Tampere University

Krista Uusi-Kinnala

ASSESSMENT OF DECARBONIZATION POTENTIAL FOR CIRCULAR ECONOMY VALUE CHAINS USING SIMPLIFIED LIFECYCLE ASSESSMENT APPROACH

Methodological development and case studies in the Turku region

Master's Thesis
Faculty of Engineering and Natural Sciences
Professor of Practice Tero Joronen
University Lecturer Hannele Auvinen
Doctoral Researcher Abhishek Singhal
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ABSTRACT

Krista Uusi-Kinnala: Assessment of decarbonization potential for circular economy value chains using SLCA approach: Methodological development and case studies in Turku region
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The study aimed to develop and test a simplified life cycle assessment (SLCA) based tool for evaluating decarbonization potential in circular economy value chains. Climate change mitigation is one of the sustainable development goals. Creating carbon sinks, where the once-emitted CO_2 can be stored or absorbed, is one way to slow down the accumulation of CO_2 in the atmosphere. Being able to quantify a company's potential to create carbon sinks is crucial to meeting the EU and regional emission goals. There was no suitable tool available for the presented issue.

The study was executed in two parts: first, the SLCA-based tool was developed and then the tool was tested with three circular economy value chains and their improvement scenarios in the Turku region. The development of the calculator started with a literature review of the value chain in question, then the most impactful flows were determined and after that, the data collection was done. Finally, the method was applied to a spreadsheet calculator, and the global warming potential and decarbonization potential results were analyzed and interpreted. The assessment system boundary was set to gate-to-gate and only the parts of the process that would be changing were assessed in the comparison study.

Three case studies were assessed: municipal solid waste incineration (MSWI) with a monoethanolamine (MEA) carbon capture unit, a biogas plant with carbon capture from membrane separation, and a comparison of natural and recycled aggregate production. The recycled aggregate raw material was demolished concrete. For the MSWI and biogas case, the decarbonization potential scenarios were identified in collaboration with industry experts and for the aggregate case, both scenarios were already in use.

The annual decarbonization potentials of the MSWI, biogas, and aggregate cases were 19.6 kt CO_2-eq/a, 5.7 kt CO_2-eq/a, and 9.97 kt CO_2-eq/a respectively. When compared to Turku's annual emissions the studied value chains would decrease the emissions by 3%, 0.8%, and 1.5% respectively. The analysis showed that the MSWI case had the greatest decarbonization potential, but the operational emission from the MEA unit was 171 kg CO_2-eq/t CO_2 captured. The biogas case resulted in less decarbonization potential but the operational emissions of only compressing the already very concentrated CO_2 stream were only 8.4 kg CO_2-eq/t CO_2 captured. The aggregate case was more affected by the choice of renewable fuel than the replacing natural aggregate with recycled aggregate.

The SLCA approach could be used in a circular economy context and the performance of different improvement scenarios could be compared. The method was sensitive to assumptions due to its simple functionality. For future studies, the simplified method should be compared to full LCA, and the captured CO_2 management should be investigated in environmental assessment. The presented method was intended to be used in case-by-case comparison studies for a simple first calculation step to quantify the decarbonization potential. Applying the method requires still expertise to avoid misleading assumptions, but the method can present valuable knowledge when comparing different improvement scenarios. The decarbonization potential assessment showed, that the CE improvement scenarios can create a 5% decrease in the area's emissions.

Keywords: SLCA, Circular economy, GWP, MSWI, Biogas, brick and concrete waste, carbon capture

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TIIVISTELMÄ

Krista Uusi-Kinnala: Kiertotalouden arvoketjujen hiilinielupotentiaalin määritys yksinkertaistetulla elinkaariarvioinnilla, metodin kehitys ja tapaustutkimukset Turun alueella
Diplomityö
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Tutkimuksen tavoitteena oli kehittää ja testata yksinkertaistettuun elinkaariarviointiin (SLCA) perustuva työkalu, jolla voidaan tutkia potentiaalisia hiilinieluja kiertotalouden arvoketjuissa. Ilmastonmuutoksen hillitseminen on yksi kestävän kehityksen tavoitteista ja hiilinielujen lisääminen on yksi tapa hidastaa hiilidioksidin kerääntymistä ilmakehään. Hiilinielu adsorboi tai pidättää hiilidioksidia, joka muuten päätyisi ilmakehään. Jotta yritykset voivat saavuttaa alueelliset ja EU:n hiilineutraaliustavoitteet, pitää potentiaalisten hiilinielujen laajuus määrittää. Määritykseen sopivaa työkalua ei ollut käytettävissä.

Työ suoritettiin kahdessa vaiheessa: ensin kehitettiin SLCA metodiin perustuva työkalu ja toisessa vaiheessa työkalua pilotoitiin määrittämällä kolmen kiertotalouden arvoketjun hiilinielupotentiaali Turun alueella. Työkalun kehitys aloitettiin kirjallisuuskatsauksella tutkittavista arvoketjuista, seuraavaksi tunnistettiin vaikuttavimmat materiaalivirrat ja suoritettiin tutkimusaineiston kokoaminen. Aineiston avulla muodostettiin Excel-laskuri. Viimeiseksi ilmastovaikutus (GWP) ja hiilinielupotentiaali analysoitiin ja tulkittiin. Ilmastovaikutuksen arviointi rajattiin koskemaan vain tuotantoprosessia ja laskentaan otettiin mukaan vain yksikköprosessit, jotka muuttuivat kehitysskenaarion lisäämisen myötä.

Työssä tutkittiin kolme arvoketjua: hiilidioksidin talteenotto hyötyvoimalaitoksen savukaasuista monoetanoliamiinilla (MEA), hiilidioksidin talteenotto biokaasun membraaniprosessista kompressoimalla ja luonnonkivimurskeen sekä betonimurskeen tuotantoprosessien vertailu. Arvoketjujen mahdollisia parannuskohteita kehitettiin yhteistyössä alan ammattilaisten kanssa.

Hyötyvoimalaitoksen tapauksessa vuotuinen hiilinielupotentiaali oli 19,6 kt CO_2-ekv/a. Biokaasulaitoksen tapauksessa vuotuinen hiilinielupotentiaali oli 5,7 kt CO_2-ekv/a. Murskeiden tapauksessa varsinaista hiilinielua ei syntynyt, mutta vuotuinen päästövähennys oli 9,97 kt CO_2-ekv/a. Tutkimuksesta kävi ilmi, että hyötyvoimalaitos voisi tuottaa suurimman nielun, mutta MEA prosessin operoinnin päästöt olivat suuret, 171 kg CO_2-ekv/t CO_2 talteenotettu. Biokaasulaitoksen tapauksessa hiilidioksidi saatiin helpommin talteen pelkästään kompressoimalla membraaniprosessin CO_2-rikas poistokaasu ja talteenoton päästöt olivat vain 8,4 kg CO_2-ekv/t CO_2. Murskeiden tapauksessa huomattiin, että käyttämällä uusiutuvaa polttoainetta, saataisiin suurempi päästövähennys aikaan kuin vaihtamalla luonnonkivimurske betonimurskeeseen.

Yksinkertaistettua elinkaariarviointia voitiin hyödyntää kiertotalouden prosessien arvioimisessa ja kehitysskenaarioiden tehokkuutta voitiin vertailla keskenään. Koska metodi oli hyvin yksinkertainen, tehdyt oletukset vaikuttivat helposti tuloksiin. Tulevissa tutkimuksissa yksinkertaistettua metodia tulisi verrata perusteelliseen elinkaariarviointiin ja talteen otetun hiilidioksidin vaatimien prosessien ympäristövaikutus tulisi ottaa mukaan tutkimukseen. Työssä esitetty metodi oli tarkoitettu käytettäväksi tapauskohtaisesti yksinkertaiseksi ensimmäiseksi askeleeksi hiilinielupotentiaalin määritykseen. Metodin käyttäminen vaatii silti ammattitaitoa, jotta harhaanjohtavilta oletuksilta vältytään. Kaiken kaikkiaan metodi tarjoaa arvokasta tietoa vaihtoehtoisista kehitysskenaarioista ja tutkituilla kiertotalouden prosessien kehittämisellä saataisiin 5 % vähennys Turun päästöihin.

Avainsanat: yksinkertaistettu elinkaariarviointi, kiertotalous, ilmastovaikutus, hyötyvoimalaitos, biokaasu, betonimurske, hiilidioksidin talteenotto

Tämän julkaisun alkuperäisyys on tarkastettu Turnitin OriginalityCheck –ohjelmalla.

PREFACE

This Master’s Thesis was done in collaboration with Gasum, LSJH, Kiertomaa, and ILPO-project group. I want to thank all the experts that gave their time to this project.

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CONTENT

LIST OF SYMBOLS AND ABBREVIATIONS

LCA	Life cycle assessment
CE	Circular economy
SLCA	Simplified life cycle assessment
MSWI	Municipal solid waste incineration
GHG	Greenhouse gas
IPCC	Intergovernmental panel of climate change
EDP	Environmental product declaration
LCI	Life cycle inventory
GWP	Global warming potential
EF	Emission factor
SA	Sensitivity analysis
CC	Carbon capture
CCS	Carbon capture and storage
CCU	Carbon capture and utilization
C&DW	Concrete and demolition waste
MEA	Monoethanolamine
MS	Membrane separation

1. INTRODUCTION

Climate change mitigation is one of the 17 sustainable development goals. The Finnish government has set a goal to be carbon-neutral by 2035. This requires emission reductions in all sectors and the creation of carbon sinks. The policy includes also the promotion of a circular economy (CE). (Muurman *et al.*, 2021) Assessing the environmental impact of CE processes is crucial for achieving the climate targets at the business and government levels. With decarbonization potential, it can be assessed how much CO_2-eq emissions are avoided with an improved process by comparing the global warming potentials (GWP) of the studied scenarios (Zhang *et al.*, 2020).

CE is an economic model that focuses on sifting consumption from owning to using services. In the ideal scene, materials have longer life cycles and the products are not discarded but recycled back to use. (Sjöstedt, 2018) By Ellen McArthur Foundation (2020) CE follows three principles: designing out waste and pollution, keeping materials in use, and regenerating natural systems.

There is yet no harmonized methodology to assess whether CE reduces environmental and social impacts. Life cycle assessment (LCA) can reveal the issue of valuing circulation over true environmental benefits and help determine the best options. CE strategies often assume that keeping materials within the economy in use or through cycling loops that require energy is the most valuable option. Environmental assessments do not advocate any specific strategy but provide an understanding of the environmental impacts related to different options. (Peña *et al.*, 2021)

The challenges of full LCA are known in the academic community and there have been actions to formulate a simplified life cycle assessment (SLCA) method to make LCA more accessible (Wu *et al.*, 2015; Beemsterboer et al. 2020; Gradin and Björklund, 2021). There has been a demand to find methodologies that are practical and fast enough to use, and accurate enough to provide valuable information to support the decision-makers. The first simplification methods have been presented already when the LCA calculations were made manually. (Gradin and Björklund, 2021) Because the goals of LCA or environmental assessments can be very wide, there is no consistent guidance on how LCA can be simplified for different applications. Wang *et al*. (2021) used SLCA to compare MSW management methods and they found that only 3.3% of all material flows studied in full LCA contributed to ≥95% of the environmental impact of the studied scenarios.

Applying LCA to CE raises questions that are not easy to answer consistently as Peña *et al.* (2021) and Haupt and Zschokke (2017) pointed out. How are credits of recycled material quantified and which actor in the value chain can claim the benefit? How are the quality differences between raw materials and products assessed when comparing linear economy products? The issue with varying quality of material flow in CE value chains was identified as one main limitation of the LCA method when studying plastic recycling (Schwarz *et al.*, 2021). How to account for multifunctionality and heterogeneous material streams? Above mentioned questions were discussed in the conference on LCA, but panelists did not have solutions to these methodological questions that limit the use of LCA in CE value chains. (Haupt and Zschokke, 2017) Generally, CE value chains include multiple companies with different functions. Assessing the full life cycle of the value chain will become more challenging when the needed inventory data is distributed among multiple companies.

Carbon capture (CC) has been identified as one solution in climate change mitigation by the Intergovernmental panel of climate change (IPCC) (Abanades *et al.*, 2005). CC can be applied to large point sources and the captured CO_2 can be eighter stored long term or utilized directly. One of the mature CC methods is amine scrubbing and it can be applied to flue gases, for example from waste incineration. Membrane processes for CC have been used on a commercial scale for syngas separation and on a demonstration scale for coal power plants. (Bui *et al.*, 2018)

Because of the nature of CE, it was interesting to study how one methodology applies to multiple value chains. The assessed value chains were chosen from one business area from the Turku region. Municipal solid waste (MSW) is still generated and treated with municipal solid waste incineration (MSWI) plants that cause 1% of the total CO_2 emissions in Finland (Tilastokeskus, 2019; Chen *et al.*, 2020). The CC from the MSWI plant was chosen to be one studied value chain. The waste framework directive recommends that the biogas process is utilized more in the waste management of MSW and it was found that the CO_2 off gas from the biogas process could be captured, so the biogas process was chosen to be the second assessed value chain (DIRECTIVE 2018/851, 2018; Yang *et al.*, 2020). The last studied value chain was chosen to be recycled aggregate from concrete waste. The government's goal to decrease the emissions from the construction sector directed the selection of the last studied value chain (Muurman *et al.*, 2021).

The full LCA requires resources that not all CE companies have, but it is still crucial to assess the environmental impacts during process development. Because there is no consistent guidance on how to conduct an SLCA for different CE value chains, step-by-step instructions are needed (Wu et al., 2015; Gradin and Björklund, 2021; Wang, Levis and Barlaz, 2021). This study aims to present and conduct an SLCA method for circular economy value chains. The goal is formulated to following research questions:

- How SLCA can be applied to CE value chains when developing an easy-to-use tool for decarbonization potential assessment of future improvement scenarios?
- How the SLCA method applies to different CE value chains?
- What is the significance of the decarbonization potentials of the studied value chains compared to original processes, and Turkus emissions?

The study of decarbonization potential is intended to help businesses critically assess the current process and evaluate which improvement would be environmentally beneficial. The SLCA approach is studied and tested with three case studies: MSWI, biogas, and recycled aggregate. These value chains and the methodology direction (Lunden, 2021) are selected earlier in the ILPO project. The theory section discusses the fundaments of LCA, SLCA, and studied value chains. It presents an overview of previous studies on CE value chains. Based on the studies covered in the theory part, the methodology is described in section 3. It includes the excel tool development and the inventory for MSWI, biogas, and aggregate cases. The decarbonization potential of the value chains is discussed in section 4, where the significance of the improvement scenarios is compared to Turku’s annual emissions. The functionality of the used method is also discussed in section 4.

2. THEORETICAL BACKGROUND

All environmental assessments have a common goal of quantifying environmental impacts. How the quantification is made differs greatly between studies that have different goals. This chapter presents a short description of different environmental assessment methods and focuses on the commonly used LCA method. The application of LCA to MSWI, biogas, and aggregate value chains is discussed. Lastly, carbon capture storage and utilization are discussed.

There are many different methods and terms for quantifying environmental impacts: LCA, global warming potential (GWP), carbon balance, SLCA, carbon footprint, and carbon handprint. LCA considers all the relevant environmental impacts that are generated in the full life cycle of the studied product or process. GWP is one of the most known impact categories that quantify the impact of greenhouse gases (GHG): CO_2, CH_4, and N_2O, in a specific timeframe (Amoo and Layi Fagbenle, 2020). The accountable and independent future-oriented fund, Sitra defines carbon balance as a net sum of carbon emissions and storage. It is usually discussed in the context of forests and wetlands. (Sitra, 2021) Terms streamlined or simplified LCA is used for life cycle assessment that applies various methods that reduce data collection time and fill data gaps (Pascual-González et al., 2016; Wang, Levis and Barlaz, 2021). Carbon footprint refers to greenhouse gas emissions caused by human activity and the term is closely related to GWP. Carbon handprint refers to climate benefits or avoided emissions that can be achieved, for example, by storing biogenic carbon. (Sitra, 2021; Troy, Schreiber and Zapp, 2016)

The Intergovernmental Panel on Climate Change (IPCC) has conducted a standardized method for collecting and reporting national GHG emissions outlined in the Kyoto Protocol. The IPCC method reports only direct emissions from different sectors within the studied area and excludes indirect emissions. For example, GHG emissions caused by the production of imported products are not included in the IPCC emission report. (Fath, 2018) (2376 Volume 4 p.254) Statistics Finland collects the National GHG inventory by IPCC recommendations and reports it to UNFCCC (Statistics Finland, 2021).

Environmental product declaration (EPD) is an LCA-based ecolabel that indicates the environmental performance of a specific product. There are product category rules to keep the different EDPs comparable and consistent. EDP can also include some information outside the scope of LCA if the information is relevant to that product.

(Curran, 2015) Both IPCC emission factors and EPD data can be used as sources for environmental assessments if there is no process data available (Henriksen *et al.*, 2019).

