the energy recovery rate (ERR).

Furthermore, Table 5.1 shows that the insect scenario stands for the lowest costs and highest material recycling. As discussed previously, benefits of the quality of material recycling should be kept in mind when performing a holistic assessment, and one should not just draw categorical conclusions from the recycling rates alone. The net GHG emissions are however not much reduced, mainly because there are less avoided emissions compared to the biogas producing scenarios.

However, the analysis also makes it evident that the introduction of FW sorting and treatment will cost. The net annual costs (subtracted the incineration revenues) is displayed to the far right of Table 5.1. At most, the annual costs are five times larger in the FW scenarios compared to the current system (reference). The investments required to establish the facilities forms the baseline for the increased costs. Of the FW scenarios, the insect scenario distances itself from the most expensive biogas Oslo scenario due to less transport and an expected lower gate fee. Table 5.2 shows the cost for each kilo of reduction in CO2-eq emissions. The evidently most cost-efficient solution is the biogas scenario with treatment in Bergen. This system represent 15 % of the costs per kg CO_2-equivalent compared to the biogas Oslo system, representing the least cost-efficient system. The reason for the difference between these two scenarios is the increased costs related to the transport.

Table 5.2: The cost per reduction in kg CO_2-equivalents for each scenario compared to the reference.

Scenario	NOK/kg CO_2-eq
Biogas Bergen	19.29
Biogas Oslo	132.9
Insects	50.46

The findings show that to achieve less emissions and an increased material recycling rate, the introduction of FW treatment will cost in monetary and energetic terms. This is not surprising results, since the introduction of a recycling scheme for a new fraction necessarily must mean addition of treatment processes and change in logistics. Nevertheless, the earnings are in increase in the recycling rate, and thus conservation of resources, in addition to a reduction in emissions. An overview of the cost per emission reduction is shown in Table 5.2 and discussed above. It is however stressed that even though the cost efficiency is placed most emphasis on when these decisions are made, this report reflects the complexity of changing such recycling schemes. This complexity should motivate a well-substantiated assessment as the grounds for making a decision when choosing the food waste treatment solution.

5.2 Strengths and Weaknesses

In this section, the analysis in its entirety, as well as the model applied and the corresponding assumptions are discussed. The following discussion aims to point at strengths and weaknesses related to the main objectives of this study.

The MFA-based model applied in this analysis has been developed and finalized by Pieter G. Callewaert in August 2017, and is used here without any further implementation changes. The emphasis has been on understanding and using the model critically. The primary strengths of the model is rooted in the MFA-based formulation. The basic mass balance theory that the MFA methodology rests on, together with high level of detail requirements for the user input allows for a high flexibility. This generic baseline makes the modelling of complex waste management systems of various characteristics possible. This MFA-based model allows for tailored scenarios that can represent any SWM system, and to the author's knowledge there currently exists no model of similar entity for waste management systems. Notwithstanding, the detailed requirements for the model can be viewed as both a strength and a weakness. Some information the model requires was not possible to acquire detailed enough from BIR or the literature, and simplifications were therefore made. For instance, the model allows for up to 15 fractions in the waste types, but to acquire the calorific value of the specific defined fractions proved difficult. Some fractions' LHV is therefore manually calculated based on average material composition and individual calorific values. This is the part of the reason for the deviation between BIR's stated LHV and the one found here.

Moreover, the model allows for a sensitivity analysis. This is easily added to a modelled scenario and supplies valuable testing of the flexibility of the model, and effects of changes in the scenario. However, this sensitivity only allows for changes in the transfer coefficients, while it in some instances could be of interest to model changes in composition of collected waste types. A more flexible sensitivity tool would increase the robustness of the model and the discussion related to the results.

The main strength of this assessment report lies in the comprehensiveness of results that enables a multilevel evaluation of the scenarios up against each other. The analysis itself has been performed in two separate parts with different system definitions. Firstly, a holistic study of the current municipal waste management system based on the 2017 results was carried out (System A). This was to obtain comprehensive performance results representing the municipal waste management system as it is today. Parallel with this analysis, a limited model was designed with the objective of representing the effects of the modelled food waste scenarios in an evident way (System B). This combination has provided results on an overall level in addition to the specific results, targeting the food waste fraction. Accordingly, the assessment of BIR's SWM system can be said to be holistic on several levels.

