Bening Mayanti

Toward circularity

Life cycle-based approach in waste management

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Author: Bening Mayanti https://orcid.org/0000-0001-5073-7375

Supervisor(s): Professor Petri Helo
University of Vaasa. School of Technology and Management, Industrial Management.

Custos: Professor Petri Helo
University of Vaasa. School of Technology and Management, Industrial Management.

Reviewers: Professor Shams Rahman
RMIT University, Royal Melbourne Institute of Technology, School of Accounting, Information Systems and Supply Chain.

Professor Angappa Gunasekaran
Penn State Harrisburg, School of Business Administration.

Opponent: Professor Kim Hua Tan
Nottingham University Business School, University of Nottingham.

Tiivistelmä

Nykyinen kertakäyttöelämäntapa aiheuttaa painetta ympäristölle. Monia raaka-aineita, joita käytetään taloudessa, hyödynnetään vain lyhyen aikaa ja hävitetään nopeasti. Tämän väitöskirjan tavoitteena on arvioida taloudellisia ja ympäristövaikutuksia yritysten siirtymisessä kohti kiertotalouden (CE) käytäntöjä, joiden avulla pyritään arvon säilyttämiseen mahdollisimman pitkään. Tutkimus toteutettiin tarkastelemalla kiertotalouden käsitteitä ja esittämällä ratkaisumalleja tapaustutkimuksiin, joissa keskityttiin tuotteen elinkaaren loppuvaiheeseen (EoL). Elinkaariarviointi (LCA) oli näissä tärkein työkalu erilaisten kiertoskenaarioiden ympäristövaikutusten arvioinnissa. Tämä työkalu yhdistettiin elinkaarikustannuslaskentaan (LCC) taloudellisen suorituskyvyn arvioimiseksi. Kolme tapaustutkimusta toteutettiin Suomessa: (1) siirtyminen biojätteen lajittelukeräykseen, (2) maatalouden muovijätteen kierrätysjärjestelmän suunnittelu ja (3) jätteen energian optimointi. Tulokset osoittivat, että kiertotalouden avulla voidaan kattaa useita arvoketjun näkökohtia; käyttöönotto voidaan toteuttaa millä tahansa arvoketjun tasolla, ja eri sidosryhmät voivat lisätä kiertoa eri toimien kautta. Tulokset viittaavat siihen, että kierron lisääminen EoL-vaiheessa voisi parantaa arvon säilyttämistä uusiomateriaalituotannon, jätteenkäsittelyn sivutuotteiden ja energian talteenoton avulla. Tyyppitapausten perusteella yritysten siirtyminen kiertotalouskäytäntöihin osoittautui sekä taloudellisesti ja ympäristön kannalta kannattavaksi. Työn tulokset ovat havainnollistaneet sidosryhmien yhteistyön tärkeyttä. Kierron rakentaminen voi vaikuttaa kaikkiin toimitusketjun toimijoihin, mukaan lukien valmistus, energiantuotanto ja yhteiskunta laajemmin. Tutkimus osoitti, että tuotteiden tai palveluiden ympäristövaikutusten kvantitatiivinen mittaaminen on tärkeää, ja LCA on edelleen sopivin väline tulosten kvantifiointiin ja erilaisten vaihtoehtojen keskinäiseen arviointiin. Elinkaarilaskelmaan yhdistettynä elinkaarikustannuslaskentaan saadaan aikaan kattavampia tuloksia, joilla voidaan vertailla ympäristö- ja talousnäkökohtien mahdollisia ristiriitoja. Kiertotaloustyö on aloitettava jostain, ja se voi alkaa siitä, että organisaatiot mittaavat ympäristötehokkuuttaan rakentaakseen parempia vaihtoehtoja, määritelläkseen tavoitteitaan ja edistääkseen kiertojen kehittymistä pitkällä aikavälillä.

Asiasanat: Kiertotalous, kiertotalous jätehuollossa, suljettu kierto, elinkaariarviointi, elinkaarikustannuslaskenta, optimointiongelma

Abstract

Our current “throwaway” lifestyle places great strain on the environment; resources that enter the economy remain for only a short period and are quickly disposed of. This dissertation aims to evaluate the economic and environmental impacts of shifting toward more circular economy (CE) practices that advocate value retention for as long as possible within the economy. The research was carried out by conceptualizing CE and solving real cases focusing on the product end-of-life (EoL) stage. Life cycle assessment (LCA) was the main tool used to assess environmental impacts of different circular scenarios. The tool was paired with life cycle costing (LCC) to evaluate economic performances. Three cases in Finland were assessed: shifting toward source-separated biowaste collection, establishing an agricultural plastics waste recycling system, and waste-to-energy optimization. It was found that CE covers multiple aspects within the value chain; thus, its adoption model can occur at any stage of the value chain, thereby enabling various stakeholders to be more circular through different actions. The cases suggested that being more circular at the EoL stage may improve value retention through secondary material production, waste treatment by-products, and energy recovery. Shifting toward circularity was shown to be economically and environmentally viable. The dissertation illustrated the importance of stakeholders’ collaboration because a circular approach could affect all actors within the supply chain, including manufacturing, the energy sector, and society. The study showed that it is important to quantify environmental impacts of products or services, and to date, LCA remains the most suitable tool for quantifying results and evaluating options. In addition, a combination with LCC will provide more comprehensive results to anticipate any trade-off between environmental and economic aspects. CE must start somewhere, so let it start with organizations evaluating their environmental performance to identify better alternatives, define targets, and foster circularity in the long run.