2.1 Principles of Life Cycle Assessment

Life cycle assessment (LCA) is an analytical method for decision support and part of environmental management tools (ISO, 2006a; Christensen, 2011). In the LCA study, the environmental impacts of the products are quantified for the whole duration of their life cycle. LCA is the industry standard in the environmental assessment of products and processes. The International Organization for Standardization has published the international standard about LCA principles and framework ISO 14040 and a more detailed standard about requirements and guidelines ISO 14044.(ISO, 2006a, 2006b)

ISO 14040/44 standards state that LCA can be used for identifying development points, informing decision-makers, selecting relevant environmental indicators, and marketing (Figure 1) (ISO, 2006a, 2006b). Even though the LCA methodology is standardized, the method allows the practitioner to make decisions that will affect the result. That makes different LCA studies challenging to compare to each other. (ISO, 2006b; Feiz *et al.*, 2020)

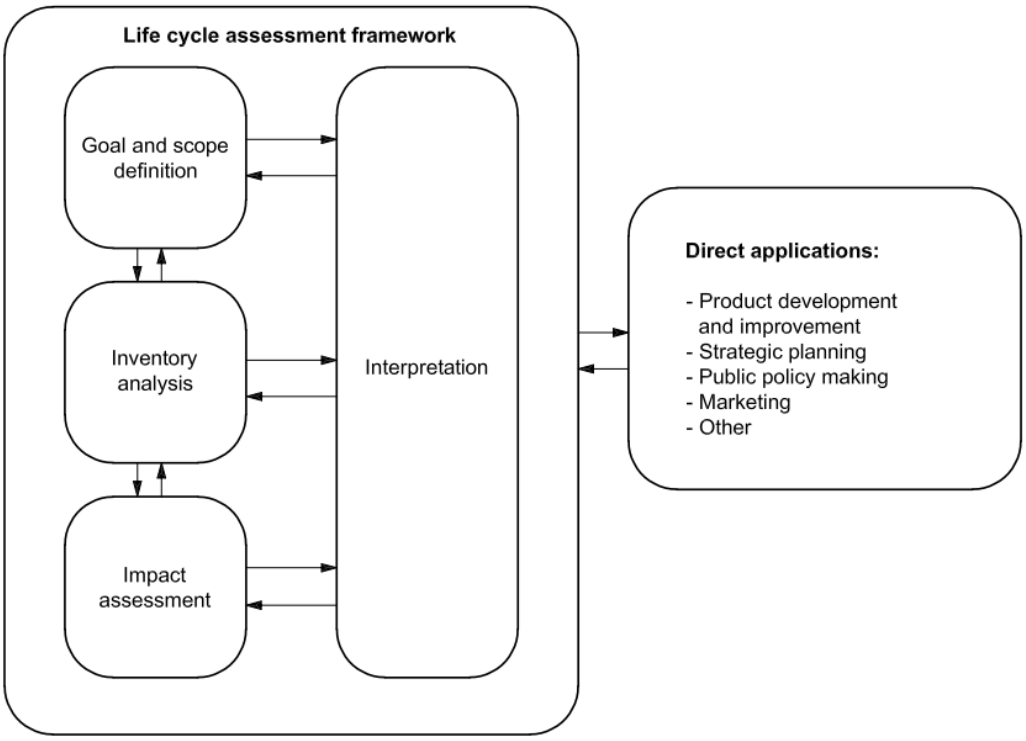

Figure 1. *LCA is an iterative method and it can be used in multiple applications, a figure from* (ISO, 2006a)

LCA study consists of four phases: goal and scope definition, inventory analysis, impact assessment, and interpretation. In the goal and scope definition phase, the study's objectives are reported so that all decisions are disclosed and justified based on the purpose of the study. Later in the inventory analysis phase, all the inputs and outputs are reported concerning the functional unit. In the impact assessment phase, the potential environmental impacts associated with inputs and outputs are quantified. In the last phase, the results are analyzed and the relevant aspects of the results are disclosed to avoid misleading conclusions. (ISO, 2006b; Curran, 2015) LCA is recommended to be performed in two phases: a preliminary or screening phase that is best done by hand or spreadsheet. The second more detailed phase of LCA is recommended to be done with LCA software and check energy and mass balances. (Jolliet *et al.*, 2015)

2.1.1 Goal and scope definition

In the goal definition, the intended application, reasoning, intended audience, and publicity of the study should be stated based on ISO 14044. In scope definition, functional unit and system boundary is defined based on the main purpose of the studied process. Also, allocation procedures, impact assessment methodology, and impact categories are reported in the first phase. (ISO, 2006b)

The functional unit should include all the relevant variables of the product or system. For example, a functional unit for concrete would be linked to volume or mass and compressive strength, because there is variability between concrete types. The functional unit represents the main function of the studied process, for example in MSWI plant the functional unit can be the energy amount produced in MWh or the waste mass treated in tons. Choosing a well representative functional unit in CE value chains is challenging because CE processes usually have multiple co-products. (Curran, 2015)

Why the study is made and for what use, affects greatly how the study is conducted. There can be a significant difference in the details of the studies that are made for environmental product declaration certificates to studies that are made for screening different options for process development. (Pérez *et al.*, 2018; Gradin and Björklund, 2021; Peña *et al.*, 2021; Wang, Levis and Barlaz, 2021) For all LCA studies reporting goal and scope is very important to the repeatability of these studies. Without thorough goal and scope definition, different LCA studies are not comparable, because only studies that are made with the same assumptions, scope, and functional units are comparable to each other. In the LCA study, there are numerous points where value-based assumptions or choices can be made, and reporting all made decisions also improves the transparency of the study (Curran, 2015).

2.1.2 Inventory analysis

In inventory analysis, the needed input and output flows within the system boundary of the studied process are collected. These flows can be raw materials, waste, products, energy, or economic flows. (ISO, 2006b; Curran, 2015) The goal of the inventory analysis is to collect all the needed data from the studied process defined in the goal and scope phase of the study. ISO 14044 recommends supporting methods such as process flow diagrams to ensure complete and organized data collection. Reporting all the influencing factors to the process flows and operating conditions is important for the repeatability and comparability of the study. Also, reporting the data collection methods, irregularities, and calculation techniques is essential for the study's transparency. (ISO, 2006b)

All unit processes and material flows that affect the conclusion of the study should be included, but the study's level of detail defines how broad the system boundary needs to be (ISO, 2006b). The amount of data needed for a full LCA study can become massive and not all flows affect the study's conclusion significantly, so LCA practitioners can apply cut-off criteria to reduce the number of flows (ISO, 2006b; Wang, Levis and Barlaz, 2021). The ISO 14044 standard presents 3 cut-off criteria based on mass, energy, and environmental significance. Leaving out flows that contribute less than a defined percentage can cause uncertainty, so the standard requires sensitivity analysis (ISO, 2006b).

Inventory data can be site-specific process data, modeled data from process plans, industry averages from databases, or also secondary data from the literature. (Curran, 2015) The quality and representativity of inventory data will affect straight to the quality of results. LCA handbook recommends documenting the time, location, characteristics of the measured processes, incoming transportation type and distance, co-products, possible calculations for needed data, and possible irregularities or assumptions related to the LCI data (ISO, 2006b; Curran, 2015). When the knowledge about the studied process grows, the before-defined scope and goal need to be adjusted to suit the available data and knowledge (ISO, 2006a).

In the LCA study, all the inventory flows are scaled to a functional unit that is defined in the goal and scope phase. Multifunctionality can be a challenge in LCA study. LCA methodology is developed to suit easily single-function processes where only one product gets all the environmental impacts (Jolliet *et al.*, 2015). Allocation is dividing the process input and output flows among the useful outputs based on the allocation key. There are multiple allocation methods and the LCA practitioner must choose without field-specific guidance. This choice can affect significantly the comparability of the LCA studies. (Timonen *et al.*, 2019) ISO 14044 presents a stepwise procedure for general allocation methods. The first allocation should be avoided by dividing the process into sub-processes or expanding the system to include additional functions for the co-products. If allocation can not be avoided, process flows should be designated to different products based on physical relationships, for example, emission can be distributed proportional to the mass of two co-products, even though it is not possible to identify which product caused the specific emission. For example, in the biogas system, the mass-based system allocates 5% of emissions to biogas and 95% to digestate (Timonen *et al.*, 2019). If allocation based on mass or energy can not be done, then the economic value can be the allocation key. (ISO, 2006b) In circular economy processes allocating is difficult, because there is rarely any suitable key, to distribute the emissions equally (Timonen *et al.*, 2019). There is no standardized guidance for choosing LCA modeling approaches and how they should be used, the only requirement is to report and justify the suitable modeling approach to the goal and scope of the study (ISO, 2006b).

2.1.3 Impact assessment

In the impact assessment phase, all the inventory data is divided into impact categories and impact indicators are calculated (Figure 2). Now a large amount of inventory data can be presented more clearly, and the magnitude and significance of the potential environmental impacts can be evaluated. (Curran, 2015)

Environmental impacts are divided into several different impact categories such as land use change, acidification, and eutrophication potentials. For example, land use change describes how changing the natural state of land can lead to extinct species. (Acero, Rodriguez and Ciroth, 2017). The value-based inclusion of different impact categories is done in the goal and scope phase. The quantification calculation is simple multiplication between the activity data from inventory and the characterization factor for specific impact categories. (Curran, 2015)

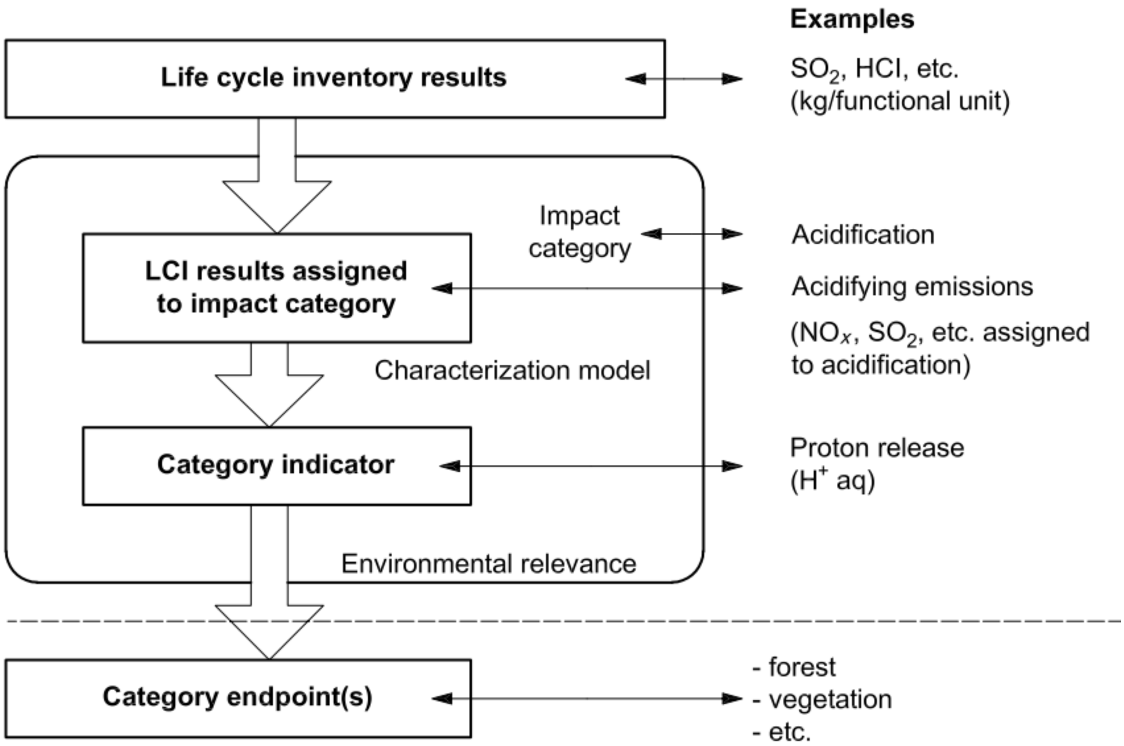

Figure 2. *Relations with environmental effect and LCA calculation method (from ISO 14044)*

Global warming potential (GWP) is the most known and widely used midpoint impact category. It is adapted to LCA from the IPCC methodology. (Curran, 2015) IPCC emission factors (EFs) are often used to calculate GWP with basic equation 1

$$GWP = \sum(EF * m_{emission}), \quad (1)$$

where EFs are from literature or IPCC databases, the mass of emission is from inventory and the unit of GWP is kg/CO_2-eq. There are different EF tiers. Tier 1 EFs are global averages that are intended to use in national or continental-level calculations. Tiers 2 and 3 are national and regional level EFs that are collected and reported to UNFCCC based on the Kyoto protocol. (Peter *et al.*, 2016) In Finland national emission factors are collected by Statistics Finland and the factors and cumulative GHG emissions are reported every 4 years (Statistics Finland, 2021). EFs are affected by local climate, used technologies, energy sources, and for example, distances between the production plant and raw material. Tier 1 EFs generally overestimated emitted GHG amounts compared to higher tier calculations and field tests (Peter *et al.*, 2016).

CO_2 emissions can be divided into two categories: fossil and biogenic carbon emissions (Curran 2015). Biogenic CO_2 emissions are released from biomass. In this study, biogas process feed produces biogenic CO_2. Fossil carbon dioxide is released from products that contain fossil-based components. Plastics and synthetic textiles are the main waste category in MSW that releases fossil carbon emissions when combusted (Christensen and Bisinella 2021). In carbon footprint calculations only fossil carbon emissions are calculated as emissions because biogenic carbon is part of the natural relatively fast carbon cycle when new growing biomass binds the CO_2. Fossil carbon emissions increase the total carbon amount in the atmosphere and amplify the climate change effect. (Curran, 2015)

2.1.4 Interpretation and uncertainty analysis

In the interpretation phase, the results of inventory and impact assessment are analyzed, and then conclusions are drawn. The main goal of this step is to give credibility to the LCA results in a transparent way that is useful to the decision-makers. (Curran, 2015) In the interpretation phase, based on ISO 14044, the significant issues about the results are discussed. The discussion includes an evaluation of the completeness, sensitivity, and consistency checks. Lastly, conclusions, limitations, and recommendations are presented. (ISO, 2006b)

The quality of the LCA can further be increased by including and discussing various uncertainties with the results. If ISO 14040/44 recommended checks will show incompleteness or that some decisions are crucial, there is a possibility to re-evaluate the inventory or assumptions and improve the LCA study. Sources of uncertainty can be the data source or collection method itself, assumptions, and calculations made in the study. Combining data from very different sources can also cause uncertainty. (Curran, 2015)

Simple sensitivity analysis (SA) can be done by changing parameters systematically and assessing how much the input change affected the results (Curran, 2015). SA can be also done to record the effects of assumptions and methodological choices on the results (ISO, 2006b). Contribution analysis tells which inventory flow or emission factor has the greatest effect on the LCA result. (Curran, 2015) Contribution analysis reveals more about the actual impacts than the final GWP. For example, an incineration plant might have low GWP but the rewards of renewable energy counterbalance the absolute fossil CO_2 emissions from plastics. (Clavreul, Guyonnet and Christensen, 2012) A completeness check should be done to ensure that all relevant data is taken into the study. (ISO, 2006b)

2.2 Streamlining methods for LCA

Streamlined LCA was developed when Svenson and Ekvall studied cost-effective ways to compare two products and they stated that streamlining is a solution for cost-effective

LCAs (Svensson and Ekvall, 1995). The main goal of simplification, and streamlining strategies is to reduce the data work involved in LCA (Beemsterboer, Baumann and Wallbaum, 2020). ISO 14040 states that simplifications and other practices are out of the scope of ISO standards so there is no general guidance for streamlining methods (ISO, 2006a). The most impactful step of streamlined LCA is to identify the most affecting unit processes in the life cycle that need to be studied without significantly affecting the overall results (Hur *et al.*, 2005). There is a risk of leaving out flows that have a significant effect on the overall SLCA result. For example, N_2O emissions have high EF (289 kg CO_2-eq/kg N_2O) so even small amounts have a significant effect when compared to CO_2 (IPCC 100 yrs).

Listed simplification methods:

1. Narrowing system boundaries to include only the most necessary processes. Simplification from cradle to grave -system to the gate to gate - system will reduce the time that is used for sourcing the data. Usually, process data is more easily available than data from upstream or downstream data. At least data for the company's own LCA is easier to find inside the same company than data from other companies. (Gradin and Björklund, 2021)
2. Limiting studied impact categories will reduce the need for data and time. Though, there is also a risk of burden-shifting if significant impact categories are left out. (Beemsterboer, Baumann and Wallbaum, 2020)
3. Mixing qualitative and quantitative data. Changing the study from quantitative to qualitative can ensure that the study can be made without all the exact data. This rough method can be used in qualitative screenings in the early design phase of products. (Gradin and Björklund, 2021)
4. Using databases to fill data gaps with proxy data. Proxy data can be found for example, in the Ecoinvent database. Proxy data is chosen based on physical, chemical, or functional similarities. (Gradin and Björklund, 2021)
5. Applying a suitable cut-off in the LCI phase will limit the complexity of the impact assessment. It is important to set the cut-off so that all relevant flows are still in the impact assessment stage. For example, in GWP nitrous oxides have a characterization factor of 289 (IPCC 100 yrs.), so it is necessary to include N_2O emissions that are all less than 1 m-% in the study. When assessing toxicity, it is also very important to apply truly suitable cut-off criteria. (Gradin and Björklund, 2021)

6. Comparative LCA with omitting identical parts of the process results in a relative difference. (Gradin and Björklund, 2021)

Beemsterboer *et al.* (2020) identified standardization, automation, and qualitative expert judgment as other simplification strategies. Standardization offers structure and guidance for methodological choices. Standardized LCA tools offer pre-structured inventory and impact assessment models, that cut out the need for data collection and calculation of the emission and characterization factors. The spectrum of standardized LCA tools differs from very simple spreadsheet tools to more specialized LCA software. The accuracy of LCA tools varies and depends mainly on the fit between the tool and the studied product system. These tools can be developed for specific product systems, but the results from the simplified LCA tool are not the replacement for detailed LCA studies. Automation in LCA can be questionaries that use prior inventory data to estimate and show the effects of design decisions. Connected material and emission databases in CAD software will cut the additional work of comparing different material choices and functionality of the design. Qualitative expert judgment is used as a simplification method to reduce the time needed for full quantification and in some cases, qualitative assumptions must be done before quantification is possible. (Beemsterboer, Baumann and Wallbaum, 2020)

Because there are different simplification strategies, the methods used for conducting SLCA methods are case-specific (Niero *et al.*, 2014). Wang *et al.* (2021) used full LCA and contribution analysis to select critical impacts and inventory flows that were used in SLCA. They found that 3.3% of the inventory flows contribute ≥95% of the overall environmental impact in a study about municipal solid waste management systems. (Wang, Levis and Barlaz, 2021) This shows that assessing only a small portion of the inventory flows can give a reliable estimation of the overall impact.