Furthermore, this partition allowed for a more extensive analysis of System B. The System B assessment has been performed for four different layers, and has a diverse set of results. These results concern materials, energy, emissions and costs, and thus encompass both economical and environmental aspects of a SWM system. Such a comprehensive analysis allows for comparisons of the scenarios and provides wide-ranging findings. For instance, the results for the energy layer and cost layer strengthens the insect scenario performance, while the emission layer strengthens the biogas scenarios.

Specifically, the insect scenario brings new technology and possibilities up for discussion in the field of food waste treatment. Insect rearing is to the author's knowledge not utilized for the purpose of large scale organic waste treatment in Norway today. Similar technology is currently only applied in research and development. This assessment serves as a contribution to a field where biogas currently is the indisputable treatment option, raising the question if enough research is being done on alternative treatment solutions for food waste. The findings of this report highlights that other alternatives than anaerobic digestion of food waste should be considered in the future.

Moreover, the insect scenario is the scenario that place the most emphasis on the literature due to the currently immature technology and few existing facilities. The results for the insect scenario is marked by a careful modelling, due to the developing technology. There is therefore reason to believe that the insect scenario has further undetected potential.

As mentioned earlier, the modelled scenarios are represented with simplified assumptions due to a lack of necessary input data. However, limitations in the model is not the reason for all of the simplifications carried out here. A lack of accurate data in combination with a limited scope and time scale has contributed to confine this analysis. In retrospect, however, some assumptions could have been adjusted or further evaluated throughout the process of data collection and analysis, and these are briefly discussed below.

Firstly, a large share of the data applied in the analysis is supplied by BIR, and is case specific. However, specifically when defining the parts of the system only present in the modelled scenarios, generic data from other systems and the literature was applied. This means that to a great extent, the reference scenario is represented in line with the existing system, but for the developed scenarios, additional assumptions needed to be made. It also adds to the uncertainty that a range of sources supplies data for the different layers. Where possible, it was attempted to use the same source as much as possible, but the distinctive characteristic of each municipal waste management system makes this difficult. The application of new technologies additionally complicated the consistency of the data collection.

For instance, all modelled future scenarios includes a pre-treatment facility established in Bergen. The pre-treatment stage of biological waste treatment is critized for not being sufficient enough to exploit the full potential of the biogas production in Norway currently (Marthinsen 2017). However, there are promising technologies on the market, and Ecopro is an example of a biogas producer who have done extensive work to reduce the reject from pre-treatment and biogas production (Bjørndal 2013). Due to a lack of specific data from separate pre-treatment and biogas production facilities, assumptions on the performance was based on several facilities. The reject flow from the pre-treatment was modelled high, according to the report by Marthinsen (2017), but lower in the biological treatment facility, in accordance with Ecopro (Bjørndal 2013). The results show a lower biogas outcome than expected, and this is seemingly due to the model which has an overall reject percentage that is larger than it would be in reality. In other words, the resulting biogas production is most likely underestimated.

Furthermore, the insect scenario represents complex physiological processes in the rearing of the insects that were not attempted to include in this scenario. This means that the simplifications were limited to an assumed mass distribution, which in reality most likely is more complex than this assessment indicates.

Furthermore, since the model currently only has implemented the material and energy layer, the emissions and cost layers are computed manually, and of a simpler character. It is acknowledged that there always is the risk of problem shifting when changes are introduced in a system. Firstly, the emission layer could have been extended to a *impact's* layer. A SWM system is complex in composition and technologies, and this reflects the variety of impact categories it influences. With that, there are effects of the implemented measures that are not illustrated with this analysis. For instance, both biogas production and insect rearing conserves valuable nutrients in circulation, while incineration of the same waste decompose such compounds. Both these technologies generates output that replaces products of organic origins, and reduces land use. The biogas produced from organic waste can for instance replace biogas produced from corn, and thus the saved cropland can be allocated to food production e.g.. Furthermore, the cost analysis is of simple character, due to limited available data. The difficulty to retrieve costs from new facilities is due to the lack of public reports for similar systems.

The findings of this analysis is also a consequence of the set system boundaries. For instance, the low material recycling rate is influenced by the selection of included waste types. Wood is a large waste flow from the recycling stations included in the analysis that goes to incineration, while the material recycled garden waste (composting) was excluded. However, the wood waste type is considered a fraction with more potential than the garden waste in terms of improved resource utilization, and is the reasoning for the selected system boundaries.