Keywords: circular economy, circular waste management, closed loop, life cycle assessment, life cycle costing, optimization problem

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I would like to convey my sincere gratitude to Professor Petri Helo, my supervisor, for the endless support and guidance through the doctoral program. Thank you for all the discussions, encouragement, and patience – and for simply being all ears during my nonsensical rants. Thank you for believing in me (I'm not crying).

To my family, especially my mom and my dad: It is not always rainbows and butterflies with you, but I would not be who I am today if you had not provided me with opportunities that you never had.

"When the big things feel out of control, focus on what you love right under your nose." - *The Boy, the Mole, the Fox and the Horse*

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Abbreviations

AD	Anaerobic digestion
B-NL	Biowaste new legislation
APW	Agricultural plastic waste
BE	Bioeconomy
C2C	Cradle-to-cradle
CE	Circular economy
CLCC	Conventional life cycle costing
ELCC	Environmental life cycle costing

EoL	End-of-life
FS	Fossil resource scarcity
FU	Functional unit
GSCM	Green supply chain management
HT-C	Human carcinogenic toxicity
HT-NC	Human non-carcinogenic toxicity
IE	Industrial ecology
LCA	Life cycle assessment
LCC	Life cycle costing
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LIME	Life cycle impact assessment method based on endpoint
Mg	Megagram
MOO	Multi-objective optimization
MW-NL	Mixed waste new legislation
MW-OL	Mixed waste old legislation
PSS	Product service system
SCM	Supply chain management
SLCC	Social life cycle costing
TA	Terrestrial acidification
WC	Water consumption
WM	Waste management
WtE	Waste-to-energy

Publications

This doctoral dissertation consists of four papers prepared for scientific journals. The papers represent different sub-subjects within the field of circular economy. In the text, the papers are referred with Roman numerals.

I. Mayanti, B., & Helo, P. (n.d.). Circular economy: From conceptualization to database tool of practical implementation. [Manuscript under review by] *International Journal of Environmental Studies*.

II. Mayanti, B., & Helo, P. (2022). Monetary and environmental damage cost assessment of source-separated biowaste collection: Implications of new waste regulation in Finland. *Waste Management & Research: The Journal for a Sustainable Circular Economy*. https://doi.org/10.1177/0734242x221123492.

III. Mayanti, B., & Helo, P. (2022). Closed-loop supply chain potential of agricultural plastic waste: Economic and environmental assessment of bale wrap waste recycling in Finland. *International Journal of Production Economics*, 244, 108347. https://doi.org/10.1016/j.ijpe.2021.108347.

IV. Mayanti, B., Songok, J., & Helo, P. (2021). Multi-objective optimization to improve energy, economic and, environmental life cycle assessment in waste-to-energy plant. *Waste Management*, 127, 147–157. https://doi.org/10.1016/j.wasman.2021.04.042.

1 INTRODUCTION

1.1 Circular economy

Humanity has embraced a "throwaway" lifestyle in which the linear system of take-make-dispose becomes the norm and planned obsolescence part of a company's business model. The history of throwaway living dates back to the 1920s, when automobile and lightbulb manufacturers boosted sales by coming up with new designs for their cars every year and limiting the lightbulb lifespan, respectively (Arablouei & Abdelfatah, 2019; Murray, 2022). The linear system creates a high demand for raw material input and generates significant waste. Annually, more than 100 billion Mg resources are consumed – triple the amount consumed in 1970 – and this figure may double by 2050 (McGinty, 2021). There has been growing attention paid to shifting from the linear system toward a system that can decouple economic growth from environmental degradation, called a circular economy (CE) (e.g., Ellen MacArthur Foundation, 2020; Ghisellini et al., 2016). It has been argued that circularity may be achieved through narrowing, slowing, and closing the loops (Bocken et al., 2016). These notions emphasize resource efficiency, prolonging resources within the economic system, and transforming waste into a resource. In general, the economic system and its relation to resource management can be classified into linear, recycling, and circular, as illustrated by Figure 1 (Rli, 2015).

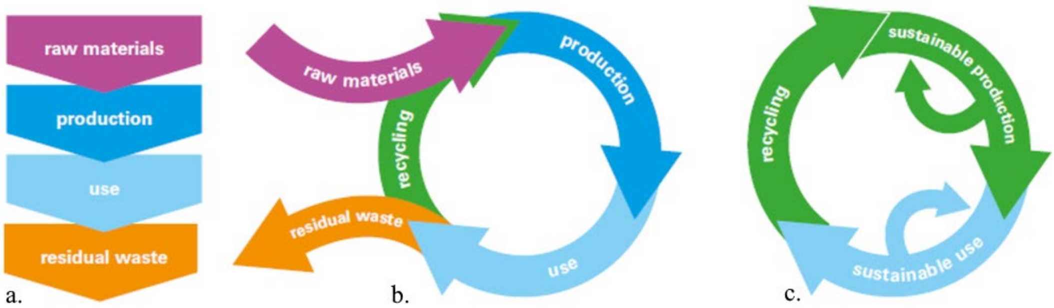

Figure 1. Difference between a) linear economy, b) recycling economy, and c) circular economy (Rli, 2015)

In the linear system, resource flows from the raw material until disposal. Meanwhile, the main difference between the recycling economy and fully CE is that the former still needs an input of raw material and generates waste, whereas the latter has a fully closed loop. The recycling economy also relies on energy consumption from external production, predominantly fossil sources. In Paper I, CE was defined as *"a regenerative economic model focusing on resource flow and management through the use of renewable resources, resource efficiency, prolonging resources at the consumption stage, and recirculating resources from discarded products into the value chain, enabled by research and technology development, the business model, consumers, and policy*."