Wu *et al.* (2015) present 3 step process for conducting SLCA for C&D waste. They used a literature review of similar processes to conduct preliminary SLCA. Then Monte Carlo analysis was applied to measure and minimize the uncertainty. Lastly, contribution analysis was made to identify the most impactful flows and also the activities that have the highest variance in the result. (Wu *et al.*, 2015) Wu *et al.* (2015) replaced the full LCA of the specific case with a literature review to save time.

The parametric model for SLCA by Niero *et al.* (2014) follows ISO 14040-44 standards and it is made for wood pallet manufacturing. The parametric model solved the problem of making LCAs for each pallet design individually. The parametric model had independent and dependent parameters that were calculated with mathematical correlations between independent parameters. The parametric model takes in the characteristics and mass of used wood, nails, and the final product, and lastly the transport distance to the customer. This type of parametric model allows companies to calculate the environmental impacts of their similar products easily when the first investment of making the parametric model is done. (Niero *et al.*, 2014)

Based on these three SLCA studies by Nierro *et al.* (2014), Wu *et al.* (2015), and Wang *et al.* (2021), it can be that SLCA can be done from different starting points and with varying simplification methods (Figure 3). Because LCA is an inherently iterative method, the application of simplifications can be done in different parts of the process. Even though researchers have recommended clear documentation of simplifications with consistent terminology it is still lacking (Gradin and Björklund, 2021). Because of the case specificity of simplifications, it is challenging to conduct very detailed and clear guidance for all circumstances.

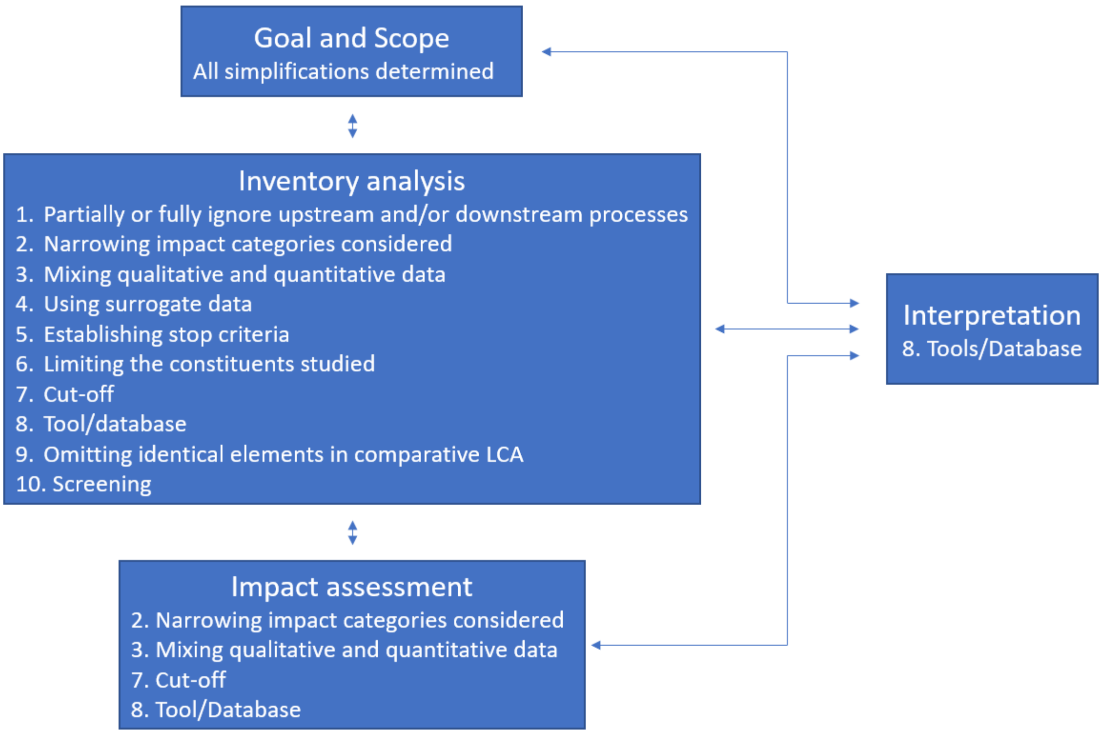

Figure 3. *Simplification methods and their application phase in the iterative LCA process. All simplifications must be in line with the goal and scope definition. If the original goals can not be met due to, for example, a lack of data, the goal and scope must be reassessed (Modified from* Gradin and Björklund (2021)*).*

Circular economy value chains usually have multifactional processes that are difficult to credit with conventional LCA methods that do not recognize system expansion or crediting of recycled materials (Curran, 2015). Comparability of different processes will be an issue that needs very close attention and standardization has been the way to increase comparability but the assumptions made in streamlined studies are difficult to standardize (Fath, 2018). The recycled product must be as useful and reliable as the original linear product so that these are truly comparable.

Even though SLCA methods will provide a reduction in time, data management, and cost of an environmental assessment, the results of these SLCAs should not be used in comparative assertions disclosed to the public. (Niero *et al.*, 2014; Pelton and Smith, 2015) Also limiting only to one impact category will not give the full picture of the environmental impacts of the studied process (Beemsterboer, Baumann and Wallbaum, 2020). For this application where the goal is to find carbon sinks and identify possibilities to meet cities' carbon emission reduction targets, GWP is the most important indication of the process.

2.3 Circular economy value chains and Life Cycle Assessment

In the next chapters, the theoretical background of the studied value chains is presented. First, the principles of the MSWI, biogas, and recycled aggregate production processes are discussed. Then the related environmental assessment studies are reviewed and lastly, the improvement scenarios for the processes are discussed.

2.3.1 LCA studies on MSWI and integrated CC

Municipal solid waste (MSW) management has been developing from landfilling to energy recovery from waste (Chen *et al.*, 2020; Statistics Finland, 2020b). The amount of waste that is incinerated is increasing with the rise of environmental consciousness and with the rise of waste produced in the world (Chen *et al.*, 2020). Waste generation is generally higher in developed countries that have more resources to manage waste (Kaza *et al.*, 2018). MSW generation by Finnish municipalities was over 1.6 million tons in 2020 (Statistics Finland, 2020b). Incineration of MSW caused 0.63 million tons of CO_2 eq emissions in 2017, which is about 1% of the total carbon emissions (56.5 million tons) (Tilastokeskus, 2019). In Nordic countries such as Finland and Denmark, the MSW is recycled, and the composition of the incinerated waste depends on the citizen's recycling habits and opportunities (Bisinella *et al.*, 2021). MSW in Finland is still containing 32 % biowaste even though there is a separate collection for the biowaste (Figure 4).

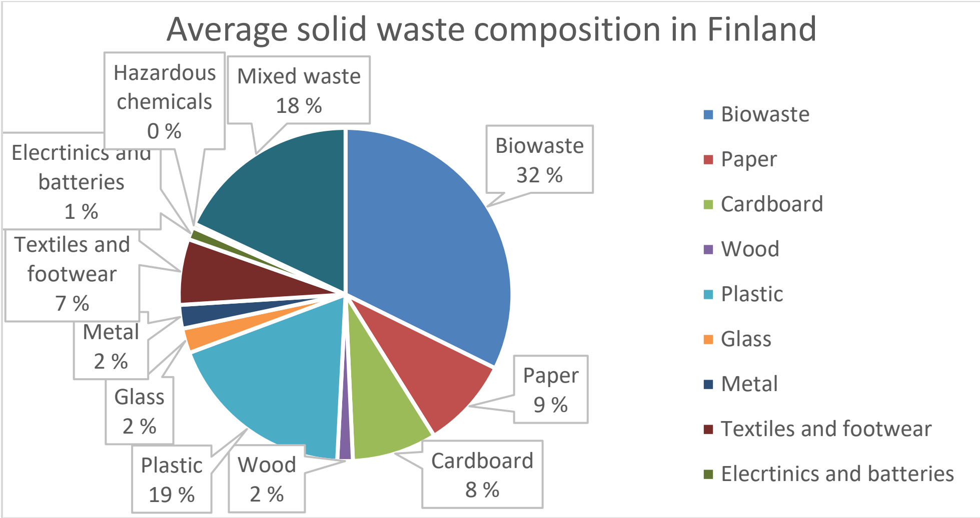

Figure 4*. Average solid waste composition in Finland.* (KIVO, 2022)

Typically, municipal incineration (MSWI) plants have a waste bunker or reception hall where waste is delivered. From the hall, waste is directed to the furnace where it is incinerated in regulation ordered 850-1100 C. (Christensen, 2011) In the furnace waste goes through drying, devolatilization, and char burning. Produced flue gas is cleaned with the EU directive required air pollution control system from particles, nitrogen oxides, heavy metals, and dioxins (Bisinella *et al.*, 2021). The remaining flue gas contains 6-10 vol-% of CO_2, which can be captured with a post-combustion carbon-capturing unit before releasing gas into the atmosphere (Bodénan and Deniard, 2003; Bui *et al.*, 2018). Noncombustible waste exits the process as gravel-like bottom ash or as light and fine fly ash. Energy can be recovered from the flue gases as heat for district heating, process steam for industrial purposes, and electricity (Figure 5). (Christensen 2011)

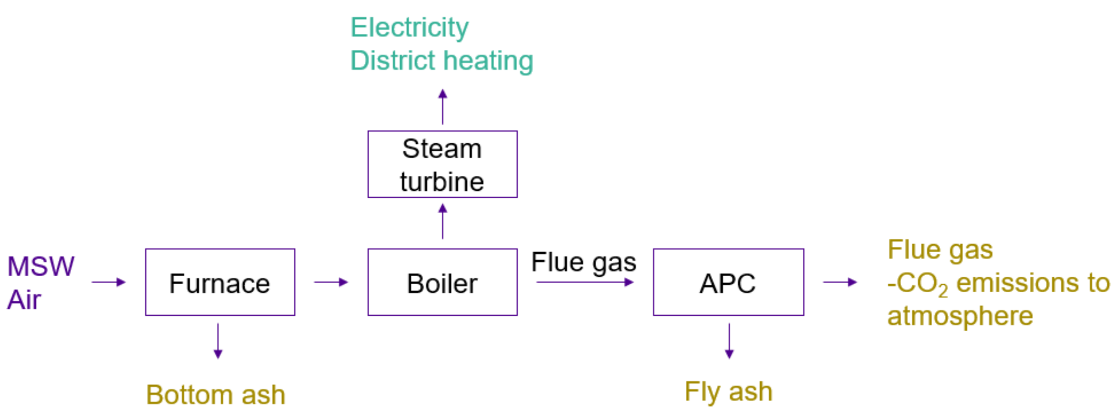

Figure 5. *Simplified schematic of the municipal solid waste incineration process. Air pollution control (APC)*

The waste management hierarchy states, that 1st priority is to prevent waste generation. Re-use and material recycling are prioritized over energy recovery from waste.

(DIRECTIVE 2008/98/EC, 2008) MSWI is at the bottom of the waste management hierarchy, but incineration plants are still needed for the waste that can not be treated otherwise. In 2018 70% of the world's waste ended up in dumps and landfills (Chen *et al.*, 2020). Chen *et al*. predict, that waste incineration will rise as a waste management method from 39% by 2050. Based on the European commission's Waste framework directive (2018), the goal is to increase recycling and move away from incinerating biowaste. Composting and biogas production are more recommended methods for biowaste management. That means that in the future the goal is to decrease the biogenic carbon content in MSW, but the need for decreasing the volume of waste with MSWI will be prominent.

LCA has been used as a decision-making tool for comparing different waste management systems, and in those studies, MSWI has been studied as one scenario (Hupponen, Grönman and Horttanainen, 2015; Pérez *et al.*, 2018; Khan and Kabir, 2020). Hupponen *et al.* (2015) studied how different locations and MSWI processes affect GWP and the costs of the system in Finland. Pérez *et al.* (2018) compared different combinations of MSWI, landfilling, recycling, composting, and biogas process to each other and found that the scenario with the most recovered products had the best GWP. Khan and Kabir (2020) compared waste-to-energy scenarios and concluded that the biogas process and MSWI are the most and least sustainable technologies, respectively. In evaluating waste management systems multiple specialized LCA-based models fulfill different purposes. EASETECH is specialized in complex systems with heterogeneous material flows (Clavreul *et al.*, 2014). CO2ZW is an Excel-based tool, that takes in waste composition, transportation distance, and treatment method, (recycling, biological treatment, incineration, or landfilling) and results in GHG emissions of waste management scenarios for European policymakers (Sevigné Itoiz *et al.*, 2013). These tools focus on guiding the waste management method decisions, not improving, or decarbonizing the MSWI process itself.

Bisinella *et al.* (2021) studied MSWI plants with a retrofitted monoethanolamine (MEA) carbon-capturing unit. Applying CCS to the MSWI plant will decrease the climate change impacts of the plant by 700 kg CO_2-eq/t waste incinerated, with CC efficiency of 85%. Sensitivity analysis showed that capturing efficiency was the main factor affecting the overall results. With renewable energy systems, the carbon-capturing efficiency was even more likely to affect the results. (Bisinella *et al.*, 2021) Utilization and storage method of captured CO_2 will affect the GWP benefits of CC. Local utilization would result in 700 kg CO_2-eq/t waste GWP reduction if the substituted CO_2 is fossil-based. The largest GWP reductions (2000 kg CO_2-eq/t waste) resulted in producing feedstock chemicals such as dimethyl ether or formic acid, but the synthesis requires energy-intensive hydrogenation of CO_2 so the benefits can be achieved only with non-fossil energy systems. (Christensen and Bisinella, 2021)

Pour *et al.* (2017) studied the environmental and economic feasibility of MSWI with the MEA CCS system in the Australian and global context. The study utilized the National inventory report, Ecoinvent 3, AusLCI database, and SimaPro. CO_2 storage and transportation were included in the assessment. Pour *et al.* (2017) stated that applying CCS to the MSWI plants would remove 700 kg CO_2-eq from the atmosphere per 1 t of MSW incinerated. Electricity price is higher from MSW with a CCS system than from coal, which means that monetary compensation for more environmentally friendly processes has a significant impact on the feasibility of those technologies (Pour, Webley and Cook, 2017).

Tang *et al.* (2018) conducted an environmental and economic assessment of the MSWI process with different CC methods; MEA, pressure swing adsorption, and oxyfuel combustion. They calculated impacts to the ReCiPe midpoint scores that are not comparable to GWP reductions that were used in other studies, but the CC with the MEA process decreased the climate change by 2 midpoint score points. Mass-based CO_2 emission reduction with MEA CC was 560 kg CO_2-eq/t of waste incinerated. Tang et al. (2018) also assessed the construction resources needed for building an MEA unit and the mass-based need was only 0.85 % more materials compared to all materials needed when building the MSWI plant. Based on their economic assessment carbon pricing should be 34-46 $/t of CO_2 emissions avoided to make CC attractive from an economic point of view (Tang, You and Frederick, 2018)

2.3.2 LCA studies on biomethane production and integrated CC

Biogas is produced naturally by the microbial decomposition of biomass in absence of oxygen (Schnürer, 2016). Biogas consists of mainly methane (50-75 Vol.-%) and carbon dioxide (25-45 Vol.-%), but there is also a small fraction of water vapor, oxygen, nitrogen, ammonia, hydrogen, and hydrogen sulfide. Commercial biogas plants utilize biobased wastes and agricultural residues to produce biogas as their main product and nutrient-rich digestate as a side product. (Al Seadi *et al.*, 2008) In Finland, biogas production was 876 GWh based on energy generation statistics from 2020 (Statistics Finland, 2020a). Based on the European commission's Waste framework directive (2018), the goal is to utilize the biogas process more for the biobased fraction of MSW. There is a techno-economic potential of 10 TWh for utilizing more agricultural residues in Finland (Virolainen-Hynnä, 2020). When biogas is upgraded to biomethane as a transportation fuel, there is an unutilized and concentrated stream of biogenic CO_2 that can be captured (Yang *et al.*, 2020). A rough estimate for the biogenic CO_2 stream in Finland is 42 000 tons/year, calculated from the current biomethane production and assumed composition of 35 vol-% CO_2 (Al Seadi *et al.*, 2008; Statistics Finland, 2020a). That is around 0.07% of the annual Finnish CO_2 eq emissions (Tilastokeskus, 2019).