On the other hand, there are also positive effects that are not represented in this assessment. the output from the biological waste treatment processes have positive effects on diverse categories. For instance, the implications of avoided production induced by the waste treatment that replaces materials or resources is only included in the manually calculated emissions layer, and thus only for the impacts emitted to the air in CO_2-equivalents. In the model, the energy indicators does not take into account the avoided energy use that a material recycling facility represents for instance, and thus these emissions are neither subtracted. Moreover, the only included emissions are the direct emissions that are replaced. For example, the emissions avoided from the extraction and production of diesel is not included in the analysis. This would however be of interest for continued work, and is worth remembering when evaluating the findings.

5.3 Implications of the Findings and Future Work

The assessment reveals aspects of municipal food waste treatments that has not yet been studied in-depth in Norwegian solid waste management. The indisputable choice for food waste treatment today is biogas production, mainly due to the lack of alternative technologies that can supply equal benefits and efficient resource utilization, namely, degradation of matter, conservation of nutrients, material recycling and energy recovery to name some. However, there are alternatives appearing. The findings of this report suggests that there are reasons to consider other options than biogas treatment when implementing food waste sorting in an urban waste management system. Furthermore, this assessment report contributes with an alternative treatment technology, where the novel technology of insect rearing with protein production based on the input of organic waste is modelled. The simple technology of insect rearing is the main benefit compared to the advanced and energy intensive biogas production technology. Furthermore, the insect protein production represents a circulation of materials that biogas production does not. However, the technology is still immature, and not readily developed for large scale production yet. The chances that a large urban waste management company will send its food waste to a insect rearing plant in the impending future is small. However, the literature review reveals promising research and technology development on the field, and just that is why it is important to bring up the alternative now. That way, the insect rearing research can develop in coherence with the waste management, particularly in systems such as BIR's, who have not introduced food waste sorting services yet. The promising insect rearing concept, bearing in mind its coherence with the circular economy, joint with findings of this report, points to that insect rearing with organic waste input should be further investigated and evaluated. Adding to the list of benefits for BIR is an assumed location of the insect rearing at Voss, limiting the transport of both the organic waste stream and bioresidues due to the unique properties of the insect rearing bioresidues. Nevertheless, it is stressed that contributions are system specific, for instance in regards to the chosen energy carriers or the transport energy requirements.

Moreover, an alternative that has not been studied in this assessment, but that BIR has launched as a possible solution is to consider the combination of biogas production and insect rearing. Based on the findings of this analysis, the optimal combination would be between the insect production in Voss and local biogas production in Bergen. This way the insect rearing plant could receive the quantities needed, and any surplus FW could be utilized in biogas production. This is a likely solution because both the biogas plant in Bergen and an insect rearing plant have limited capacity (Igesund et al. 2014). An assessment of this kind could also be executed to secure optmization of utilization of the resources and benefits. It is worth considering to diversify the outsourcing of food waste treatment, and thus avoiding dependency on one alternative that might not be the best solution in the long-term.

Furthermore, an important part of the results in this system analysis is the applied collection scheme. The collection phase of a waste system is energy and labour intensive and have an impact on all layers in this analysis. however, the limited time scale of this analysis forced an emphasis on the treatment solutions, and thus the collection scheme was modelled equal for all scenarios. Future work could be to examine the nuances of the collection system, in terms of energy carriers, logistics, infrastructure and other relevant factors.

In regards to the assessment of BIR's urban waste system, the analysis shows a currently low material recycling rate. The assessment also reveals that the implementation of food waste treatment does not contribute enough for BIR to reach the EU target of 65 % municipal waste material recycling. Here, further research should be done on what other initiatives there are that can increase the material recycling rate of the BIR system additionally. The emphasis should, however, not just be on reaching the EU target, but preferably also take into account the quality of the recycling output, in addition to other effects.

This assessment has verified that there is no simple solution to achieve significant improvements in complex urban waste management systems. Several initiatives need to be implemented to target multiple fractions. Moreover, if several targets are implemented in parallel, the benefits are amplified, whilst the marginal costs are lower. For instance, if implementing an optical sorting facility, the comprehensiveness should be evaluated, and it should be considered to implement technology for recognition of more fractions than just organic waste. Plastics is, e.g., evidently a problem fraction in many municipal waste management systems around Norway, and there currently exists technology for optical recognition of plastic items in waste sorting[19].

To conclude this section, it is stressed that future work on organic waste treatment should not only target the much needed improvement of the biogas technology, but also encompass and evaluate other existing and emerging technologies. The development and optimizing of resource exploitation will profit on an increased research activity.

[19] ROAF has implemented plastic recognition in their optical sorting facility.