Due to the potential gains offered by CE, influential actors such as companies and the government are interested in implementing the system. Because CE requires system-level change, it involves multiple stakeholders addressing the whole supply chain, including raw material extraction, design, production, consumption, and waste management (WM) systems (Reike et al., 2018); that is, to be successful, CE demands the active engagement of stakeholders through top-down and bottom-up approaches (Grafström & Aasma, 2021). The potential benefits of CE include protecting the environment while realizing economic opportunities of up to 4.5 trillion USD through waste reduction, innovation, and job opportunities (McGinty, 2021). Moreover, adopting circularity will make organizations less dependent on the supply of virgin materials for their production.

However, it is important to bear in mind that CE is not a panacea for our planet's environmental degradation. Indeed, it may be technically impossible to go fully circular, due to the laws of thermodynamics, because infinite recycling may require immense energy and will be incomplete due to its waste of by-products (Korhonen, Nuur, et al., 2018; van den Bergh, 2020). However, CE can offer benefits through cascading material or energy to maximize value retention for as long as possible in the economic system. However, the potential benefits are still rarely translated into circular action. In Europe, many food companies such as Danone, Coca-Cola, Nestle, and Ferrero failed to fulfill their plastic pledges with only 12% of pledges were achieved (Schacht, 2022). When this situation occurs, they will either not address it or will move the goal post farther into the future. Organizations still view circularity as a cost driver; about half of executives surveyed in a study by (Liebig et al., 2012) said that circular attitudes are prevention measures to avoid stricter regulation in the future that may be more costly. However, more circular approaches have also been shown to be financially viable; companies such as Hewlett-Packard can gain about 9% higher net profit margin (Liebig et al., 2012). Fairphone is another electronic equipment company that illustrates the viability of a more circular approach. It offers long-lasting phones and aims to be electronic waste-neutral; in 2020 they took back 18% of the phones they sold for reuse and recycling (Fairphone, 2022).

Shifting toward circularity requires suitable approaches that are not easy to implement. The value chain extends from raw material extraction to the end-of-life (EoL) stage, where key stakeholders vary. A concrete example can be found in Interface, a global floor covering manufacturer that learned the hard way to employ the right circular approach only after struggling for years using the wrong circular approach (Atasu et al., 2021). They started with a leasing program requiring the customer to pay a monthly fee for the covering and maintenance. The customers opted to buy instead of lease because the care was left to a janitorial service, giving them no reason to pay high maintenance costs. The program was then stopped, and they switched to producing modular floor covering using recyclable materials, resulting in an overall carbon reduction of 69%. Their experience shows that stakeholders play an important role in determining the success of circular actions. It also illustrates that there are no single routes to shift toward circularity which requires suitable pathways. This realization prompts the present work, in which there is an emphasis on how practitioners need analytical approaches to assess their products or services and assist them with granular circular thinking. The approaches should be able to measure the environmental and economic impacts of their current practices and possible circular alternatives, thereby assisting them in decision-making.

Companies should also possess appropriate knowledge regarding possible circular options to pursue. The knowledge is important considering a significant gap between CE theory and practice (Barreiro-Gen & Lozano, 2020). Thus, Paper I provides practical implementations of CE categorized by different product life cycle stages, including raw material, design, distribution, use or consumption, and EoL to expedite the transition. Related stakeholders should know their alternatives to avoid becoming stuck with one unfeasible option.

1.2 Circular economy and waste management: Prior works

Research around CE is progressing. Several key works can be found that highlight different aspects of CE. Kirchherr et al. (2017) conceptualized CE by analyzing 114 definitions. The CE dimensions covered reduce, reuse, recycle, system perspective, and economic prosperity. Kirchherr et al. (2017) emphasized that the core of CE lies in phasing out the EoL notion, which can be achieved on different levels, such as the micro level (products, consumers, companies), meso level (eco-industrial parks), and macro level (city, country, region). Business model and product design to transition toward a circular economies are also important topics (Bocken et al., 2016). The study discussed that a circular economy was about resource management within the supply chain; thus, circular strategies can be categorized as narrowing the loop, slowing the loop, and closing the loop. Companies can use resources more efficiently to produce products, prolonging the duration of products within the supply chain or re-entering secondary material back into the economy. Some reviews focused on CE's historical development, feature, characteristics, and trajectory (Blomsma & Brennan, 2017; Ghisellini et al., 2016). Ghisellini et al. (2016) highlighted the principles of CE: design, reduction, reuse, recycling, material classification (technical and biological), and renewable energy. They also explained how CE worked on the micro, meso, and macro levels while underlining the potential rebound effect of efficiency strategy within CE. Although CE has gained traction in the past years, the root of the concept is not new. CE provides a new framing in the resource and waste management discourse, which focuses on extending the resources' productive lives (Blomsma & Brennan, 2017). Others have argued that CE contributes little to new scientific discourse. That small contributions only focus on how the material should stay longer within the economy and the possibilities of implementing sharing economy (Korhonen, Honkasalo, et al., 2018). The study also underlined that CE lacks focus on the sustainable consumption area.

CE transforms how resources should be managed throughout the value chain. Previous CE research has heavily emphasized the EoL stage; however, Paper I argues that CE encompasses the whole value chain. Nevertheless, EoL management is important in closing the loop and conserving resource. Moreover, the overall EoL management remains poor, leaving much room for improvement. Globally, about 37% of waste is disposed of in various forms of landfill (only 3% of landfills are equipped with gas collection equipment), 33% of waste goes to open dumping, and only about 13.5% is being recycled (The World Bank, 2018). Clearly, humanity is failing to recover potential resources from about two-thirds of all waste. These dynamics between waste management (WM) and CE have become a topic of interest being widely researched by scholars.