The biogas process consists of feedstock pretreatment, anaerobic digestion, biogas treatment, and digestate treatment (Figure 6) (Feiz *et al.*, 2020). In pretreatment, the feedstock can be cleaned from impurities, for example, plastic and sand that are harmful to the rest of the process. Anaerobic digestion can be done in a continuously stirred tank reactor, plug flow reactor, or batch reactor. Each of these reactors requires several unit processes that prepare the feedstock for the reactor, upgrade the produced biogas, and treat the digestate. Anaerobic digestion requires heat for providing optimal conditions for the bacteria. Biogas composition and plant configuration are dependent on the feedstock because the anaerobic bacteria break the varying compounds differently, and the needed unit processes are different for slurry biomasses compared to dryer biomasses. (Al Seadi *et al.*, 2008)

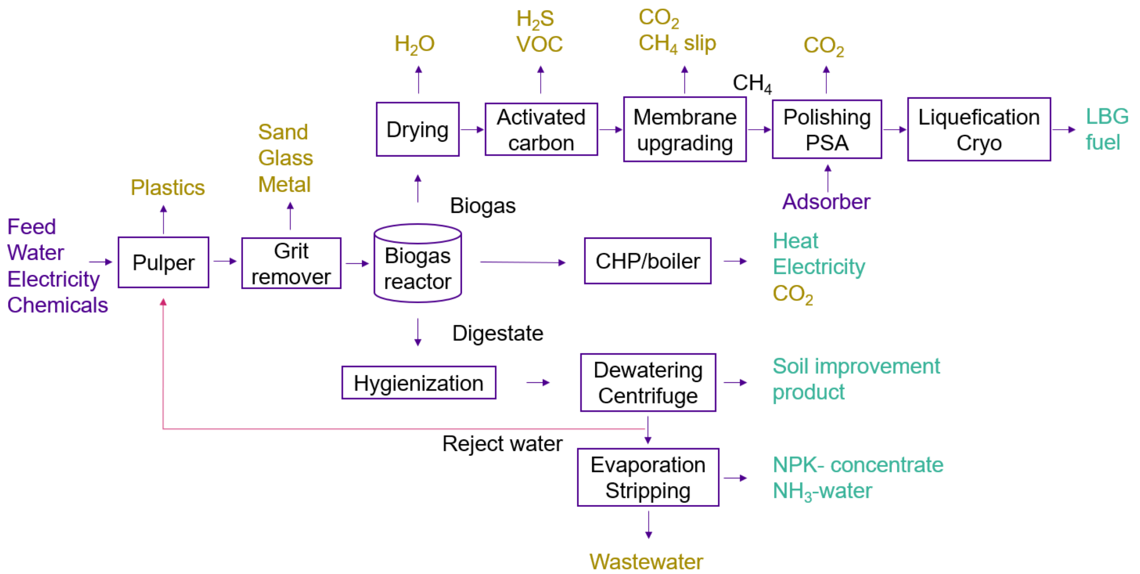

Figure 6. *Process chart of biogas plant that produces liquefied methane, heat, electricity, and digestate-based products. The whole biogas process is presented above but the current study focuses only on the membrane upgrading process because that was identified as the easiest point source to capture carbon dioxide.*

Biogas can be utilized through combined heat and power system to produce heat and electricity for the plant's own needs, and the excess electricity can be injected into the national grid (Ardolino *et al.*, 2021). By upgrading biogas to biomethane, organic material can be converted to transportation fuel in road transport to replace diesel (Yang *et al.*, 2020). Upgrading is generally done with membrane separation, water scrubbing, chemical adsorption, or pressure swing adsorption. Membrane separation uses hollow fibers made from polymeric materials, that have high selectivity for CO_2 and CH_4 separation. (Ardolino *et al.*, 2021) Biogas is passed through the membrane process with pressure and CO_2 goes through the membrane as permeate and CH_4 stays on the retentate side (Baena-Moreno *et al.*, 2020).

The CO_2 stream from membrane separation is high concentration (> 96%), so it can be captured straight from the membrane process with compression (Yang *et al.*, 2020). Purity requirements for captured CO_2 depend on the utilizing or storing technology and method. For storage, CO_2 purity requirements are not clear, but the gas mixture should not cause harm to transportation and storage infrastructure, environment, or health. For example, in Canyon Reef (USA) CO_2 pipeline the required CO_2 content is 95%. When the compressed CO_2 stream has impurities the required electricity consumption and storage volume increase but the optimal CO_2 purity level and utilization or storage application need to be defined on a case-by-case basis. (Markewitz *et al.*, 2012)

The biogas process can be seen as an energy production or waste management process depending on the study's goal. This perspective difference leads to different functional units for biogas LCAs. The functional unit can be the energy amount of electricity or heat produced, the volume of biogas or biomethane produced, or the mass of biomass processed. (Bacenetti *et al.*, 2016; Heiker *et al.*, 2021) Even though the LCA studies from biogas productions are made by following the LCA-related ISO 14040/44 standards, the studies are not comparable to each other because of the number of different assumptions and value-based choices (Feiz *et al.*, 2020). For example, in comparison of CO_2 utilization in greenhouses and geological storage, the result was affected by the fuel or material that CO_2 is substituted for (Oreggioni, Luberti and Tassou, 2019). Only studies that have the same assumptions are comparable to each other. In biogas LCAs, the uncertainty rises from heterogeneous feed and the complexity of the different unit processes in the biogas plant itself. (Feiz *et al.*, 2020) For example, the true composition and biogas yield can vary especially in smaller and newer plants. In general biogas process is multifunctional and there is no consensus on how to solve that issue with LCA methods (Bacenetti *et al.*, 2016). For example, a single best and fair allocation key could not be found for digestate use and biogas. The mass, energy, and value-based allocation keys gave different options, but the goal to see the digestate as a valuable coproduct was not possible, because the common allocation keys will divide the emissions eighter mostly to digestate or biogas. (Timonen *et al.*, 2019)

Starr *et al.* (2012) studied different biogas upgrading technologies and found out that energy demand and regent use are the most impactful factors when comparing upgrading technologies. They also conducted a more simplified LCA for comparing novel upgrading methods to more mature ones in the market. The SLCA focused on energy demand and reagent use and it was calculated for a functional unit of 1 t of CO_2 separated from the biogas that had assumed composition of 50 vol-% CH_4 and CO_2. The bottom ash upgrading process that was able to immediately store the removed CO_2 had the lowest impacts overall. (Starr *et al.*, 2012) Ardolino *et al.* (2021) found that membrane separation had the overall best performance, but their LCA and life cycle cost study had only the commercially available technologies for biogas upgrading (Ardolino *et al.*, 2021). Bacenetti *et al.* (2016) said that energy crops cultivation, operating the biogas plant, and the emissions from open digestate tanks are the most important GHG emission sources. They suggest that covering digestate tanks and building smaller biogas plants with sludge and manure feedstock would be more environmentally beneficial than operating large plants with energy crops because utilizing waste materials from other processes has zero production emissions. (Bacenetti *et al.*, 2016) Biogas is beneficial to upgrade to biomethane because the membrane upgrading will provide GWP reductions of 58-79% compared to straight utilization to energy with CHP unit (Ardolino, Parrillo and Arena, 2018). Avoiding the production emissions of fossil transportation fuels is the main contributor to the high GWP reduction in the study by Ardolino *et al.* (2018). Yang *et al.* (2020) also recommend upgrading biogas from cellulosic biomass to transport fuel and applying carbon capturing and storage to maximize the GWP reductions when the storage facilities are accessible (Yang *et al.*, 2020).

2.3.3 LCA studies on natural and recycled aggregates

Construction and demolition waste (C&DW) is the largest waste stream in the world, and that is why it is crucial to utilize the produced material as well as possible (Colangelo *et al.*, 2020). Mineral construction and demolition waste generation were 360 million tons in The EU in 2018 (Eurostat, 2022). In the EU the goal is to recycle 70% of the C&DW by 2020, but the recycling rate is closer to 50% (Colangelo *et al.*, 2020). In Finland, the current utilization rate is still under 60% (Hakaste, 2022). A large amount of construction and demolition waste can be easily reused as a replacement for natural stone or gravel aggregate in road base coarse aggregate, construction fill, and drainage systems. Fore mentioned utilization applications have been in use widely around the world and this subject has been studied for decades. (Yazdanbakhsh *et al.*, 2017) Moved masses in the construction industry are huge and cause significant transportation emissions. Producing cement is very energy-intensive and causes 5-8% of the world's emissions (Teja Kusuma *et al.*, 2022). Recycled aggregate can be used in earthworks with the CE mark and applying MARA regulation in Finland. There are also several studies about using recycled aggregate in concrete, but the broad conclusion is that a replacement rate above 25% changes the properties of concrete which limit the safe applications of recycled concrete (Zaidi and Mujahid, 2009; Qasrawi and Marie, 2013; Vijaya and Senthil Selvan, 2015).

Natural aggregate production starts with excavating the topsoil of the natural rock. Then the detonation holes are drilled and filled with explosives. After detonation, the largest particles are broken to fit in the mouth of the crusher with an excavator equipped whit a hydraulic breaker. In Figure 7 the exploded rock is moved to the crushing unit with an excavator, and the crushing is executed with 2 different machines. The first mobile crusher will produce natural aggregate with particle sizes ranging from 0 mm to 150 mm. The second unit will crush the coarse aggregate to a finer more controlled particle size range. All these machines run on diesel and the consumption is around

Figure 7. *Crushing equipment used in small-scale production of natural aggregate in southern Finland.*

In Finland, demolition waste has a waste status that will limit its use as recovered material. Waste needs to be treated according to the environmental law and it usually needs an environmental permit to use and treat the waste, which makes crushed concrete challenging material to use in the fast passed construction business. There is an easement of this law that will allow using demolished concrete as recycled material with only MARA notice. To apply the MARA regulation firstly, the demolition site needs to be inspected, for example, oil-contaminated concrete, and during the demolition different materials need to be sorted. When concrete and tiles are crushed there is a set of analyses that needs to be done before the product can be used. These analyses consist of solubility or leaching tests, concentration analysis of different toxins and heavy metals, material distribution, and impurities such as plastics. (FINLEX, 2017) After the analyses and the selective demolition is done the demolished concrete that is transported to the production plant has a particle size smaller than 300 mm. The crushing is executed with only the first unit that produces recycled aggregate with particle sizes ranging from 0 to 150 mm. Metal is separated mostly before crushing in the demolition site with a magnet and the rest of the metal is separated during the crushing. The finished product has a notable amount of fine matter and a small fraction of bricks as can be seen in Figure 8.

Figure 8. *MARA-approved recycled aggregate from demolished concrete with particle sizes 0-150 mm.*

Recycled aggregate production is seen generally as a waste management process and the goal of the C&DW LCAs is to compare natural aggregates to recycled aggregates. The functional unit is based on the mass of the waste or product or the volume of concrete (Colangelo *et al.*, 2020). Recycled aggregate can be also used in the replacement of natural aggregate in concrete and LCAs considering recycled aggregate concrete the functional unit is tied to the volume and strength of new concrete. (P Visintin, Xie and Bennett, 2020) Setting the system boundary to take important parts of the processing can be a challenge when considering recycled materials. Colangelo et al. (2020) decided to consider the selective demolition that is needed for recycled aggregate production even though some can say that demolition is part of the building's end-of-life treatment and that's why does not belong to the production of recycled aggregate's life cycle. Borghi *et al.* (2018) set the system boundary for recycled aggregate to start when the material enters the management system and end when the product leaves the system. Yazdanbakhsh *et al.* (2017) studied recycled aggregates in the context of a large city and concluded that the environmental impact of recycled aggregate is sensitive to transportation distances and the impact difference to natural aggregate is not significant in their case study, because the production emissions of the aggregates are similar to each other. The land use change was not considered in the study.

Wu *et al.* (2015) studied construction waste management with a streamlined LCA method where the inventory data was collected from Ecoinvent. They found that concrete waste has a material embody impact of 63.60 kg CO_2-eq/t waste. They concluded that reducing concrete waste is the most effective way to reduce the impact of construction waste, if reducing is not possible then recycling is the best option. The study did not present any applications of how concrete can be recycled. (Wu *et al.*, 2015) and Borghi *et al.* (2018) conducted an LCA study where natural and recycled aggregate production was compared in a local study in Italy. Their process data shows that diesel consumption of natural aggregate extraction is 0.39 l/t aggregate, and the aggregate production is 0.46 l/t aggregate. Depending on the plant fuel type (electric or diesel) the diesel consumption of the recycled aggregate plant is 0.25 l/t aggregate and 0.64 l/t aggregate respectively. The current C&DW recycling system produces 3.4 kg CO_2-eq/t and the best-case scenario creates a positive impact of -1.78 kg CO_2-eq/t, where the recycling plants are electric and the transportation distances are minimal. (Borghi, Pantini and Rigamonti, 2018) Ghanbari *et al.* (2017) compared natural aggregate production and recycled aggregate production in a centralized recycling plant. Natural aggregate production caused 4.45 kg CO_2-eq/t aggregate and recycled aggregate production caused 1.25 kg CO_2-eq/t aggregate (Ghanbari, Abbasi and Ravanshadnia, 2017).

Hardened concrete can over time absorb some of the CO_2 that is released in cement manufacturing (Zhang *et al.*, 2019). The CO_2 uptake happens in the exposed surface area of concrete and crushing the concrete will expose uncarbonated calcium hydroxide (Eq 2) (CO_2NCRETE SOLUTION, 2020).

$$Ca(OH)_2 + CO_2 \rightarrow CaCO_3 + H_2O \qquad (2)$$

The carbonation rate decreases over time, but when demolished concrete is crushed carbonation rate is faster again, but it is also dependent on how carbonated the concrete was before crushing. The carbonation rate is also affected by the cement content of concrete, the CO_2 amount in the atmosphere, the diffusion process between concrete and the atmosphere, and the environmental conditions of the concrete such as is the concrete buried, exposed, indoors, or sheltered and the compressive strength of the concrete. Crushed concrete that is buried will take up 40-55% of CO_2 emissions in the production of cement. (P. Visintin, Xie and Bennett, 2020) Zhang *et al.* (2019) state that ignoring carbonation can lead to a 13-48% overestimation of CO_2 emissions from recycled concrete aggregate. Estimating carbonation and especially the additional carbonation caused by crushing the concrete is challenging (CO_2NCRETE SOLUTION, 2020)

2.4 Carbon capturing, storage, and utilization

Carbon capture (CC) is identified as one solution to meeting the global warming targets set by IPCC. Carbon dioxide (CO_2) can be captured from large point sources such as fossil fuel or biomass energy facilities, cement production, the iron and steel industry, and the chemical industry. (Abanades *et al.*, 2005) Captured carbon can be stored for the long term taking the CO_2 out of the atmosphere or utilized in industry replacing other resources (Cuéllar-Franca and Azapagic, 2015).

The CC technologies can be classified as post-conversion or combustion, pre-conversion, or combustion, and oxyfuel combustion. Post-conversion is done from the waste streams of the studied process, such as flue gases. One of the methods is amine scrubbing from flue gases. (Cuéllar-Franca and Azapagic, 2015) It is used for removing CO_2 from natural gas for decades and it is also used in commercial-scale post-combustion carbon capture in coal-fired power plants. Membrane separation has been used on a commercial scale for CO_2 separation from syngas. Other membrane separation applications have been on a demonstration scale. (Bui *et al.*, 2018) Absorption by solid sorbents, cryogenic separation, and pressure swing adsorption are also post-conversion technologies. Pre-conversion carbon capture refers to capturing CO_2 as an undesired co-product of an intermediate reaction of a conversion process. The same amine solvents can be done in ammonia production where pre-conversion capture is applied. Oxy-fuel combustion is a combustion method where fuel is burned with pure oxygen to produce flue gas with high CO_2 concentration and free from nitrogen compounds. This method will eliminate the need for CO_2 separation from the flue gases, but it is expensive and energy-intensive due to pure oxygen production. (Cuéllar-Franca and Azapagic, 2015)

CO_2 can be stored in the ground, ocean, or as a mineral carbonate. Carbon capturing and storage (CCS) can be done in depleted oil and gas reservoirs. That is one of the most promising storage solutions due to the existing infrastructure and understanding of the storage area. (Cuéllar-Franca and Azapagic, 2015) One example of utilizing empty gas reservoirs is the project Porthos where the captured CO_2 from industry will be transported and stored in empty gas fields beneath the North Sea. The compression pressure for the gas is 130 bars for the pipeline transmission and the injection. Porthos has a capacity of storing 2.5 M t CO_2 per year for 15 years. The operation is expected to start by 2024. (*Project - Porthos*, 2021) Ocean storage is still in the early phases of development and there are concerns about the potential acidification and eutrophication impacts (Bui *et al.*, 2018; Bisinella *et al.*, 2021).

CC is already done in the waste incineration field, but it is far from standard practice. Based on (Wienchol, Szłęk and Ditaranto, 2020) there are four incineration plants with CC systems; Klemetsrud CHP in Norway, Saga City in Japan, Twence, and AVR plants in the Netherlands. These plants use the CO_2 in local greenhouses, algae farming concrete or chemical industry for plastic and biofuel production. The plant in Norway will store the CO_2 in the North Sea. (Wienchol, Szłęk and Ditaranto, 2020)

A review article by (Bui *et al.*, 2018) states that CO_2 transport technologies: pipelines, trucks, and ships are well established. In the US the pipeline infrastructure is well established because of enhanced oil recovery. (Bui *et al.*, 2018) In Finland, the problem with CCS scenarios is that there is no production chain to the storage facilities. The closest planned storage location is eighter in Norway or in Rotterdam where the project Porthos takes place (*Project - Porthos*, 2021).