6 Conclusions

This report has assessed the effects of implementing food waste sorting and treatment in the urban waste management system surrounding Bergen and operated by the municipal waste management company BIR. From the findings presented in this report, some final reflections are made and careful conclusions are drawn.

Through the application of an NTNU-developed model finalized in 2017, scenarios with food waste sorting are developed and modelled and the environmental performance of BIR's household waste collection system has been analyzed. The material flows of selected fractions in BIR's collection of household waste was implemented in the model, in addition to specific energy data for waste types, transport laps and processes in the system.

Two separate analyses are carried out. They both have a time frame from 2017 and until 2030 when the EU circular economy material recycling targets are due. The first analysis targets a holistic representation of the current BIR household waste collection and treatment system and includes waste collected en-route, at central collection points and recycling stations. Furthermore, the pathways for the treatment of six significant waste types are modelled and analyzed. The resulting material recycling efficiency is 20.9 %, and seemingly low compared to other similar system. This result is partly framed by the system boundaries. Moreover, the plastic fraction is identified as a critical fraction, due to a large share of plastics in the residual waste. These findings are based on a picking analysis and should be carefully used to draw categorical conclusions, but a current collection of 6.13 kg per inhabitant in the plastic collection is below the national collected average of this fraction. The plastic is critical for several reasons, such the fossil origin, the variety in quality and properties as well as the slow degradation rate, making it harmful for the natural environment.

Secondly, a separate analysis is carried out for the current residual waste (RW) type. In addition to a reference scenario representing the current situation, scenarios were developed where the service of separate food waste collection and treatment is introduced, and thus a food waste (FW) type is defined. The food waste scenarios are motivated by the desire to increase material recycling and improve resource utilization in the BIR system, in line with the EU circular economy targets. Food waste is currently the largest fraction in the RW and is therefore a natural starting point when working to increase the material recycling. Three treatment alternatives are modelled for 2030: Biogas production in Bergen, biogas production in Oslo and insect rearing in Voss (Hordaland).

The source separation of food waste improves the material recycling efficiency

(MRE) in all scenarios, compared to the current situation. In the scenarios with food waste sorting and treatment, the MRE is increased to 28.9 % with biogas production and 29.7 % with insect rearing. The insect rearing has a marginally larger performance due to a simpler technology, and less loss in the treatment process. It is also a given that all alternatives considered in this assessment have a better resource utilization and conservation than the current incineration of the food waste fraction. The trade-offs of the introduction of food waste sorting are increased use of energy in processes and transport. This also causes increased emissions, but simultaneously the avoided emissions increases, largely due to the savings generated by the output of the biological treatment processes. More precisely, the production of biogas, fertilizer and insect proteins replaces combustion of fossil fuels, production of inorganic fertilizer and soy protein production, respectively.

The findings in material recycling illustrate that more measures need to be taken for BIR to increase the material recycling and improve resource exploitation. Food waste is a fraction with significant potential but is not the only fraction that should be targeted. A *perfect collection* simulation of the food waste fraction shows that the maximum resulting material recycling efficiency in the BIR system is 37 %, and this is still far from the EU target of 65 % municipal waste material recycling rate.

Simultaneously it is stressed that well-considered and substantiated choices should be taken to avoid problem shifting. In regards to food waste sorting, this study illustrates that there are emerging technologies that can challenge anaerobic digestion and biogas production as the biological treatment solution, and this should increasingly be acknowledged when improving resource utilization in municipal waste management systems. Moreover, decision-makers should encourage an evaluation of the existing and emerging alternatives in food waste treatment. This analysis points to that there are alternatives that can challenge and potentially outdistance biogas production in a circular economy.

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Appendices

Appendix A The System

Table A.1: SSB projections of the population development and estimations for the scenario years in the BIR municipalities.

			Linear interpolation		
Municipality	**2016**	**2040**	**2017**	**2022**	**2030**
Askøy	28 380	41 900	28 943	31 760	36 266
Bergen	277 391	328 700	279 529	290 218	307 321
Fusa	3 876	4 300	3 894	3 982	4 123
Kvam	8 475	9 100	8 501	8 631	8 840
Os	19 742	30 100	20 174	22 332	25 784
Osterøy	7 957	10 200	8 050	8 518	9 265
Samnanger	2 443	2 600	2 450	2 482	2 534
Sund	6 975	9 900	7 097	7 706	8 681
Total	355 239	440 675	358 638	375 629	402 814

Table A.2: Estimated distribution of collection technologies allocated to car types. The data basis used for collection technologies in the model.