On the academic level, the interest in CE and WM can be illustrated by the results of a literature search in Scopus using the keywords "circular economy" AND "waste management." The search was conducted for English documents within 2016–2020 in engineering, science, business, and management Figure 2 shows the bibliographic coupling of 789 records, illustrating the relatedness of articles based on references they share among countries. Each country has at least ten citations and ten documents as cut-off criteria. Node size represents the number of articles, whereas the color indicates the cluster, which is a closely related node. The key players are Spain, the United Kingdom, and Italy, with documents of 114, 106, and 83, respectively.

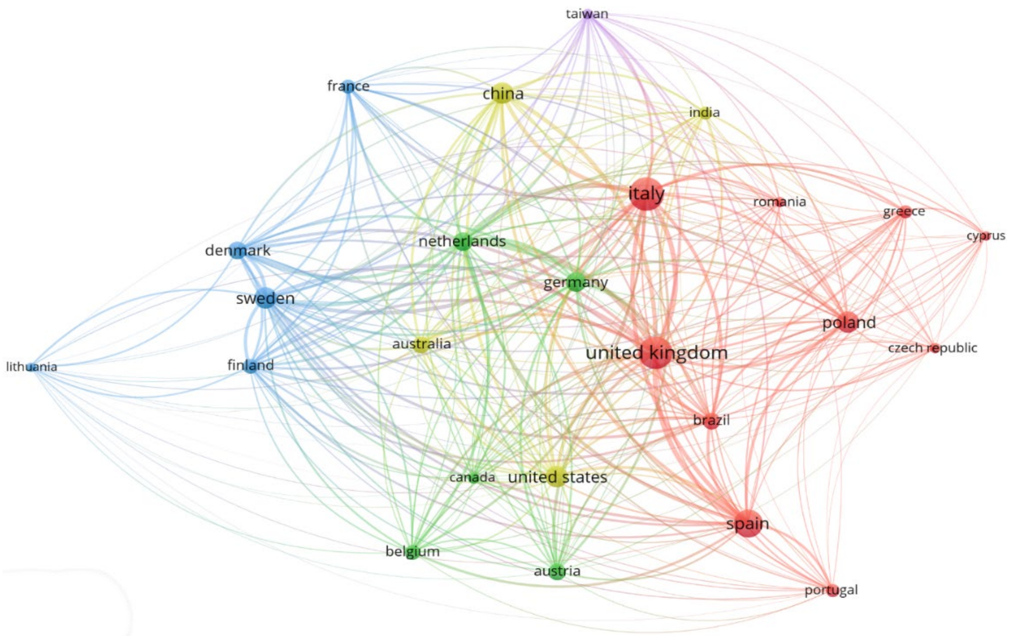

Figure 2. Bibliographic coupling among countries for research within CE and WM

Because the focus of the study was within the Finnish context, having an idea of how EU member nations are ranked in terms of CE performance can provide more context. The results of bibliographic coupling from the academic research scientific area can be assessed against the circularity ranking done by a media *Politico* ranks countries based on municipal waste generation (kg/year/capita), food waste (kg/year/capita), municipal recycling rate, share of traded goods that are recyclable, material reuse rate, patents related to CE, and investment in the CE sector. Figure 3 shows that the top three countries were Germany, the United Kingdom, and France (Hervey, 2018).

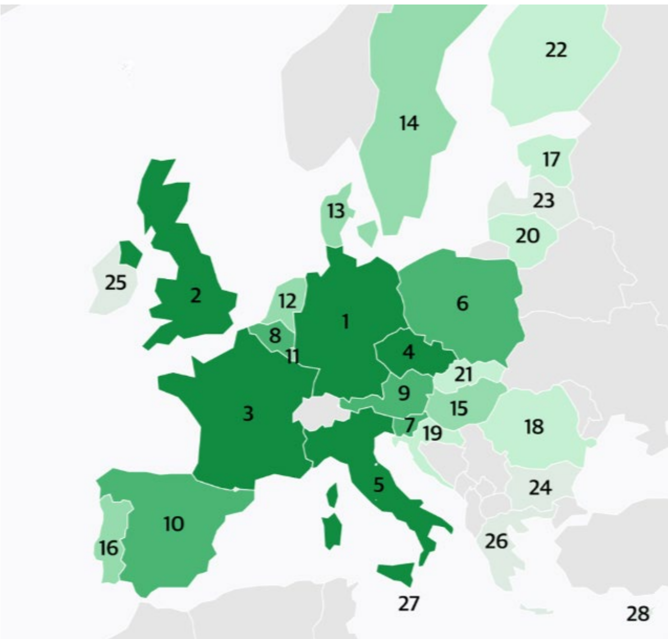

Figure 3. Circular ranking in EU (Hervey, 2018)

Bibliographic coupling was also applied among sources, with the criteria of a minimum of 50 citations, yielding 19 journals (Figure 4). The highest number of documents were found in *Resource, Conservation, and Recycling* (99), *Waste Management* (93), and *Journal of Cleaner Production* (78). These journals keep up with the latest developments in CE and WM through their focus on resource management, cleaner production, and EoL management.