CO_2 is possible to be utilized directly as growth enhancement in greenhouses in the summertime when plants' CO_2 uptake is highest due to the high light intensity. Carbon capturing and utilization (CCU) can be done in algae farming where biomass or valuable products are produced. CO_2 can be used as an additive in light concrete to increase pore size and enhance the hardening process. (Christensen and Bisinella, 2021) based on Leung et al 2014 stated that adding CO_2 to concrete would potentially decrease the needed cement amount. CO_2 can be also used in the food industry as a protective gas and in beverages. This application will need food-grade CO_2 that is purified after the capturing process. CO_2 can be used as a coolant in industrial applications and in some special cases in dry ice form. Compressed CO_2 can be utilized as a fire extinguisher. One very site-specific CO_2 utilization application is enhanced oil recovery, where CO_2 is pumped into the oil fields. Thermal ashes can be stabilized with CO_2 mineralization. (Leung, Caramanna and Maroto-Valer, 2014; Christensen and Bisinella, 2021)

The decarbonization potential of applying carbon capture to MSWI and biogas cases are studied in this paper. For the MSWI case, the chosen capturing method is MEA absorption because MEA has been applied to the MSWI plants on the technical scale before and it can be retrofitted to the MSWI plant (Christensen and Bisinella, 2021). Also (Wienchol, Szłęk and Ditaranto, 2020) stated that post-combustion carbon capture is the most mature technology and can be retrofitted with the least impact on existing systems. For the biogas case, membrane separation is already utilized in the studied process the choice of separation method was obvious. Membrane separation (MS) is already done in the natural gas industry and the capturing technology is very simple because the exhaust gas from MS contains > 96% CO_2. (Bui *et al.*, 2018; Yang *et al.*, 2020) With 3 stage MS process, the off-gas contains 98.9% CO_2 (Ardolino, Parrillo and Arena, 2018) Depending on the utilization scenario only compression is a sufficient way to capture CO_2 from the MS process (Yang *et al.*, 2020).

3. MATERIALS AND METHODS

The study was done in two parts. First, the decarbonization potential calculation method and tool were developed based on the SLCA approach. Secondly, the tool was applied to three case studies to assess its applicability to CE value chains. The first goal was to apply simplification methods to develop an unlicensed and easy-to-use LCA tool for initial decarbonization potential assessment of future improvement scenarios for CE businesses. The tool was applied to three CE value chains from the Topinoja business area: municipal solid waste incineration, biogas production, and concrete waste recycling. The goal was to determine how the aforementioned simplification methods can be applied to different circular economy value chains. The calculation was done based on actual process data and literature data. The research question of each value chain was: Is the identified decarbonization potential significant compared to the original value chain and Turku's emissions? The used assessment method is described in chapter 3.1 and the case studies are presented in chapter 3.2 with the used data and assumptions. Snapshots from the tool are presented in appendix 1. Calculations were made with Microsoft Excel and the used inventory and emission factors are presented in sections 3.1.4 and 3. Results are compared with existing EPDs and literature values of similar processes. The developed calculator is available on the CircHubs website (CircHubs, 2022).

3.1 Excel tool development

The excel tool is developed to calculate the decarbonization potential between the current case and the improved scenario. Reaching the carbon neutrality goals requires carbon emission assessment, but CE companies can lack the resources to make full environmental assessments. The need for an unlicensed and easy-to-use carbon calculator was identified in the previous study, where they did not encounter suitable tools for an unlicensed and easy-to-use calculator for CE value chains (Lunden, 2021). The tool is intended to be used in early development phases for assessing the decarbonization potential of different improvement scenarios.

The tool takes in the current process data and produces initial decarbonization potential for improvement scenario based on literature values. The tool consists of three specific cases for CE value chains chosen beforehand in the Climate positive business areas - project. Each value chain has one improvement scenario that was chosen with local experts. The tool calculates the decarbonization potential for future improvement scenario that could be implemented on the value chain. The SLCA method was applied to decrease the amount of needed inputs to the tool. Due to the nature of the LCA method discussed in Chapters 2.1 and 2.2, the development of the tool was iterative process. Discussions with industry experts were also used in the process to direct the calculation methodology to serve industrial purposes. The materials and made assumptions concerning the method and tool development are presented in the next chapters.

3.1.1 SLCA based approach

The basic idea of SLCA is to systematically simplify the LCA process and make sure that all important factors are considered. The importance of the factors is tested by comparing the results to the literature and making sensitivity tests (Wu *et al.*, 2015; Wang, Levis and Barlaz, 2021). For this study, the value chains were chosen earlier in the ILPO project. In the previous study, suitable assessment tool was researched, and conducting SLCA based assessment was recommended (Lunden, 2021). The current study focuses on developing a method for quantifying the decarbonization potential of earlier identified improvement scenarios to the value chains.

The core procedure was based on the ISO 14044 recommended steps. Definition of simplification methods was made based on Gradin and Björklund (2021), Beemsterboer *et al.* (2020), and the goal of this study. In the goal and scope phase, the literature review and the decisions about the system boundary, functional unit, assumptions, and data sources were made (Chapter 3.2). Because this study is focused on a specific business area the system boundary was narrowed to a gate-to-gate -approach instead of assessing the whole lifecycle of generated products. The assessment was simplified to consider the changing segment of the process to avoid excess work in conducting full LCA of the facility because the goal is to assess and identify the additional decarbonization potential within the CE business.

In full LCA multiple impact categories are described shortly in Chapter 2.1.3, but in this study the main focus was GWP. GWP assessment of value chains will provide relevant data to the local businesses and companies. Using the results of the GWP calculation, they can reduce their GHG emission and achieve emission targets. Choosing to asses only GWP can shift the environmental impact to other impact categories, but in this case, the goal is not to make a full environmental assessment (Pelton and Smith, 2015). The decarbonization potential of each case study was calculated with Equation 3.

$$Decrabonization\ potential = GWP_{current} - GWP_{improved} \quad (3)$$

With cut-off criteria, the production emissions from needed infrastructure changes were left out of the study because the overall effect was assumed to be negligible compared to operational GHG emissions (Tang, You and Frederick, 2018). All needed process data were not available, so databases such as ecoinvent 3.6 and other studies were utilized to fill the data caps.

The most important and effective simplification method was omitting identical parts in a comparative study (Gradin and Björklund, 2021). In MSWI and biogas case, the studied improvement was only part of a larger process, so we could keep the original process and only compare the effect of adding a carbon capture unit to the process. With that simplification method, all the original process emissions that are not changed by the addition of the improvement unit can be left out of the calculations. For example, in the biogas case, the digestate management is not changed by the addition of the CC unit to the upgrading process, so we can omit the digestate management emissions from the calculations. This simplification method will give information about the possible improvement scenarios to the original process, and it is not suited for studies that aim to quantify the whole GWP of the process.

3.1.2 Identification of most impactful flows

The first task in the SLCA approach is to identify the most impactful flows and focus more on such flows during data collection. Other flows are important to consider but it can be done with literature values and general emission factors without affecting the accuracy. More crucial is to get all the flows in the assessment. (Laurin and Dhaliwal, 2017)

In this study, the most impactful flows were identified based on literature and in collaboration with local experts, studying value chains, and process charts (Christensen and Bisinella, 2021; Wang, Levis and Barlaz, 2021). The studied processes were simplified, and they resulted in less than 10 flows (Chapter 3.2). Mass-based evaluation of emissions was done to choose the most significant flows. In the MSWI case, the most impactful flow was required heat, in the concrete case it was fuel consumption, and in the biogas case, the carbon-capturing rate, and electricity demand were the most impactful flows.

When other emissions are affecting GWP, in such cases only mass-based assessment is not the correct approach. It is critical to consider the higher emission factors for methane (CH_4) and dinitrogen monoxide (N_2O). Even small amounts of N_2O can cause significant emissions even though the mass-based contribution is under the cut-off- value (Table 1).

3.1.3 Hierarchical structure for inventory collection

After identifying the most impactful flows, the required emission factors were collected, and missing data gaps are filled with literature data (Henriksen *et al.*, 2019). These flows need to be as accurate as possible to ensure that the results are reliable. Location and the level of industrial development in the region are the defining parameters that cause the difference in true emissions (Sevigné Itoiz *et al.*, 2013). In different parts of the world, energy is produced in different ways that do not emit equally and the energy emissions change significantly the total GWP results (Ardolino, Parrillo and Arena, 2018). Also, distances will affect the emissions of producing goods via needed extra transportation and infrastructure. For example, if the production of a specific product is located in a remote area the transportation emissions are generally counted into the emission factor that can be found in ecoinvent. That is why emission factors and data caps should be filled with local data wherever possible and from similar processes or studies with the same assumptions (Sevigné Itoiz *et al.*, 2013).

The used data in the most impactful flows must be suited for the Finnish industry level of practice and processes. In large industries, the emissions for the Finnish average can be found in the National inventory report (Statistics Finland, 2021). For energy values, the Finnish energy authority is the main source of electricity and gas emissions (Energy Authority, 2021). If there weren't any process data, the first choice was the Finnish average and the second choice was similar studies from suitable areas or EPDs, preferably from Nordic countries or from Europe. The less impactful flows were taken from Ecoinvent 3.6. Ecoinvent is an extensive databank that is easy to navigate so using it will reduce the time needed for data collection. The drawback to ecoinvent is that the data can be old and not necessarily represent current technology in all cases. If Ecoinvent does not have the needed emissions, then the last resort was IPCC defaults which are a rough and conservative average of global emissions (IPCC, 2021).

By using the knowledge of the most impactful flows, the efforts for data collection can be focused on the most impactful flows and by the standard of the iterative approach improve the data quality of those flows. With that same principle, the data for less impactful flows can be more easily accessible than average data from databases or default inventories.

3.1.4 Emission factors

Emission factors are collected from different sources, but to keep the calculations as accurate as possible, it is important to use local emission factors that represent local practices. There are also global emission factors that are average of the whole industry's environmental stage that can vary significantly between developed and developing countries. IPCC is the main actor that collects the emission factors under the Kyoto protocol.

For this study, the emission factors are collected from different sources and the aim is to be as local as possible for Finnish cases. The general emission factors are used in Table 1, the MSWI case specific EFs are in Table 2, and the aggregate case-specific emissions are in Table 3.

Table 1. *General emission factors (EF) and their sources.*

General emission factors			
Flow	EF	Unit	Source
Biogenic CO_2	0	kg CO_2-eq/kg CO_2	[1]
Fossil CO_2	1	kg CO_2-eq/kg CO_2	[1]
Captured fossil CO_2	0	kg CO_2-eq/kg CO_2	[1]
Captured biogenic CO_2	-1	kg CO_2-eq/kg CO_2	[1]
CH_4	25	kg CO_2-eq/kg CH_4	[2]
N_2O	289	kg CO_2-eq/kg N_2O	[2]
Electricity	0.106	kg CO_2-eq/kWh	[3]

[1] (Christensen *et al.*, 2009) [2] IPCC (100 yrs.) [3] Energy Authority (2019)

Table 2. *MSWI case-specific emission factors and their sources.*

MSWI case-specific emissions			
Flow	EF	Unit	Source
District heating	148	kg CO_2-eq/MWh	[1]
MEA production (global)	3	kg CO_2-eq/kg MEA	[2]
Activated carbon	18.24	kg CO_2-eq/kg AC	[3]
NaOH (27%) dry	1.4	kg CO_2-eq/kg NaOH	[2]
Helsinki region tap water	35	g CO_2-eq/m^3	[4]

[1] Motiva (2020), [2] Ecoinvent 3.6, [3] (Gu *et al.*, 2018), [4] HSY news article

Table 3: *Aggregate case-specific emission factors and their sources.*

Aggregate case-specific emission factors			
Flow	EF	Unit	Source
Dynamite	1.13	kg CO_2-eq/kg dynamite produced	[1]
Emulsion explosive (Kemiitti 510)	1.17	kg CO_2-eq/kg Kemiitti produced	[2]
Natural aggregate general	3.6	kg CO_2-eq/ton aggregate	[3]
Natural aggregate 0-16, 0-90	3.9	kg CO_2-eq/ton aggregate	[4]
NCC mobile crusher 0-150, 0-90	2.2	kg CO_2-eq/ton aggregate	[5]
Diesel	2.66	kg CO_2-eq/l fuel	[6]

[1] EDP (Salminen, 2020), [2] EDP (Komulainen, 2020), [3] EPD (Vilniaus karjerai, 2021), [4](NCC Industry Nordic AB, 2020), [5] (Vainio-Kaila, 2020), [6] LIPASTO (2016)

3.1.5 Interpretation and Sensitivity analysis method

Sensitivity analysis was done by changing the values of different parameters and assessing the change in results (Curran, 2015). The analysis was done by changing one parameter at a time in the calculator developed in the present study (CircHubs, 2022). The parameters were chosen based on the available data and identified uncertainties (Table 4).

In the municipal solid waste incineration case, the parameters that were changed were the share of fossil and biogenic carbon in MSW content. The MSWI case was tested with 100 % fossil carbon and with 100 % biogenic carbon. The goal is in the future that all biomass is directed to biogas production instead of the incineration plant. This change will reduce the biogenic carbon content to very low which then lowers the carbon sink potential of CCS or CCU. Also, capturing efficiency can vary between 85-95% but there was no process data available from more efficient MEA separation, so the change in required resources could not be assessed.

In the biogas case, four possible parameters could be changed: electricity demand, the emission factor of electricity, the biogas content, and the carbon-capturing efficiency. The effect of biogas content was studied based on process data. The difference between electricity sources was studied with the general emission factor of the Finnish electricity mix and using zero-emission renewable electricity as an alternative. The electricity demand for compressing the CO_2 and capturing efficiency was not studied due to a lack of data.

In the aggregate case, three possible parameters could be changed: fuel consumption, transportation distance, and explosive consumption. The ability to change transportation distance (15 km) was built into the calculator, so the sensitivity of the demolition site distance can be assessed case-by-case basis. Similarly, fuel and explosive consumption can be assessed on a case-by-case basis. Because the fuel consumption was identified as the most impactful flow, the sensitivity of the fuel choice was assessed.

Table 4. *Sensitivity analysis parameters*

Parameter	Min	Max	Source
MSWI			
Biogenic carbon share	0 %	100 %	
Biogas			
Electricity EF	0.106	0.262	[1],[2]
CO_2 content in biogas	32 %	42 %	[3]
Aggregate			
Fuel EF	0.66 kg CO_2-eq/l fuel	0.266 kg CO_2-eq/l fuel	[4],[5]

[1] Energy Authority (2019), [2] Ecoinvent 3.6, [3] Process data, [4] (Neste, 2022), [5] LIPASTO (2016)

3.2 Case study descriptions

In this chapter, the used inventories for the MSWI, biogas, and aggregate value chains are presented. Case-specific assumptions and sources are also discussed. Assessed value chains were chosen earlier in ILPO- project. The goal of the ILPO-project was to help businesses to find new low-emission and carbon sequestration solutions, produce a carbon roadmap for the business area and create an internationally interesting reference about the carbon-positive business area (6Aika Turussa, 2020). Chosen value chains are from the studied business area or they are connected to the area addressed in ILPO-project. The decarbonization potential of each three value chain is calculated from the GWP difference between current case A and improvement case B. The improved case Bs were chosen with local experts and based on the literature review presented in Chapter 2.3.

3.2.1 Municipal solid waste incineration case study

In the MSWI plant case study the current average emissions of incinerating 1 t of municipal solid waste were compared to the emissions of the MSWI plant with a flue gas carbon capturing unit (Figure 9). In case A the input to the system is MSW and it produces electricity, district heating, emissions to air, and solid residue that is mainly ash. In the improved case B, the MSWI process itself stays the same and it can be omitted from the decarbonization calculations (Gradin and Björklund, 2021). The additional carbon capturing unit requires chemicals and energy to function, so the added resources are counted into the assessment. The captured CO_2 is seen as a product of the system and the final result is that the MSWI plant emits less CO_2.

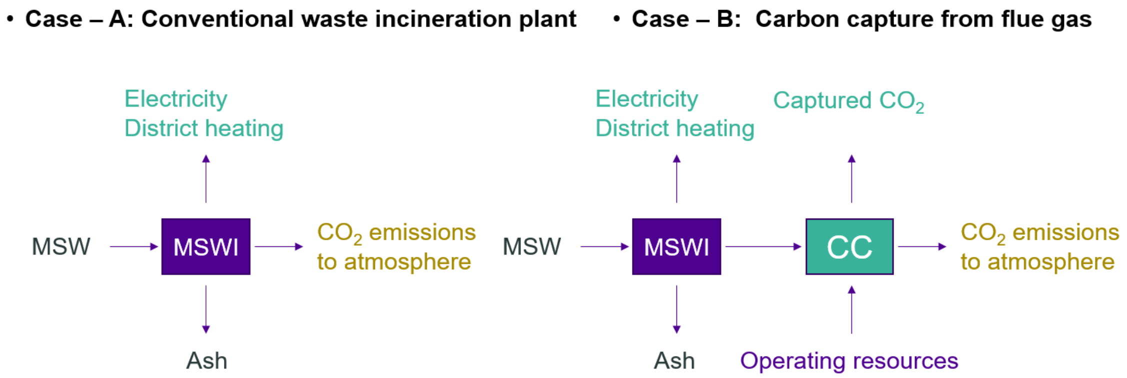

Figure 9. *MSWI case A with current practice and the improved case B with added flue gas carbon capturing unit.*

The MSWI case study was made based on inventory from Christensen and Bisinella (2021) and emission factors that were applicable to Finland. MEA carbon capturing unit was chosen for the study because it is possible to retrofit to an incineration plant and the technology is mature enough to have process data available (Bui *et al.*, 2018). MEA process has a large energy requirement compared to more advanced Oxyfuel combustion, but researchers state that the system has challenges with low quality fuels like MSW and retrofitting to an existing power plant (Wienchol, Szlęk and Ditaranto, 2020). The system boundary included only the actions in the MSWI plant, and all emissions caused by infrastructure were excluded because Tang *et al.* (2018) showed that the MEA-process infrastructure needed only 0.85 % more materials compared to the MSWI plant without the MEA process.