Collection Techn.	**Car Type**	**Size [l]**	Housing Units
Small & Large Bins	Compacting Car		
		60	452
		140	87 468
		240	19 098
		400	9 721
		600	90
		660	31 577
Containers	Hook & lift Car		
		4 000	5 363
		6 000	10 046
		16 000	810
Underground	Crane Truck		1 884
Mobile Vacuum	Suction Car		10 297
Central Vacuum	Container Car		6 062
Total			**183 785**

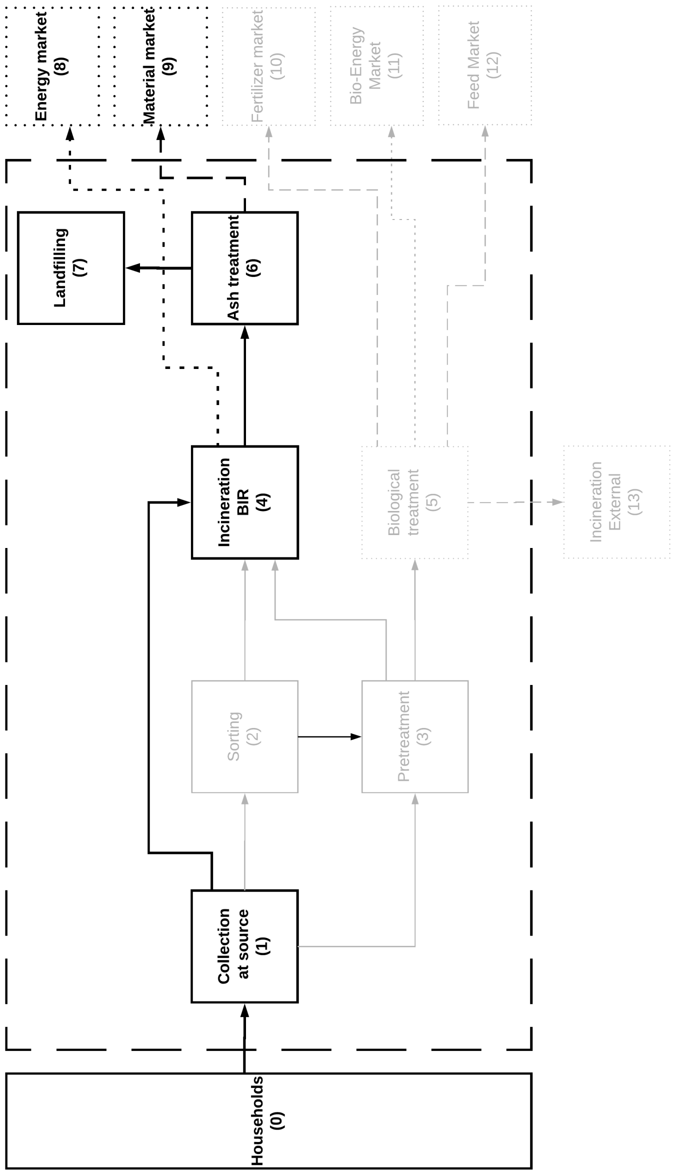

Figure A.1: System B: Flow chart representation of the Reference scenario system.

Appendix B Material layer

Table B.1: Definition of the fractions found in the picking analysis of residual waste.

Fraction	Description
P&C	Materials produced of plant fibers from wood pulp
Plastics	Fossil material from packaging and other products
Glass	Packaging glass
Organic waste	Edible and non-edible organic waste originating from food production
Garden waste	Soil, plants and other organic plant waste
Textiles	All types of textile fibres, leather and shoes
HWEEE	Waste containing compounds harmful for humans or the environment, also including electronic waste
Other combustibles	Combustible compunds that cannot be recycled (e.g. diapers, non-recyclable paper, cat sand, vacuum bags)
Other non-combustible	Mainly stones, ceramics and porcelain. Also including mineral wool, plaster (not desired in incinerator), in addition to glass and metals that is not package material
Metals	Scrap metals and large, pure metal units

Table B.2: Assumed composition of incoming source separated waste types in *all* scenarios.

	Waste type [%]				
Fraction	Plastic[a]	Paper[b]	G&M[c]	Textiles[d]	Wood[e]
P&C	0	98.0	0	0	0
Plastics	77.0	0	0	0	0
Glass	0	0	91.7	0	0
Food waste	0	0	0	0	0
Garden waste	0	0	0	0	0
Textiles	0	0	0	85	0
HWEEE	0	0	0	0	0
Other combustibles	23.0	2.0	5.1	15	0
Other non-combustible	0	0	0	0	0
Wood	0	0	0	0	100
Metals	0	0	3.2	0	0

[a]Based on Bjørnerud (2017b).