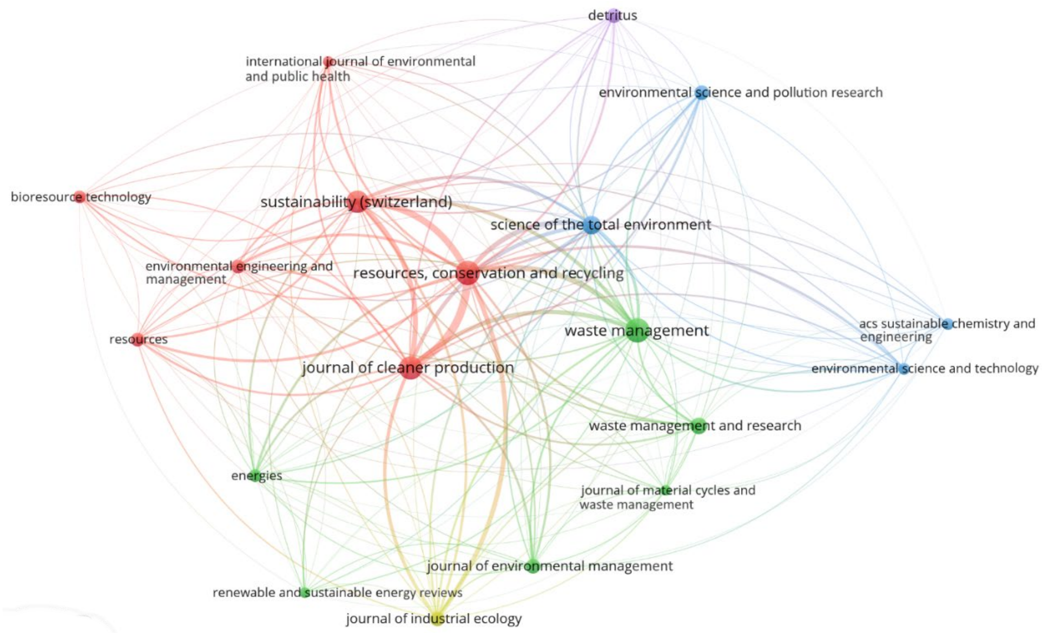

Figure 4. Bibliographic coupling among journal sources for research within CE and WM

Within a CE context, waste must be phased out, and the business model should shift its focus to service instead of ownership (Werning & Spinler, 2020). Although completely eliminating waste may not be possible, CE advocates optimizing value retention at different product life cycle stages. When products cannot be reused, repaired, refurbished, remanufactured, or repurposed, the end-of-life (EoL) products must be managed by following priority orders, namely recycling and recovery (energy) (Potting et al., 2017). Different approaches to handling waste will generate different environmental as well as economic outcomes. Before any waste management approach is selected, it should be assessed based on life cycle thinking to verify that it may prove beneficial (European Commission, 2008). The benefit can be obtained from avoiding production that consumes virgin material due to substituting recycled material or energy substitution from waste treatment.

Life cycle assessment (LCA) is a quantitative tool to assess environmental impacts caused by products or services. By using LCA, practitioners can quantify the environmental performance of different value chain configurations, making LCA a perfect pairing with CE principles. Various circular strategies can be compared to ensure the highest environmental benefits. At the EoL stage, it is sometimes difficult to determine the best circular waste management strategies to apply, considering their potential environmental impacts and benefits. LCA is a great tool to evaluate these strategies as well as assist decision-makers in defining and setting environmental targets. The tool can be combined with life cycle costing (LCC) to provide more comprehensive insights into the environmental and economic aspects. LCC itself can be categorized into conventional LCC (CLCC), environmental LCC (ELCC), and social LCC (LCC) (Hunkeler et al., 2008). CLCC is traditional financial assessment or cost accounting, ELCC is conducted to complete LCA with the same boundaries and functional unit, and SLCC expands the CLCC by including externalities (Martinez-Sanchez et al., 2015).

The ongoing attempts at continuous change and improvement in the waste management sector create a need for LCA models, and the shifting trends toward circularity make it important to study how these modifications affect environmental and economic outcomes. This study covers those outcomes to provide recommendations for decision-making based on scientific evidence.

1.3 Aim and thesis structure

The CE concept is critically important in retaining the value of products throughout the supply chain, and it changes how we view waste. Numerous studies have shown the close conceptual links between CE and WM (e.g., Blomsma & Brennan, 2017; Kirchherr et al., 2017), including its potential economic as well as environmental benefits (e.g., Ellen MacArthur Foundation, 2020; McGinty, 2021). However, the applied level of the studies regarding CE and WM focus mainly on the environmental aspects on the treatment stage (e.g., Bianco et al., 2021; Cascone et al., 2020). Existing studies also frequently exclude the collection stage, which is known to be expensive and requires proper planning approaches, such as route optimization. As a result, there is a lack of research that provides more comprehensive outcomes combining economic and environmental perspectives; a trade-off between these two aspects can potentially occur as well, thereby affecting decision-making. Failing to systematically address the costs issue can lead to unintended higher expenses, which may create problems among stakeholders.

This dissertation investigates the economic and environmental implications of transitioning toward more CE practices. The focus is on the EoL stage, where waste is transformed back into resources so that its value as material and/or energy can remain within the economic system for as long as possible. The study developed and employed a methodology to comprehensively assess economic and environmental impacts, after which the methodology is applied to real cases in Finland. There is a need for such practical research because Finland has amended the Waste Act (646/2011) and adjusted its recycling target to 55% in 2025, 60% in 2030, and 65% in 2035. This schedule highlights the need to shift the WM options from landfilling or energy recovery into material recovery to achieve the targets. The shift calls for assessment to improve decision-making and implementation, and to anticipate unexpected outcomes. Additionally, the dissertation intends to obtain a current perspective regarding the relationship between WM and CE as well as the adoption model of circular actions within value chain. The aims of the research were achieved by answering the following research questions:

RQ.1 What is the CE, and what is its relationship with WM?