MEA process data was from Christensen and Bisinella (2021) that had been done on retrofitted carbon-capturing units for actual MSWI plants (Table 5). These plants are comparable to Finnish MSWI plants because they have typical grate furnace technology, advanced flue gas cleaning, and energy recovery. The original functional unit from Christensen and Bisinella (2021) was 1 ton of captured and compressed carbon. In this study, it was chosen to assess the process through the functionality of the MSWI plant, so the second functional unit was chosen to be 1 ton of Finnish MSW incinerated. The biogenic share of emitted CO_2 was assumed to be 50% based on the national inventory report (Statistics Finland, 2021). Solid reclaimer waste of MEA regeneration was ignored (Christensen and Bisinella, 2021).

Table 5. *MSWI case inventory*

Parameter	Value for capturing 1 t of CO2	Source	Value for incinerating 1 t of MSW	Source
Input				
Fossil CO_2	588 kg	mod. [1]	400 kg	mod. [1]
Biogenic CO_2	588 kg	mod. [1]	400 kg	mod. [1]
H_2O	840 kg	[2]	571 kg	mod. [2]
Activated carbon	0.07 kg	[2]	0.05 kg	mod. [2]
MEA	1.5 kg	[2]	1.02 kg	mod. [2]
NaOH	0.12 kg	[2]	0.08 kg	mod. [2]
Electricity	100 kWh	[2]	68 kWh	mod. [2]
Heat	3.7 GJ	[2]	2.5 GJ	mod. [2]
Output				
CO_2 pressurized	1000 kg	[2]	680 kg	mod. [2]
CO_2 to air	175 kg	[2]	120 kg	mod. [2]
NH_3	0.15 kg	[2]	0.10 kg	mod. [2]

[1] (Statistics Finland, 2021) [2] (Christensen and Bisinella, 2021)

3.2.2 Biogas case study

The decarbonization potential of adding carbon capturing unit to the biogas membrane upgrading process was assessed (Figure 10). Preventing biogenic CO_2 from entering the atmosphere would create a carbon sink and the goal was to identify the possible carbon sequestration solutions (6Aika Turussa, 2020). The improvement scenario of biogas was chosen to be CC from the membrane process because it was identified to be easy to apply and the application based on Yang *et al*. (2020) required only flue gas copressioning from the membrane process. Bacenetti *at al.* (2016) suggested using only waste materials as feed and covering the digestate tanks, but these actions were already applied to the studied biogas plant. The system boundary was chosen to include only the membrane upgrading part of the whole biogas process because that was the unit process affected by the CC unit and everything else was staying the same. The system boundary included only the flue gas from the membrane upgrading process because the unit process was identified to be the most accessible unit process for carbon capture. The comparison between case A and case B included the CO_2 emissions from the membrane process, the needed electricity to operate the compressor, and the amount of CO_2 removed from the process (Figure 10).

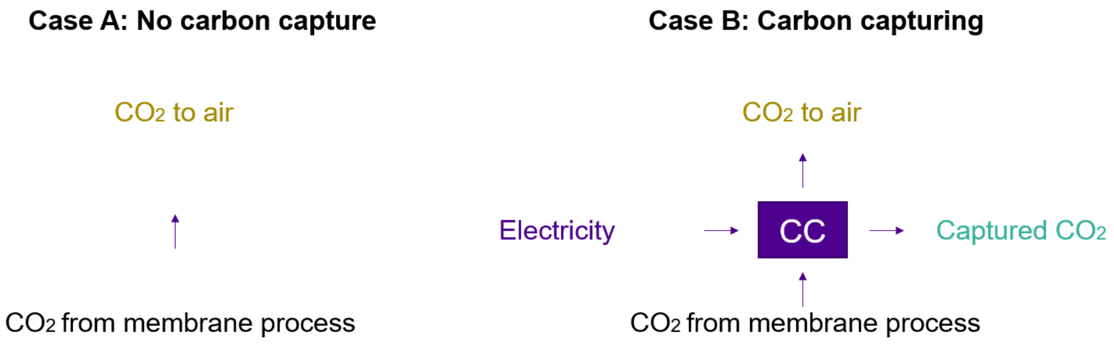

Figure 10. *Biogas case A with current practice and the improved case B with carbon capturing from the membrane process.*

Inventory was calculated based on process data from one Finnish biogas plant and literature values of carbon capture (Tables 6 and 7). The average biogas composition was 36 % CO_2 and 64 % CH_4. The biogas CO_2 fraction variates between 32-42 % based on process data. Biomethane production capacities were also either actual process data or designing the capacity of the plant in question. Capturing efficiency, and electricity consumption of CO_2 compressing was based on Yang *et al.* (2020) and Christensen and Bisinella (2021). Infrastructure was left out of the assessment based on Tang *et al.* (2018).

Table 6. *Biogas case inventory for functional units of 1 MWh of upgraded biomethane and*

Parameter	Value	Source
Input		
Electricity	8.2 (6.9-11) kWh	[1]
Output		
CO_2 to air	10 (8.7-13) kg	[2]
CO_2 pressurized	94 (78-120) kg	[2]
Biomethane	1 MWh	[3]

[1] (Christensen and Bisinella, 2021) [2] (Yang *et al.*, 2020) [3] Process data

***Table 7.** Biogas case inventory for the designed producing capacity of upgraded bio-methane (61 GWh) and functional unit of 1 ton CO_2 captured.*

Parameter	Value	Source
The functional unit is the designed producing capacity of upgraded biomethane		
Input		
Electricity	501 (419-645) MWh	[1]
Output		
CO_2 to air	634 (531-817) t	[2]
CO_2 pressurized	5710 (4777-7350) t	[2]
Biomethane	61 GWh	[3]
The functional unit is 1 MWh of upgraded biomethane		
Input		
Electricity	87 (73-113) kWh	[1]
Output		
CO_2 to air	111 (92-143) kg	[2]
CO_2 pressurized	1000 (836-1287) kg	[2]
Biomethane	11 MWh	[3]

[1] (Christensen and Bisinella, 2021) [2] (Yang *et al.*, 2020) [3] Process data

Based on process data the upgraded share was 50% of the whole biogas production. For obtaining the total decarbonization potential of the studied biogas plant, a full designed production capacity was considered in the decarbonization assessment. The biogas was cleaned from water and hydrogen sulfide before the membrane separation to protect the membrane. Due to the cleaning, it can be assumed that only CO_2 and CH_4 are directed to the first upgrading process. In membrane separation, the methane slip of 0.7% was assumed to be unchanged, so it was excluded from the assessment. The purity of captured CO_2 was assumed to be 99% and the compression pressure was assumed to be 150 bars (Yang *et al.*, 2020; Bisinella *et al.*, 2021).

3.2.3 Natural and recycled aggregate case study

In the last case study, the decarbonization potential of using recycled aggregate (RA) from demolished concrete instead of natural aggregate (NA) was assessed (Figure 11). NA and RA can be used interchangeably in earthworks, for example in road construction (Borghi, Pantini and Rigamonti, 2018). Production of the aggregates was included in the assessment. Production emissions were from the fuel and explosives used in different steps in the process (Chapter 2.3.3). The land use change in this specific case would have been challening to assess without double counting because the area is changed from forest to commercial area regardless of the quarrying.

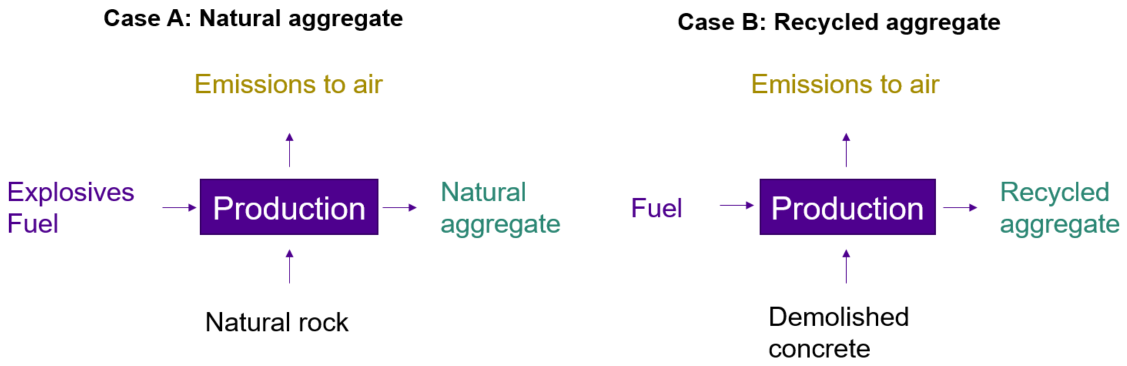

Figure 11. *Natural aggregate case A and improved case B with recycled aggregate.*

The system boundary was determined to be gate-to-gate and it included the exposure of the rock, the detonation of the rock, and all the machine work needed to achieve aggregate with desired particle size. All these actions were executed on site and the crushing was done with an outsourced subcontractor. The resource requirement data and produced aggregate amount were from past invoices from the year 2020 (Table 8). The functional unit was chosen to be 1 t of produced aggregate. All data for this case is acquired from one quarry that produces approximately 163 000 t of natural aggregates in various particle sizes.

Table 8. *Natural aggregate inventory*

Parameter	Value 1 ton aggregate produced	Source
Input		
Dynamite	11 g	[1]
Kemiitti 510	334 g	[1]
Fuel	0.83 l	[1]
Output		
Natural aggregate	1 ton	[1]

[1] Process data

The recycled aggregate inventory was collected from the same company as the natural aggregate inventory. The inventory data was collected from invoices from the year 2020 (Table 9). The system boundary includes the transportation of the demolished concrete to the production plant and the needed machine work to produce recycled aggregate.

Transportation is included in the assessment because it is an extra step that is needed for the production of RA.

Table 9. *Recycled aggregate inventory.*

Parameter	Value 1-ton aggregate produced	Source
Input		
Demolished concrete	1.003 ton	[1]
Fuel	0.46 l	[1]
Transportation	0.716 tkm	[1]
Output		
Recycled aggregate	1 ton	[1]
Steel	3.2 kg	[1]

[1] Process data

The production capacity of the larger recycled aggregate producer was 15 500 t/a. The transportation distance and mass were calculated from the main demolition sites of the year 2021 and scale data. The fuel consumption data is from the subcontractors so there are no guarantees on the accuracy of the data because it is challenging to separate tasks and the used fuel amount to different aggregates that are produced in the same plant.

The theoretical CO_2 binding capacity of cement is 485 kg CO_2/t cement. The normal CO_2 binding capacity of cement can be estimated to be 349 kg CO_2/t cement when the available CaO content for carbonation is around 72% of the total CaO amount. (CO_2NCRETE SOLUTION, 2020) 1 ton of concrete contains about 8-16 m-% of cement so in optimal conditions 1 ton of concrete can absorb 28-56 kg CO_2 back to $CaCO_3$ within its whole lifetime. The carbonation process is very dependent on the environment and time. It is challenging to accurately estimate how much exposing uncarbonated concrete by crushing increases the natural carbonation process, so the carbonation was not assessed in the current study.

4. RESULTS AND DISCUSSION

The goal of the present study was to utilize the SLCA based method developed and compare the decarbonization potential of different value chains in one business area in Turku, Finland. The GWP of the MSWI, biogas, and aggregate case studies and their improvement scenarios were calculated based on literature values and process data. The decarbonization potential was calculated with Equation 3 from the GWP results. The simplification methods were implemented in all case studies and the significance of improvement scenarios was compared and discussed. The results of these assessments and the functionality of the used method are presented and discussed in the following chapters.

4.1 Decarbonization potential of carbon capturing in MSWI

The decarbonization potential was assessed based on MSWI average emissions, MEA carbon-capturing, and emission factors presented in sections 3.1.4 and 3.2.1 (Tables 1, 2, and 5). The assessment showed 564 kg CO_2-eq/t MSW decarbonization potential between MSWI plant flue gas emissions without CC unit and with CC unit (Figure 12). In the study, the current scenario of the average Finnish MSWI plant is compared to the improved scenario of the MSWI plant with a retrofitted MEA carbon-capturing unit (Christensen and Bisinella, 2021; Statistics Finland, 2021).

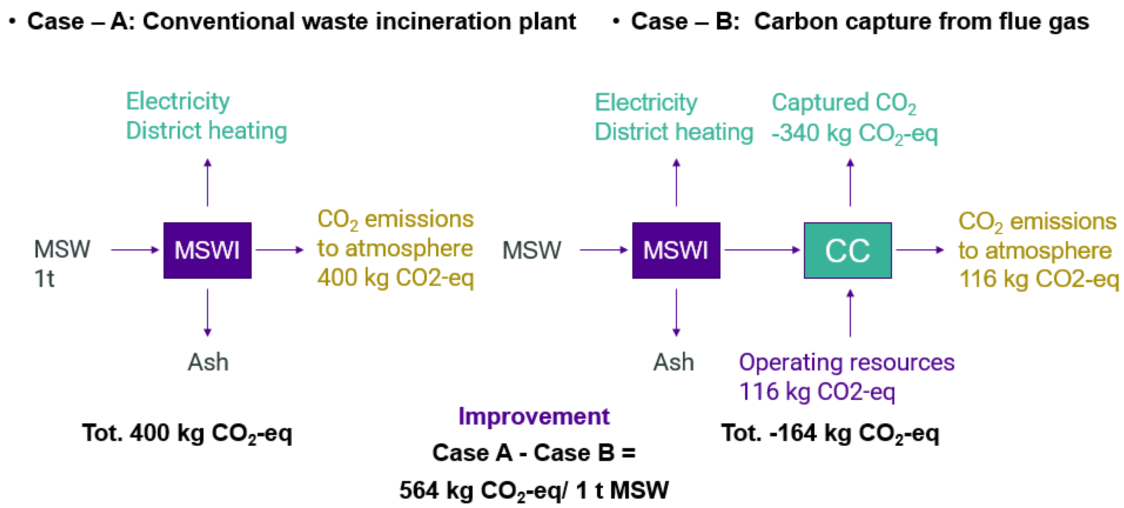

Figure 12. *MSWI case overall results.*

The current scenario emissions are 400 kg CO_2-eq/t of waste incinerated based on the Finnish literature average (Statistics Finland, 2021). In the Finnish emission inventory report the biogenic carbon share was assumed to be 50% (Statistics Finland, 2021). GWP after integrating the MEA process in MSWI is -163 kg CO_2-eq/t of waste incinerated (Table 10). Assessing the operational emissions of the MEA process resulted in 116 kg CO_2-eq/t of waste incinerated. 15% of the fossil carbon is still emitted into the air having a GWP of 60 kg CO_2-eq emissions. Degradation of MEA produces ammonia (NH_3) emissions to air that are estimated with IPCC default emission factors that are not regional. When the CO_2 is removed from the system it is calculated as 0 emissions or negative emissions depending on if the CO_2 has fossil or biogenic origin (Table 1). If the MEA case is applied to the current Finnish MSWI plant, the decarbonization potential would be 56 kg CO_2-eq/t of waste incinerated (Eq 3). When looking at the annual decarbonization potential of one plant with a waste incineration capacity of 120 000 t/a, the MEA process would create a 19.6 kt CO_2-eq carbon sink. The decarbonization potential represents 3% of Turku's annual emissions (Liljeström and Monni, 2020).

Table 10. *Emissions and emission reductions from MEA CC unit with 85% capturing efficiency. The functional unit was chosen to be 1 t of incinerated waste based on the main function of the whole process. CO_2 management after capturing and compressing was excluded from the assessment.*

Input	**Inventory**	**GWP (1 t waste)**
CO_2	800 kg	400
H_2O	571 kg	0.02
Activated carbon	0.048 kg	0.87
MEA	1.02 kg	3.06
NaOH	0.0816 kg	0.11
Electricity	68 kWh	8.91
Heat/ steam	2.52 GJ	103
	Total	**116 kg CO_2-eq**
Output		
CO_2 pressurized	680 kg	-340
CO_2 to air	120 kg	60
NH_3	0.10 kg	0.46
	Total	**-163 kg CO_2-eq**

The most impactful flows can be seen in the inventory and GWP results (Table 10). Heat is contributing 89% of the operating emissions. When applying the MEA process to practice it is crucial to investigate the available source of heat that is needed for the process. If the heat must be generated by fossil fuels such as peat or oil, the GWP from heat generation would be 260 kg CO2-eq/t waste or 184 kg CO2-eq/t waste based on emission factors from the national inventory report (Statistics Finland, 2021). The change in heat source can double the operation emissions.

GWP was calculated with the functional unit of 1 t of CO_2 captured so that different cases in the study could be compared with each other (Table 11). The total GWP of carbon capturing is -240 kg CO_2-eq/t CO_2 captured. The operation emissions are 171 kg CO_2-eq/t CO_2 captured. Capturing 1 t of CO_2 will cause the emissions of operating, but the overall process results in a GWP of -240 kg CO_2-eq/t CO_2 captured. Without carbon capturing the GWP would be 590 kg CO_2-eq. The decarbonization potential is then 830 kg CO_2-eq/t CO_2 captured (Eq 3).

Table 11. *Emissions and emission reductions from MEA CC unit with 85% capturing efficiency. The functional unit was chosen to be 1 t of CO_2 captured too. CO_2 management after capturing and compressing was excluded from the assessment.*

Input	**Inventory**	**GWP (1 t CO_2 captured)**
CO_2	1175 kg	588
H_2O	840 kg	0.0294
Activated carbon	0.07 kg	1.28
MEA	1.5 kg	4.5
NaOH	0.12 kg	0.17
Electricity	100 kWh	13.1
Heat/ steam	3.7 GJ	152
	Total	**171 kg CO_2-eq**
Output		
CO_2 pressurized	1000 kg	-500
CO_2 to air	175 kg	87.5
NH_3	0.15 kg	0.68
	Total	**-240 kg CO_2-eq**

The study by Christensen and Bisinella (2021) resulted in 323 kg CO_2-eq/t MSW for the conventional MSWI plant without crediting the system from energy recovery. The result is 77 kg CO_2-eq/t MSW lower than the present study, which can be explained by using average data for case A of the present study and the difference in biogenic carbon share of the incinerated waste. Because Christensen and Bisinella (2021) used the consequential environmental assessment method, the case A MSWI plant GWP emissions are smaller than in the present study. The electricity and heat recovery are credited to the MSWI plants, but crediting does not cover the overall emissions of the plant. After Christensen and Bisinella (2021) credited the energy production, the GWP was 221 kg CO_2-eq/t MSW and 65 kg CO_2 -eq/t MSW for the two studied plants.