[b]Based on BIR employee Kirsten Grevskotts estimations.

[c]Based on Syklus (2016).

[d]Based on *Loopedia: Klær og sko* (2017).

[e]Simplified assumption.

Table B.3: Assumed composition of affected waste types in FW sorting scenarios.

	Waste type [%]		
Fraction	RW[a]	FW in bags[b]	FW in bins[c]
P&C	9.13	0	0
Plastics	16.6	0.9	1.8
Glass	3.86	0	0
Food waste	24.95	94.4	83.9
Garden waste	5.40	0	8.6
Textiles	5.92	0	0
HWEEE	1.54	0	0
Other combustibles	25.26	4.7	5.6
Other non-combustible	3.22	0	0
Metals	4.12	0	0
Sorting degree		49	57

[a]Based on BIR's 2017 picking analysis and adjustments for FW sorting.

[b]Based on a picking analysis from ROAF by Bjørnerud (2017b).

[c]Based on a picking analysis from RfD by Mepex Consulting (2015).

Table B.4: Definition of the flows in the system A.

Flow	Waste types	Description
X01	RW, P&C and plastics	Home Collection
X02	G&M and textiles	Collection Points
X03	RW large units, RW small units, P&C, plastic, metals, wood	Recycling Stations
X14	P&C, plastics	Home collected fractions for sorting/packing
X16	RW	Home collected fractions for incineration (BIR)
X24	G&M	Fractions from collection points to packing (BIR)
X25	Textiles	Textiles for material recycling
X34	RW large units, P&C, plastics	Grinded RW and residue P&C for incineration
X35	Metals	Scrap metals for recycling
X36	RW small units, wood	RW and wood for local incineration
X313	Wood	Wood to be incinerated externally
X45	G&M, P&C, plastics	Home Collection
X46	RW large units, P&C	Grinded RW and P&C residues
X59	G&M	Metals to market
X511	G&M, P&C, plastics, textiles	Materials to market
X512	Textiles	Textiles to retail (no treatment)
X513	G&M, P&C, plastics, textiles	Loss/bad quality materials to incineration (External)
X67	Ashes, metals	Residues from incineration to ash treatment
X610		Energy from incineration to market
X78	Ashes	Residue materials to landfill
X79	Metals	Metals to market

Table B.5: Definition of the flows in the system B.

Flow	Waste type	Description
X01	RW & FW	Home Collection
X12	RW & FW	Types collected in same bin sent for optic sorting
X13	FW	Food waste collected in separate bins.
X14	RW	Collected in separate bin, sent directly to incineration
X23	FW*	FW bags separated from RW and sent to biological pre-treatment
X24	RW	RW separated from FW bags and sent for incineration
X34	RW	Reject from pre-treatment sent for incineration
X35	FW	Pre-treated organic substrate sent for biological treatment
X46	RW	Residues from incineration to ash treatment
X48		Energy from incineration to market
X510		Residues from biological treatment with nutrient content to fertilizer product
X511		Biogas fuel to bioenergy market
X512		Proteins from dried insect larvaes to the feed market
X513		Reject from biological treatment to incineration
X67		Inorganic, non-metal compounds to landfill
X69		Recovered metals to metal market

*Only FW but the flow may contain non-organic elements.

Table B.6: The predefined transfer coefficients (TC) for System B by waste flows in percentage.

	Reference		Biogas		Insects	
	RW	FW	RW	FW	RW	FW
TC13	0	0	0		0	
TC14	0	0		0		0
TC23	0	0	11.4	78	11.4	78
TC24	0	0	88.6	22	88.6	22
TC510	0	0	0	34.75	0	71.03
TC511	0	0	0	57.75	0	0
TC512	0	0	0	0	0	28.97
TC513	0	0	0	7.5	0	0

Table B.7: Predefined transfer coefficients (TC) common for all scenarios per fraction in percentage.