To transition toward a more circular practice, one should understand the CE concept, its building blocks, its enablers, and possible adoption models within the value chain. Understanding the CE concept could assist in setting up the vision and goals for CE, while knowing more detail about CE strategies and approaches could support an organization to expedite CE by implementing concrete action.

RQ.2 What are the environmental and economic impacts of implementing a more circular waste management practice?

Implementing a more circular strategy is usually deemed more environmentally friendly—and more expensive, because it requires additional planning, new infrastructure, and labor. For instance, a source-separated waste collection will require a new collection system that can contribute up to 70% of total waste management costs. When switching from energy recovery to material recovery, it is uncertain whether the recycled material will be marketable, considering the low price of virgin material. These few aspects can cast doubt on whether one should consider a more circular strategy in managing their waste. Thus, environmental and economic assessments are pivotal to providing more comprehensive knowledge so that concerned actors can make informed decisions.

RQ.3 How is it possible to be more circular in energy recovery when waste management options cannot be switched?

In CE, material or resource recirculation is prioritized over energy recovery. However, there are situations where switching the waste management option is not possible. For instance, in a sparsely populated area, multiple types of waste are collected as mixed waste and go into the incinerator. Phasing out that waste remains impossible, so there is always a portion that goes into the incinerator (e.g., the loss/residual in the plastic recycling process or material that has been repeatedly recycled and cannot be used anymore). In such a situation, an optimization process in the WtE plant that adjusts the operating parameter can be applied to improve technical, environmental, and economic outcomes.

Each research question was addressed through papers presented in Table 1.

Table 1. Contribution of each paper in answering research questions

Paper	RQ to answer	Primary WM stages	Approach
I	RQ 1	General overview of CE in value chain	Narrative review
II	RQ 2	Collection, transportation, and treatment using anaerobic digestion (AD)	Cost model, damage cost, optimization environmental problem
III	RQ 2	Collection, transportation, treatment (mechanical recycling)	LCA, ELCC, optimization problem
IV	RQ 3	Treatment using WtE	LCA, cost model, optimization problem

2 MATERIALS AND METHODS

2.1 Case studies

Moving toward a more circular practice will require numerous changes on a practical level, and evidence is needed to persuade organizations to shift toward a more circular practice. Such evidence can be obtained through case studies where the methods show their reliability and the results indicate potential benefits. Thus, the case studies are a cornerstone of this dissertation. Korhonen et al. (2018) also have emphasized that the benefits of going circular are a matter that requires case-by-case assessment. On a more specific level, the case studies in Finland were used because the stakeholders were eager to make the shift and were less concerned about knowing the possible outcomes, due to the ambitious recycling targets set by the Finnish government. This is particularly true for Papers II and III, where top-down (legislation change) and bottom-up (waste collection company's initiative) approaches, respectively, drove the changes. Knowing possible outcomes from the change could help the actors to make decisions, improve the implementation stage, and anticipate undesirable situations. Although similar studies may have been applied before (e.g., Cascone et al., 2020; Faraca et al., 2019), generalization of the results may not be possible due differences in factors such as demographics, land mass, climate, and energy mix.

The focus of this dissertation was biowaste, agricultural plastic waste (APW), and mixed waste. In general, high income countries generate 32% biowaste, and in Helsinki the proportion goes up to 40% (HSY, 2021; The World Bank, 2018). Transforming the treatment of biowaste to achieve better environmental and economic outcomes is important. Moreover, new Finnish legislation has imposed stricter rules: biowaste must be collected separately for residential properties with at least five apartments, compared to ten apartments in the old legislation. For the APW, there is an untapped potential concerning the waste that is primarily landfilled instead of recycled. APW is also made of a limited range of resin, making it a good recycling input. Finally, mixed waste is a practice still found in many places, including Finland, and especially in sparsely populated areas where source-separated waste becomes too expensive. Thus, there is a need to investigate the possibility of improving energy recovery from WtE when changing treatment options is not possible.

The dissertation consists of four papers built on a review of CE and three case studies with a distinct focus on waste management. Each case applied a certain type of life cycle-based approach complemented by other approaches (Table 1). Figure 5 illustrates the papers' coverage of different life cycle stages of products.