Wang and Geng (2015) calculated MSWI GWP to be 110 kg CO_2-eq/t MSW higher than the case A GWP result of the current study. The GWP difference can be explained with less advanced technology available at the time of the study and with more conservative assumptions. They used IPCC-recommended methods to formulate equations that could be used to estimate CO_2 emissions based on annual waste production, the average carbon content of the waste, and the oxidation factor in China between the years 2003-2012 (Wang and Geng, 2015).

Case B improved scenario GWP results and data were compared to previous studies. The introduction of the MEA process reduced the amount of recovered energy produced in the process (Christensen and Bisinella, 2021). Carbon capturing reduced the emission from 323 to 65 kg CO_2-eq/t MSW in the study by Christensen *et al.* (2021). The reduction is 100 kg CO_2-eq less than in the present study (-360 kg CO_2-eq), but the operational emissions were calculated differently by Christensen *et al.* (2021). Only the energy recovery penalty is calculated when in the current study emissions produced by operational resources are assessed as one.

Pour *et al*. (2017) stated that applying MEA-based carbon capture and storage system to the MSWI plants would remove 700 kg CO_2-eq/t MSW incinerated from the atmosphere in the global and Australian context. The study was made with SimaPro and the inventory was from databases. The GWP reduction was significantly higher compared to the present study's results. The overall GWP reduction was higher than the actual amount of CO_2 that was captured in the study by Christensen *et al.* (2021).

The chemical requirement increases when the MEA unit is applied to the MSWI plant. The production emissions are considered in the assessment, but transportation is included in the conservative emission factor. The solid waste caused by operating the MEA unit is not considered. Even though emissions from the infrastructure are determined to be less than 0.85% of the whole operation emissions, the construction, and needed materials will cause emissions. The assumed lifetime of 30 years will spread the emissions for the whole amount of waste that is incinerated in the time, so the impact for 1 t of waste is insignificant with the cut-off rules.

The goal in the future is to source separate all biobased waste into biogas production. This will reduce the amount of biogenic carbon that can be captured and simultaneously reduce the carbon sink potential of CCS from the current -240 kg CO_2-eq/t of CO_2 captured to 350 kg CO_2-eq/t of CO_2 captured (Figure 13). These changes take time, but if the waste sector will be included in the emission trading system, then the monetary value of CO_2 can motivate to use of more advanced solutions to reduce carbon emissions.

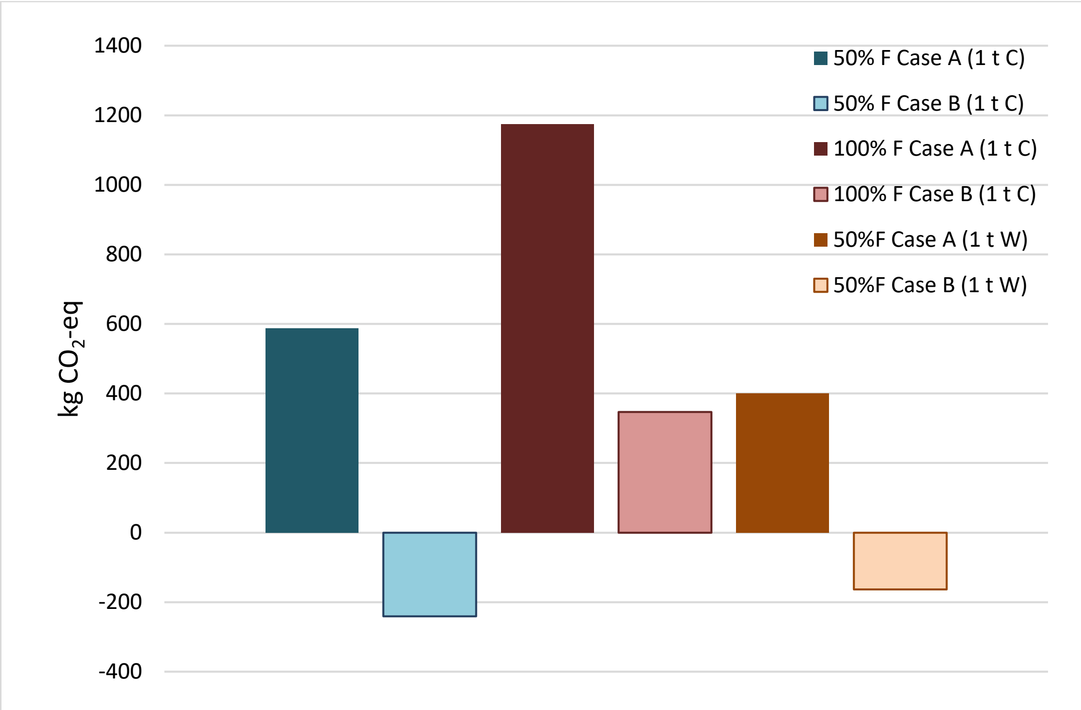

Figure 13. *Effect of the functional unit and origin of carbon to potential GWP reduction that can be achieved with applying carbon-capturing unit to MSWI plant. Case A is without a CC unit and Case B includes a flue gas CC unit. F symbolizes the fossil carbon share in the assessment. 1t C: functional unit is 1 t of CO_2 captured. 1 t W: functional unit is 1 t of MSW incinerated.*

Fossil and biogenic carbon share in the waste effects to the carbon sink potential that is possible to achieve with CC. The reduction of CO_2 from the system is always the same, but the emission factor for biogenic carbon is 0, so it is not shown in the GWP result. Also capturing biogenic carbon will bring the total GWP significantly lower if there is more biogenic CO_2 captured. From that perspective, it is possible to say that incinerating biogenic waste in plants with CC is a more beneficial climate change mitigation action compared to incinerating fossil waste, even though some other material management methods would result in more valuable products.

The problem with CO_2 storing or utilization infrastructure in Finland is that transportation can minimize the benefits of capturing CO_2. In the calculations, it is assumed that the captured CO_2 is permanently removed from the environment and the emissions related to the storage or utilization are not considered. Only the local decarbonization potential is calculated which can lead to optimistic conclusions. The present study shows the initial quantification method to help identify possible carbon sink scenarios.

The present study focuses only on GWP but the other environmental impacts of applying the CC unit are not considered. For future studies, it is important to assess human health and other environmental impacts such as eutrophication and acidification potentials. Using only GWP to determine the final decision of investments can cause the burden to be shifted to another impact category.

4.2 Decarbonization potential of carbon capturing in biogas upgrading

The biogas case study aimed to calculate the decarbonization potential of carbon capture from membrane separation. Only the membrane separation process was considered in the present study. The current membrane process was compared to the improved scenario where the CO_2 gas was captured from the membrane separation process by compressing it (Figure 14). The decarbonization potential of CC in biogas membrane upgrading process was 990 kg CO_2-eq/t CO_2 captured. The GWP assessment was done with two functional units, 1 MWh upgraded biomethane, and 1 t CO_2 captured. Also, the plant-specific annual decarbonization potential was studied.

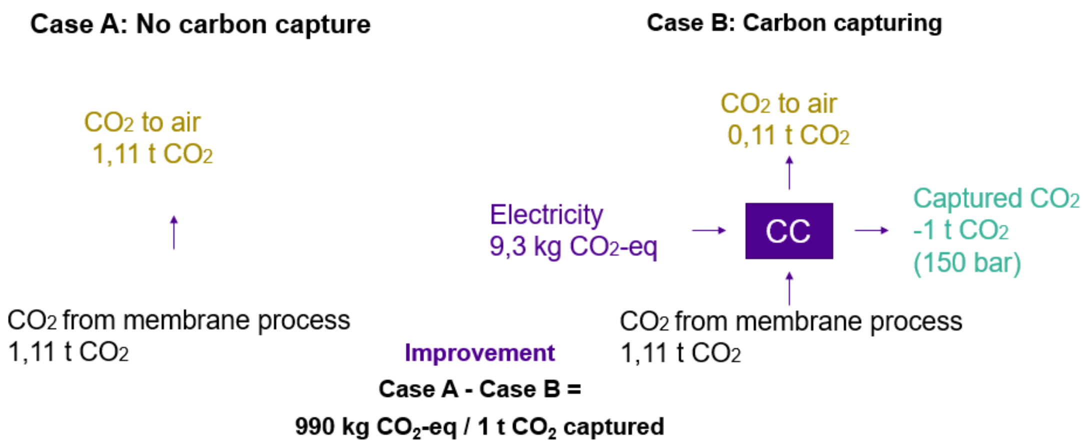

Figure 14. *Carbon capturing from the membrane upgrading process overall results.*

All the CO_2 emissions from biogas upgrading are biogenic because all the feed is biomass (Table 12). In this comparison study, the methane slip was the same in both cases, so it was excluded from the assessment. CO_2 concentration in biogas is fluctuating between 32 % and 42 %, due to the changes in feedstock and operational parameters (Al Seadi *et al.*, 2008). In case A biogenic emissions are 104 kg CO_2/MWh and 1110 kg CO_2/t CO_2 captured. The overall GWP with both functional units is 0 kg CO_2-eq.

Table 12. *Business as usual biogas membrane upgrading emissions to air calculated to the functional unit of 1 MWh of upgraded biomethane fuel and 1 t CO_2 captured.*

1 MWh upgraded biomethane			**1 t CO2 captured**		
Input	**Inventory**	**GWP**	**Input**	**Inventory**	**GWP**
-			-		
Output			**Output**		
CO_2 to air			CO_2 to air		
- Min	87 kg		- Min	930 kg	
- Max	134 kg		- Max	1430 kg	
- Av	104 kg		- Av	1111 kg	
	Total	**0 kg CO_2-eq**		**Total**	**0 kg CO_2-eq**

After applying simplifications to case B, the only flows were electricity to run the compressor and the CO_2 exhaust gas from the membrane separation process (Figure 14). 90% of the CO_2 can be captured and it will consume 8.2 kWh of electricity for 1 MWh of upgraded biomethane (Table 13). The total GWP of CC from membrane separation was -93 kg CO_2-eq/MWh biomethane. For capturing 1 t of CO_2, the electricity demand was 88 kWh, and the total GWP was -990 kg CO_2-eq/t CO_2 captured. Because the GWP of case A is 0 kg CO2-eq, the decarbonization potentials of CC are 93 CO_2-eq/MWh biomethane and 990 kg CO_2-eq/t CO_2 captured (Eq 3). The decarbonization potential of one Finnish biogas plant with a design capacity of 61 GWh/a was 5.7 kt CO_2-eq /a. The decarbonization potential equals 0.8% of Turku's annual emissions. The additional electricity demand for capturing the CO_2 was estimated to be 501 MWh/a which equals a 12% increase in the plant's current electricity demand.

The sensitivity of the electricity EF was assessed with a sensitivity analysis (Chapter 3.1.5). In ecoinvent 3.4. EF for high voltage production mix electricity was 0.262 kg CO_2-eq/kWh, which was double the EF of the electricity currently used in the studied biogas plant (0.106 kg CO_2-eq/kWh). The change lowered the annual decarbonization potential to 5.6 kt CO_2-eq/a from 5.7 kt CO_2-eq/a so the decarbonization potential loss was 1.2%. When comparing the current electricity EF to wind energy EF from ecoinvent 3.4 (0.015 kg CO_2-eq/kWh), the difference was 0.091 kg CO_2-eq/kWh. With wind electricity, the annual decarbonization potential was 0.7% more than with the current electricity EF.

Based on the sensitivity assessment it can be said that the source of the electricity did not affect the overall decarbonization potential significantly.

Table 13. *GWP of carbon-capturing with compressing from membrane separation process. The functional unit is 1 MWh of upgraded biomethane, and 1 t of CO_2 captured.*

1 MWh upgraded biomethane			**1 t CO_2 captured**		
Input	**Inventory**	**GWP**	**Input**	**Inventory**	**GWP**
Electricity			Electricity		
- Min	6.87 kWh	0.73	- Min	73.4 kWh	7.8
- Max	10.6 kWh	1.1	- Max	113 kWh	12.0
- Av	8.21 kWh	0.87	- Av	87.8 kWh	9.3
Output			**Output**		
CO_2 to air			CO_2 to air		
- min	8.70 kg	0	- min	93.0 kg	0
-max	13.4 kg	0	-max	143 kg	0
-Av	10.4 kg	0	-Av	111 kg	0
Compressed CO_2			Compressed CO_2		
- Min	78.3 kg	78.3	- Min	837 kg	-837
- Max	120.5 kg	-120	- Max	1287 kg	-1287
- Av	93.6 kg	-93.6	- Av	1000 kg	-1000
Total	**Min -80 kg CO_2-eq Max -200 kg CO_2-eq Av -90 kg CO_2-eq**		**Total**	**Min -830 kg CO_2-eq Max -130 kg CO_2-eq Av -990 kg CO_2-eq**	

Comparison of the current study to other studies was challenging due to the differences in biogas processes and the choice of functional units. Also, there were not many studies that quantified the decarbonization potential of the biogas plant because it is already a biogenic process, and the focus is more on the fossil carbon capturing LCAs. Even though IPCC has a prediction that 3.6 Gt of biogenic carbon must be sequestered by 2030 if the target of limiting global warming to under 2 °C (Rogelj *et al.*, 2018).

Suitable utilization opportunities for CO_2 from biogas plants were identified in the literature review (Chapter 2.4). For example, CO_2 can be used as a methanation agent in the biogas process. In the summertime, CO_2 can be used locally as a growth-boosting agent in commercial greenhouses. One opportunity would be utilizing CO_2 in the synthesis of different chemicals. Synthesis requires energy-intensive electrolysis of water to produce H_2 but combining the electrolysis with renewable energy production can be a possible way to utilize CO_2.

4.3 Decarbonization potential of replacing natural aggregate with recycled aggregate

The decarbonization potential of replacing natural aggregate with recycled aggregate was assessed with the tool developed in the study (Figure 15). The inventory data was collected from one company in the Turku region. Natural aggregate and recycled aggregate GWP were compared with data from one company in the Turku region. The assessment was done with the inventory data from one company in the Turku region and the SLCA approach was applied to it (Table 14). Both processes were currently in practice so the data was actual process data. The assessment showed the GWP difference between producing natural aggregate and recycled aggregate.

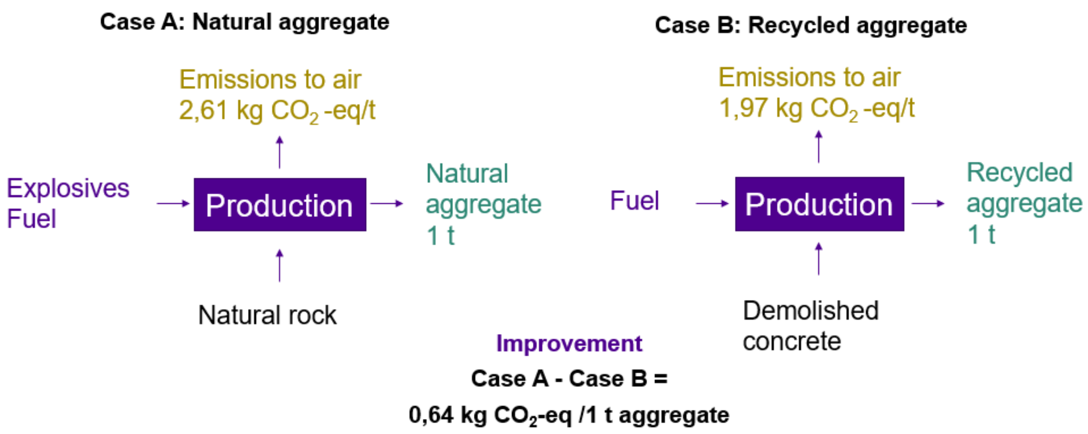

Figure 15. *Aggregate case overall results.*

The natural aggregate production emitted 2.61 kg CO_2-eq/t aggregate (Table 14). The emissions were mainly from fuel consumption, which contributed to 84.5% of the total emissions of the process. The recycled aggregate production emitted 1.97 kg CO_2-eq/t aggregate. 63% of the emissions are from fuel consumption during crushing. The rest 37% of the emissions were from transporting the demolished concrete to the production plant. The overall GWP difference was 0.64 kg CO_2-eq/t aggregate, but there was no true carbon sink in this case. The decarbonization potential from using the annual capacity of 163 000 t of natural aggregates was 9.97 kt CO_2-eq/a which is equal to 1.5% of Turku's emissions.

A sensitivity analysis of fuel choice and its effect on the total GWP of aggregate production was made based on assumptions that renewable diesel would lower the EF of diesel by 75-95% (Neste, 2022). The overall relationship between the natural and recycled aggregate production would not change due to unchanged diesel consumption and the achieved decarbonization potential would still be 7 kt CO_2-eq/a. The natural and recycled aggregate production GWP with renewable diesel were 0.95 kg CO_2-eq/t and 0.49 kg CO_2-eq/t respectively. The GWP reduction achieved with renewable diesel in the production process of both aggregates is significant because fuel consumption is the main contributor to the process.