	TC46		TC48		TC67		TC69	
	RW	FW	RW	FW	RW	FW	RW	FW
P&C	20	20	80	80	100	100	0	0
Plastics	20	20	80	80	100	100	0	0
Glass	20	20	80	80	100	100	0	0
Organic waste	20	20	80	80	100	100	0	0
Garden waste	20	20	80	80	100	100	0	0
Textiles	20	20	80	80	100	100	0	0
HWEEE	20	20	80	80	100	100	0	0
Other Combustibles	20	20	80	80	100	100	0	0
Other non-combustible	100	100	0	0	100	100	0	0
Metals	100	100	0	0	8	8	92	92

Table B.8: The predefined transfer coefficient TC34 for all scenarios in percentage of the flow in percentage.

	Reference		Biogas		Insects	
	RW	FW	RW	FW	RW	FW
P&C	0	0	100	0	100	0
Plastics	0	0	100	100	100	100
Glass	0	0	100	0	100	0
Organic waste	0	0	100	10.45	100	10.45
Garden waste	0	0	100	100	100	100
Textiles	0	0	100	0	100	0
HWEEE	0	0	100	0	100	0
Other Combustibles	0	0	100	100	100	100
Other non-combustible	0	0	100	0	100	0
Metals	0	0	100	0	100	0

Table B.9: The predfined transfer coefficient TC35 for all scenarios in percentage of the flow in percentage.

	Reference		Biogas		Insects	
	RW	FW	RW	FW	RW	FW
P&C	0	0	0	0	0	0
Plastics	0	0	0	0	0	0
Glass	0	0	0	0	0	0
Organic waste	0	0	0	89.55	0	89.55
Garden waste	0	0	0	0	0	0
Textiles	0	0	0	0	0	0
HWEEE	0	0	0	0	0	0
Other Combustibles	0	0	0	0	0	0
Other non-combustible	0	0	0	0	0	0
Metals	0	0	0	0	0	0

Appendix C Transport Energy

Table C.1: Estimated transport distances (km).

		Scenarios		
Flow	**Ref.**	**BG Oslo**	**BG Bergen**	**Insect**
X12[b]	-	94.41	94.41	94.41
X12[c]	-	23.40	23.40	23.40
X12[d]	-	157.34	157.34	157.34
X12[e]	-	87.41	87.41	87.41
X12[f]	-	34.00	34.00	34.00
X13	-	94.41	94.41	94.41
X14[a]	94.41	94.41	94.41	94.41
X14[b]	94.41	-	-	-
X14[c]	23.43	-	-	-
X14[d]	157.34	-	-	-
X14[e]	87.41	-	-	-
X14[f]	34.00	-	-	-
X23	-	0	0	0
X24	-	0	0	0
X34	-	0	0	0
X35	-	1000	0	220
X46	84	84	84	84
X48	0	0	0	0
X510	-	0	1000	0
X511	-	0	0	0
X512	-	0	0	0
X67	0	0	0	0
X69	0	0	0	0

[a] Collection techn: Small bins

[b] Collection techn: Large bins

[c] Collection techn: Containers

[d] Collection techn: Underground

[e] Collection techn: Mobile vacuum

[f] Collection techn: Central vacuum

Table C.2: The assumed distribution of total kilometres driven within BIR for MSW collection in 2017, reference case (based on 2015 data).

Waste Type	Transport share [%]	Kilometers
RW	72	1 104 988
P&C	12.5	191 838
Plastics	12.5	191 838
G&M	3	46 041
Total		**1 534 705**

Table C.3: Estimated ton per trip for each technology and waste type.

	Ton/trip						
	RW	P&C	Plastics	G&M	Textiles	FW	RW + FW
Bins 140-660 l	5.4	2.7	1.8			7.0	3.78
Containers	1.34	0.67	0.34	3.2	1.59*		1.34
Underground	9.0	4.5	2.25				9.0
Mobile Vacuum	5.0	2.5	1.25				5.0
Central Vacuum	8.33	4.165	2.01				5.83
Source	K.G.	K.G. + ρ	K.G. + ρ	Dalen et al. (2017)		Seldal (2014)	Syversen and Schefte (2007)

*Estimated with the use of textile waste density EPAV (n.d.) and load capacity of 17.5 m^2 (Nilsson 2010).

K.G. = Kirsten Grevskott (BIR privat AS)

Table C.4: Estimated km per trip for each technology and waste type in reference system A.

	Km/trip				
	RW	P&C	Plastics	G&M	Textiles
Bins	94.4	34.5	168.6		
Containers	23.4	8.6	31.4	34.9	34.9
Underground	157.3	57.5	210.8		
Mobile Vacuum	87.4	31.9	117.1		
Central Vacuum	34	34	34		

Table C.5: Estimated tonne-kilometres for each waste type.