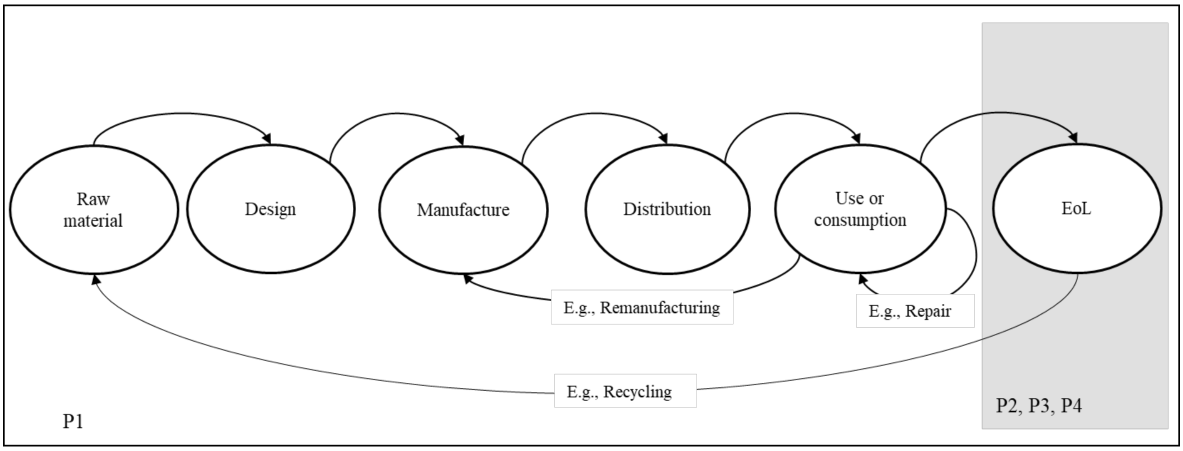

Figure 5. The papers' coverage within context of product life cycle

Paper I presents the review of CE in the context of the product life cycle. It was conducted using narrative review where all the data sources were from literature. It describes CE building blocks, definition, strategies, and approaches, including a broad list of practical implementations of CE in different stages of the product life cycle. Papers II, III, and IV focus on the EoL stage of products, which covers waste generation, collection, transportation, and treatment. Different types of waste (biowaste, agricultural plastic waste, and mixed waste) and treatments (AD, WtE, and recycling) were investigated using a combination of LCA, LCC, and optimization problems. Paper II emphasized the development of a system for source-separated biowaste collection. Separating waste from the source aims to improve the value recovered from the waste; however, waste collection is known to be costly. Building and applying a cost model is explored in Paper II. In Paper III, the "closing the loop" strategy was investigated through economic and environmental assessment of recycled APW. This strategy aimed to tap into the potential of plastic waste film from the agricultural sector. Paper III showed that 70% of APW is landfilled, 10% is recycled, and that open burning is still a common practice. Paper IV focused on maximizing energy recovery from the WtE plant by changing operating parameters while the inputs remain the same. These four papers build the dissertation and tackle specific research questions within this study. Conclusions and recommendations were established based on the optimized solution and the results of economic and environmental performance obtained in this research.

2.2 Life cycle assessment

LCA is a standardized environmental assessment tool used to evaluate the environmental performance of a product or service (ISO, 2006a, 2006b). LCA consists of four distinct phases, as illustrated in Figure 6.

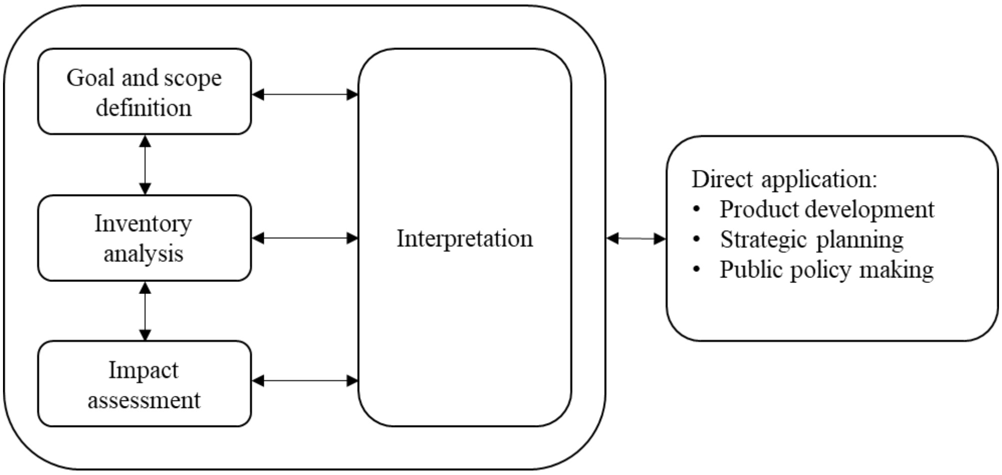

Figure 6. Four phases in LCA (ISO, 2006a, 2006b)

2.2.1 The goal, scope, and system description

LCA starts with specifying the goal and scope of the study related to the intended application. For WM application, the goal is related to LCA accounting, hotspot analysis, comparison between different waste management options, and benefit from waste treatment (e.g., recycled material) (Dahlbo, 2018; Gala et al., 2015). Scope definition specifies the system to be assessed, including functional unit (FU), system boundaries, assumptions, and level of details (Baumann & Tillman, 2004). The FU serves as a quantitative description for the system's main function under investigation and allows equal comparison of various systems that provide the same service/function (Dahlbo, 2018). Laurent et al. (2014) classify four categories of FU in WM, namely: i) unitary FU defined by measurement unit (e.g., 1 Mg of waste), ii) generation-based FU indicated by waste generation in a specific region within a specific period, iii) input-based FU denoted by waste quantity entering a treatment facility, and iv) output-based FU described by the by-product of waste treatment, such as the amount of energy recovered. Most LCA studies employ unitary FU, although combinations of more than one FU can be found. It was as shown by a study employing two FU namely 1 kg waste and the output based on 1 MJ heat produced (Lausselet et al., 2016).

System boundaries must be specified regarding what is included and excluded in the study. The delimitation of the studied system reassures that all relevant inputs, outputs, and processes are considered. The boundaries are formulated based on the goal of the study, which cover raw material extraction up to EoL management (cradle-to-grave), raw material extraction up to the manufacturing stage (cradle-to-gate) or only the manufacturing stage (gate-to-gate) (Cao, 2017). When the studied system focuses specifically on WM, the zero-burden principle is applied. This means that the upstream process is excluded, and the assessment starts at the point where the waste is generated. This principle assumes that the same type of waste will be generated regardless of its EoL management. However, if the comparison between WM systems involves a waste minimization strategy, the upstream process—such as production and use—should be included (Björklund et al., 2010).