Table 14. *GWP difference between natural and recycled aggregate production.*

Case A Natural aggregate			
Input	**Inventory**	**GWP**	**Contribution**
Natural rock	1 t		
Dynamite	0.011 kg	0.013	0.5 %
Kemiitti 510	0.33 kg	0.39	15 %
Fuel	0.85 l	2.20	84.5 %
Output			
Natural aggregate	1 ton		
Total	**2.61 kg CO_2-eq/t aggregate**		
Case B Recycled aggregate			
Input	**Inventory**	**GWP**	**Contribution**
Demolished concrete	1.003 t		
Fuel consumption	0.46 l	1.23	63 %
- Excavator	0.18 l	0.49	25 %
- Crusher	0.20 l	0.53	27 %
- Frontloader	0.08 l	0.22	11%
Transport	10.0 tkm	0.73	37%
Total	**1.97 kg CO_2-eq/t aggregate**		

Because of the system boundary defined in Chapter 3.2.3, the land use change was excluded from the assessment. Also, the material masses used in the assessment can have errors, because the inventory was calculated from annual averages (Chapter 3.2.3). It can not be guaranteed that all the fuel marked during the billing period was from the specific aggregate production so there can be errors in the calculation. Natural aggregate results were lower than the EDPs showed (3.1-3.6 kg CO_2-eq/t) (NCC Industry Nordic AB, 2020; Vilniaus karjerai, 2021). The construction material database collected from Swedish and Finnish EDPs had GWP values closer to the present study's results.

The database values ranged from 1.9 kg CO_2-eq/t to 2.7 kg CO_2-eq/t for coarse aggregate comparable to recycled aggregate (Vainio-Kaila, 2020).

Ghanbari *et al.* (2017) compared natural and recycled aggregate production with a fixed crushing system studied in Iran. The processes studied by Ghandari *et al.* (2017) utilized more equipment and the capacity was 200 t/h, 8h per day, the whole year around. Their results were 4.45 kg CO_2-eq/t natural aggregate and 1.25 kg CO_2-eq/t recycled aggregate. The natural aggregate emissions are close to double the present study's result. The recycled aggregate emissions are lower in the study by Ghanbari *et al.* (2017) compared to the recent study. These differences can be explained by different types of machinery and transportation distances. Based on the differences in EDPs and previous studies the present results are in a similar range, but the results may have significant variances between different plants.

4.4 Decarbonization potential comparison of the case studies

All the case studies resulted in annual decarbonization potential. The MSWI case, the biogas case, and the aggregate case had decarbonization potentials of 19.6 kt CO_2-eq/a, 5.7 kt CO_2-eq/a, and 9.97 kt CO_2-eq/a respectively. The MSWI and biogas cases created true carbon sink potentials when the aggregate case resulted in emission savings.

When comparing the operation emissions to capturing 1 t of CO_2, the environmental efficiency of the capturing method can be assessed. Biogas and MSWI case operation emissions were 9.3 kg CO_2-eq/t CO_2 captured and 171 kg CO_2-eq/t CO_2 captured. Even though the MSWI case had more decarbonization potential the operational emissions were 95% higher than in the biogas case.

Before the decarbonization potential can be realized, there can be other challenges along the way. For example, recycled aggregate is not still fully accepted in the industry in the Turku region, even though it has been in the market for years already. It can be explained by hesitant attitudes toward the end of waste products. In the aggrgate case there might be reagional limitation for recycled aggregate that can lower the decarbonization potential. Also, the technical maturity, investment, and operation costs affect the applicability of the presented methods for decarbonating the business area.

4.5 Method functionality and future improvements

The method can be used as the first quantification tool in the search for different decarbonization opportunities. Without any development, it can be used in the presented CE value chains. For further application to other value chains, the preliminary literature review and model development must be done first. The time investment to develop new models with the presented SLCA approach can be significant. Depending on the goal of the assessment the presented method can be one-dimensional with only GWP assessment and simplified processes.

The studied assessment method does not consider CO_2 utilization or storage opportunities, or the emissions related to the needed management of the captured CO_2. Assessing the realistic market for CO_2 is required before any of the presented CC applications can be implemented with environmental benefits. Currently, the required infrastructure for CO_2 utilization or storage is missing in Finland. The closest permanent CO_2 storage project is in the Netherlands and the transportation emissions can defeat the emission achieved emission reductions (*Project - Porthos*, 2021). During the discussions with industry representatives, it came up that, there is interest to study local CO_2 utilization applications but the timeframe for any concrete action was undefined.

The current method considers only the GWP and excludes all the other impact categories. Focusing only on one impact category can lead to burden shifting to other impacts such as acidification, eutrophication, or human toxicity. Adding more impact categories to the assessment would require more data so the simplicity would suffer from the addition.

In future studies, the simplification method could be compared to full LCA to assess the true effect of simplifying the assessment. A more detailed sensitivity analysis would increase the credibility and usability of the method. Adding more value chains and improvement scenarios would increase the applicability of the tool. Currently, the studied method can be used as case-by-case comparison studies where the assumptions apply to both cases and the assessment is seen as the first screening tool.

5. CONCLUSION

The study assessed SLCA approach development and testing in the CE value chain context. Three value chains were selected and assessed from the Turku region business area. The value chains were MSWI with MEA CC unit, biogas plant with CC from membrane separation, and comparison of natural and recycled aggregate production.

The calculation tool with the simplification method was possible to make and formulate. The method was sensitive to assumptions because it is possible to leave out impactful parts of the process during the inventory collection and the method itself does not have a built-in comparison to the full LCA of the assessed process.

The accuracy of the assumptions must be based on literature to avoid misleading simplifications, but it was possible to assess the scope of decarbonization potential with the simplified method presented in the current study. The method can be applied to other value chains, but the addition of new value chains requires time and expertise. Determining the most impactful flows for each new value chain will require a literature review and expertise to find the most suitable improvement scenarios. Also gathering emission data for new value chains can be time-consuming. The results from the method developed in the present study can be used in comparison studies after the first identification of improvement scenarios.

The answer to the research question of how SLCA can be applied to the CE context was a simple four-step process. At first literature review was conducted from the value chain in question, then the most impactful flows were determined and after that, the data collection was done. Finally, the method was applied to a spreadsheet calculator and the results were analyzed and interpreted.

The addition of the MEA CC unit to the MSWI plant resulted in decarbonization potentials of 830 kg CO_2-eq/t CO_2 captured and 563 kg CO_2-eq/t MSW incinerated. The annual decarbonization potential with MSWI plant capacity of 120 000 t MSWI/a resulted in a decarbonization potential of 19.6 kt CO_2-eq/a which is equal to 3% of Turku's emissions from 2019. Adding a CC unit to the biogas upgrading process resulted in a decarbonization potential of 990 kg CO_2-eq/t CO_2 captured and 93 kg CO_2-eq/MWh biomethane produced. The annual decarbonization potential with a biogas plant capacity of 61 GWh was 5.7 kt CO_2-eq/a which is equal to 0.8 % of Turku's emissions. Replacing natural aggregate with recycled aggregate resulted in a decarbonization potential of 0.64

kg CO_2-eq/t aggregate. The annual decarbonization potential with a plant capacity of 163 000 t of natural aggregates was 9.97 kt CO_2-eq/a which is 1.5% from Turku's emissions.

Biogas produces the smallest decarbonization potential, but the process to achieve it was very simple and efficient. MSWI has the biggest decarbonization potential, but the CC process requires infrastructure investments and uses more resources. The aggregate case does not provide a carbon sink, but carbon emission savings and it was in use already. The challenge in the aggregate case was the difficulty to change the attitudes toward the end of waste products.

Further development of the presented method would be including more impact categories in the assessment to ensure, that burden shifting does not happen. The current method does not include CO_2 management and utilization scenarios in the assessment. It is crucial to assess the environmental and economical feasibility of the suggested improvement scenarios before they are applied.

The presented method was intended to be used in case-by-case comparison studies for a simple first calculation step to quantify the decarbonization potential. Applying the method requires still expertise to avoid misleading assumptions, but the method can present valuable knowledge when comparing different improvement scenarios.

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APPENDIX 1: SNAPSHOTS FROM THE CALCULATOR

ILPO-calculator

The calculator is intended for circular economy value chains. Users can apply process data to check the decarbonization potential of 3 different cases: MSWI, biogas production, and recycled aggregate.

This calculator was developed in part of Krista Uusi-Kinnala's Master's Thesis at Tampere University.
Abhishek Singhal, Jere Lunden, and Tero Joronen are also been involved with the development of the tool.

Instructions:
Blue spaces are meant for process data or editing of emission factors.
Results can be found in the light blue box.
Inventory is presented below.

The calculator was developed as a part of the ILPO-project. Link to circhubs

CircHubs 6Aika Uudenmaan liitto Nylands förbund Euroopan unioni Vipuvoimaa EU:lta 2014–2020

MSWI plant with MEA CC

MSWI plant capasity	120000	t/a
Biogenic carbon share	50 %	default 50% [8]
Emission factors		
Biogenic carbon emitted	0	kg CO2 eq/kg CO2 [1]
Biogenic carbon captured	-1	kg CO2 eq/kg CO2 [1]
Fossil carbon emitted	1	kg CO2 eq/kg CO2 [1]
Fossil carbon captured	0	kg CO2 eq/kg CO2 [1]
H2O	3,5E-05	kgCO2/kg H2O [2]
Activated carbon	18	kg CO2 eq/1 kg of AC [3]
MEA	3	kg CO2 eq/1 kg MEA [4]
NaOH	1,4	kg CO2 eq/kg NaOH [4]
Electricity	0,131	kg CO2 eq/kWh [5]
Heat/ steam	41	kg CO2/GJ [5]
NH3 emitted	4,5414	kg CO2/kg NH3 [6]

Case A: Conventional MSWI [8]

Functional unit = 1 t CO2 captured			Functional unit = 1 t MSW incinerated			Emission factors	
Input:	Elemental flow		**Input:**	Elemental flow			
MSW	1,47	t	MSW	1	t		
Output:			**Output:**				
CO2	1175	kg	CO2	800	kg		
-BIO	588	kg	-BIO	400	kg	0	kg CO2 eq/kg CO2 [1]
-FOS	588	kg	-FOS	400	kg	1	kg CO2 eq/kg CO2 [1]
TOTAL			**TOTAL**				
GWP	588	**kg CO2 eq**	**GWP**	**400**	**kg CO2 eq**		

Case B: MSWI with MEA CC [7]

Functional unit = 1 t CO2 captured			Functional unit = 1 t MSW incinerated				
Input	Elemental flow		**Input**	Elemental flow			
CO2	1175	kg	CO2	800	kg		
H2O	840	kg	H2O	571,2	kg	0,000035	kgCO2/kg H2O [2]
Activated carbon	0,07	kg	Activated carbon	0,0476	kg	18	kg CO2 eq/1 kg of AC [3]
MEA	1,5	kg	MEA	1,02	kg	3	kg CO2 eq/1 kg MEA [4]
NaOH	0,12	kg	NaOH	0,0816	kg	1,4	kg CO2 eq/kg NaOH [4]
Electricity	100	kWh	Electricity	68	kWh	0,131	kg CO2 eq/kWh [5]
Heat/ steam	3,7	GJ	Heat/ steam	2,5	GJ	41	kg CO2/GJ [5]
TOTAL	171	kg CO2 eq	**TOTAL**	116	kg CO2 eq		
Output			**Output**				
CO2 pressurized	1000	kg	CO2 pressurized	680	kg		
-BIO	500	kg	-BIO	340	kg	-1	kg CO2 eq/kg CO2 [1]
-FOS	500	kg	-FOS	340	kg	0	kg CO2 eq/kg CO2 [1]
CO2 to air	175	kg	CO2 to air	120	kg		
-BIO	87,5	kg	-BIO	60	kg	0	kg CO2 eq/kg CO2 [1]
-FOS	87,5	kg	-FOS	60	kg	1	kg CO2 eq/kg CO2 [1]
NH3	0,15	kg	NH3	0,102	kg	4,5414	kg CO2/kg NH3 [6]
TOTAL			**TOTAL**				
GWP	**-241**	**kg CO2 eq**	**GWP**	**-163**	**kg CO2 eq**		

Results:

Decarbonisation potential	**19,6**	**kt/a**
Case A GWP (1 t MSW incinerated	400	kg CO2 eq/ 1 t MSW
Case B GWP (1 t MSW incinerated	-163	kg CO2 eq/ 1 t MSW
Emission reduction	**563**	**kg CO2 eq/ 1 t MSW**
Case A GWP (1 t CO2 captured)	588	kg CO2 eq/ 1 t CO2
Case B GWP (1 t CO2 captured)	-241	kg CO2 eq/ 1 t CO2
Emission reduction	**828**	**kg CO2 eq/ 1 t CO2**

Biogas membrane upgrading process with CC

Biogas composition (mol-%):			
	av	min	max
CO2	36 %	32 %	42 %
CH4	64 %	68 %	58 %

Biogas plant capasity	61000	MWh upgraded biomethane
Emission factors		
Biogenic carbon emitted	0	kg CO2 eq/kg CO2[1]
Biogenic carbon captured	-1	kg CO2 eq/kg CO2[1]
Electricity	0,106	kg CO2 eq/kWh [2]

Results:

Decarbonisation potential	**5,7**	**kt/a**
Case A GWP (1 MWh biomethane	0	kg CO2 eq/ 1 MWh
Case B GWP (1 MWh biomethane	-93	kg CO2 eq/ 1 MWh
Emission reduction	**93**	**kg CO2 eq/ 1 MWh**
Case A GWP (1 t CO2 captured)	0	kg CO2 eq/ 1 t CO2
Case B GWP (1 t CO2 captured)	-992	kg CO2 eq/ 1 t CO2
Emission reduction	**992**	**kg CO2 eq/ 1 t CO2**

Case A: Membrane upgrading process

ınctional unit = 1 MWh upgraded biomethan			**Functional unit = 1 t CO2 captured**			**Emission factors**
Input	Elemental flow		**Input**	Elemental flow		
-			-			
Output			**Output**			
CO2 to air [3,4]			CO2 to air [3,4]			0 kg CO2 eq/kg CO2[1]
-Average	104	kg	-Average	1111	kg	
-Min	87	kg				
-Max	134	kg				
TOTAL			**TOTAL**			
GWP	**0**	**kg CO2 eq**	**GWP**	**0**	**kg CO2 eq**	

Case B: Membarne upgrading process with CC [3]

ınctional unit = 1 MWh upgraded biomethan			**Functional unit = 1 t CO2 captured**			
Input	Elemental flow		**Input**	Elemental flow		
Electricity [5]			Electricity [5]			0,106 kg CO2 eq/kWh [2]
-Average	7,4	kWh	-Average	79	kWh	
-Min	6,2	kWh				
-Max	9,5	kWh				
TOTAL	0,78	kg CO2 eq	**TOTAL**	8,4	kg CO2 eq	
Output			**Output**			
CO2 to air			CO2 to air			0 kg CO2 eq/kg CO2[1]
-Average	10,4	kg	-Average	111	kg	
-Min	8,70	kg				
-Max	13,4	kg				
CO2 pressurized			CO2 pressurized			-1 kg CO2 eq/kg CO2[1]
-Average	93,6	kg	-Average	1000	kg	
-Min	78,3	kg				
-Max	120,5	kg				
TOTAL			**TOTAL**			
GWP	**-93**	**kg CO2 eq**	**GWP**	**-992**	**kg CO2 eq**	
min	-78	kg CO2 eq				
max	-119	kg CO2 eq				

Natural aggregate VS Recycled aggregate

Natural aggregate annual process data:		
Natural aggregate production capacity	162852	t/a
Fuel consumption	135133	l/a
Explosives used:		
-Dynamite	1807,2	kg/a
-Kemiitti 510	54369	kg/a
Recycled aggregate annual process data:		
Recycled aggregate production capacity	15497	t/a
Demolition waste	15547,1	t/a
Fuel consumption	7182	l/a
Transportation distance from demolition sit	14	km
Mass transported	11098,4	t
Emission factors		
Dynamite (EDP)	1,1268	kg CO2 eq/kg dynamite produced and used [1]
Kemiitti 510 (EDP)	1,1674	kg CO2 eq/kg kemiitti 510 produced and used [2]
LIPASTO Mpö (diesel)	2,65524	kg CO2 eq/l fuel [3]
LIPASTO soil transport (full loads)	0,073	kg CO2 eq/tkm [4]

Results:

Decarbonisation potentia	**9,97 kt/a**
Case A GWP (1 t aggregate	2,61 kg CO2 eq/ 1 t aggregate
Case B GWP (1 t agrregate)	1,96 kg CO2 eq/ 1 t aggregate
Emission reduction	**0,64 kg CO2 eq/ 1 t aggregate**

Case A: Natural aggregate

Functional unit = 1 ton aggregate

Input	Elemental flow		Emission factors	
Natural Rock				
Dynamite	0,0111	kg	1,1268	kg CO2 eq/kg dynamite produced and used [1]
Kemiitti 510	0,334	kg	1,1674	kg CO2 eq/kg kemiitti 510 produced and used [2]
Fuel consumption	0,830	l	2,65524	kg CO2 eq/l fuel [3]
Output				
Natural aggregate	1	t		
TOTAL				
GWP	**2,61**	**kg CO2 eq**		

Case B: Recycled aggregate

Functional unit = 1 ton aggregate

Input	Elemental flow			
Demolition waste	1,003	t		
Fuel consumption	0,463	l	2,65524	kg CO2 eq/l fuel [3]
Transport	10,03	tkm	0,073	kg CO2 eq/tkm [4]
Output				
Recycled aggregate	1	t		
TOTAL				
GWP	**1,96**	**kg CO2 eq**		