	Tkm/year					
	Reference Scenario			Food Sorting Scenarios		
	RW	P&C	Plastics	FW in SB	RW w/o FW	FW + RW
Bins	2.41E+06	2.09E+05	1.39E+05	3.67E+05	1.74E+06	2.27E+06
Containers	6.97E+04	6.05E+03	3.03E+03		6.97E+04	
Underground	5.12E+04	4.45E+03	2.22E+03		5.12E+04	
Mobile Vacuum	1.55E+05	1.34E+04	6.71E+03		1.55E+05	
Central Vacuum	3.44E+04	8.16E+03	1.11E+03		3.44E+04	
Total	2.72E+06	2.41E+05	1.52E+05	3.67E+05	2.05E+06	2.27E+06
G&M	7.37E+04					
Textiles	3.05E+04					

Table C.6: Transport energy consumption results for all scenarios (System B).

#	Reference	Bergen	Oslo	Voss
		MWh		
X12	0	6 236	6 236	6 236
X13	0	1 122	1 122	1 122
X14	12 292	5 921	5 921	5 921
X35	0	0	18 476	894
X46	529	452	452	452
X510	0	3 210	0	0
Total	12 822	16 942	32 207	14 626

Appendix D Process Energy

Table D.1: Estimated calorific value (LHV) for the waste fractions present in the system.

Fraction	**LHV** *KJ/kg*	**Source**
P&C	6434.85	
Plastics	20146.95	
Glass	-73.35	
Organic waste	1905.3	
Textiles	11794.2	Hulgaard and Vehlow (2010)
Other combustibles	7650	
Other non-combustibles	-245	
Wood	9310	
Metals	-147	
Garden waste	6445.3	Komilis et al. (2012)
HWEEE	16106.5	Estimated

Table D.2: Energy use input for the processes in system B.

Process	Name (energy carrier)	kWh/ton	Source
1	Collection	0	
2	Optical Sorting (el)	9.34	Seldal (2014)
3	Pre-treatment FW (el)	26.4	Khoshnevisan et al. (2018)
4[a]	Incineration (el)	4.61	BIR AS
4	Incineration (diesel)	3.04	BIR AS
5	Biogas production (el)	75	Bernstad and la Cour Jansen (2011)
5	Insect rearing (el)	12.9	Salomone et al. (2016)
6	Ash treatment	0	
7	Landfill	0	

[a] Similar accounted for process 13.

Table D.3: Process energy results for all scenarios in 2030 (System B).

#	Name	Reference	Bergen	Oslo	Voss
			MWh		
2	Optical Sorting	0	313.9	313.9	313.9
3	Pre-treatment	0	520.4	520.4	520.4
4	Incineration	615.4	505.4	505.4	505.4
5	Biological treatment	0	1 078.5	1 078.5	185.5
4	WtE production	-122 151.4	-115 039.9	-115 039.9	-115 039.9
5	Biogas production	0	-8 636.7	-8 636.7	0

Appendix E Emissions layer

Table E.1: Calculated emission contributions for each scenario in 2030.

	Reference	Bergen	Oslo	Voss
		kg CO_2-eq/inhabitant		
Transport	8.44	11.14	21.19	9.62
Incineration	102.24	101.74	101.74	101.74
Recycling	0.32	0.38	0.38	0.38
Avoided	-71.74	-85.85	-85.85	-76.44
Net emissions	39.26	27.41	37.46	35.27

Appendix F Cost layer

Table F.1: Calculated cost contributions for each scenario in 2030.

	Reference	Bergen	Oslo	Voss
		MNOK/year		
Collection	57.7	86.54	86.54	86.54
Sorting	0	32.66	32.66	32.66
Pre-treatment	0	8.01	8.01	8.01
Incineration	29.06	23.87	23.87	23.87
Biological treatment	0	15.03	15.03	4.46
Ash treatment	0	2.48	2.48	2.48
Transport (substrate)	0	1.75	5.89	1.30
Incineration revenues	-63.98	-52.54	-52.54	-52.54
Net costs	25.84	117.8	121.93	106.78

Appendix G Cost-efficiency

Table G.1: The cost and emission balance relative to the reference scenario, and corresponding annual cost-efficiency.

	Cost balance	Emission balance	Cost-efficiency
	MNOK/year	*kg CO_2-eq/year*	*NOK /kg CO_2-eq*
Biogas Bergen	91 951	4 768	**19.29**
Biogas Oslo	96 090	723	**132.99**
Insects	80 932	1 603	**50.46**