In addition to providing services in treating waste, WM also offers other functions, such as generating energy or material from WtE or recycling. These can be seen as by-products in the WM; thus, the total impacts from treating waste should be fairly assigned between the treatment and the by-products generated. ISO (2006b) recommends using system expansion instead of allocation to handle by-products. This means that the boundaries are being expanded to consider the by-products as alternatives to other products within the global market. Thus, the environmental benefit can be credited to the system due to the by-products (e.g., recycled plastic) that replace virgin material (e.g., virgin plastic) because its production can be avoided.

This study focused on the WM options in which various LCA goals differed among the papers. It focused on the different stages of WM, employing real cases in Finland. Figure 7 shows a general overview of WM systems and the boundaries applied in each paper.

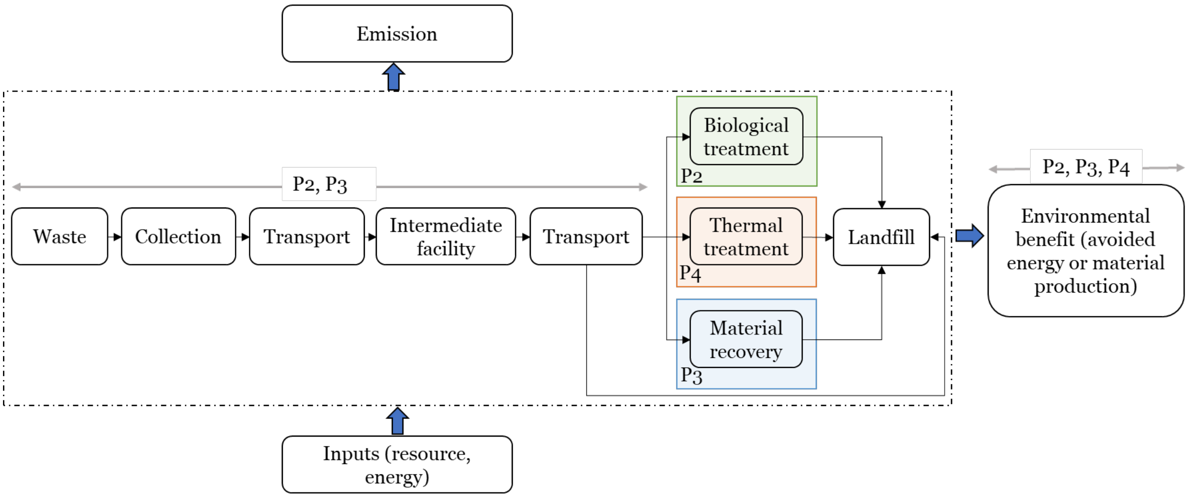

Figure 7. General overview of WM stages and boundaries of the study

Papers II and III present a change in waste management options to shift toward more circular practices. They assessed the economic and environmental impacts of source-separated biowaste to be treated in AD and recycling APW. The assessment was compared with the baseline situation where the biowaste was not collected separately and treated in WtE, and APW was landfilled. Paper IV depicts a condition where a more circular strategy was implemented without changing the waste treatment option. Thus, the optimization problem was employed in the WtE plant to investigate whether improvement could be achieved. Collection, transportation, and treatment were included in Papers II and III, whereas Paper IV included the treatment stage. Table 2 displays a summary of the goal and scope of each paper.

Table 2. Summary of goals and scopes of the papers

Paper	Objective	Waste type	Boundaries or scope	Handling by-products
I	Conceptualizing CE within product life cycle context and its practical implementation	N.A.	• Relationship of CE and WM • CE definition • Practical implementation of CE throughout different stages of product life cycle	N.A.
II	Assessing economic and environmental damage costs of the collection, transportation, and treatment of source-separated biowaste	Biowaste	• Collection of source-separated biowaste • Transportation of source-separated biowaste • Treatment of biowaste (AD)	System expansion for energy recovery and biosolids application

Paper	Objective	Waste type	Boundaries or scope	Handling by-products
III	Assessing environmental and economic consequences of APW collection and recycling	Agricultural plastic film (APW)	• Collection of APW • Transportation of APW • Mechanical recycling of APW • System expansion (avoided virgin plastic)	System expansion for recycled plastic
IV	Assessing environmental, economic, and technical impacts of optimized incineration when diversion strategy is not possible	Mixed waste	• WtE treatment for mixed waste • System expansion (avoided energy production)	System expansion for energy recovery

Crucial assumptions were applied for material and energy production avoided in the system expansion. The system expansion method involves identifying the products being substituted by the goods produced through waste treatment. The environmental impacts of the substituted products are quantified, and the avoidance of these impacts due to substitution is credited to the waste treatment process (Brander & Wylie, 2011). For biosolids application, approaches drawn from the existing literature were used to estimate the nutrient content of the biosolids and its replacement rate for the artificial fertilizer. Meanwhile, assessing the benefit obtained from recycled plastic was not as simple. The most common ratio for the substitution factor of recycled material is 1:1, which is not a fair assumption because quality decrease is often an issue with recycled material (Laurent et al., 2014). Therefore, the substitution factor of 54.5% was estimated and obtained based on the acceptance of secondary material using the market price of recycled material and virgin material. The avoided impacts equal the multiplication result of the substitution factor of 54.5% and the impact of producing virgin plastic material.

2.2.2 Life cycle inventory

The second phase focuses on the development of the system model in accordance with the defined goal and scope. The system model in LCA refers to a flow model within certain system boundaries (Baumann & Tillman, 2004). It displays a flow of different stages of the product life cycle as well as inputs and outputs of energy and resource/material (see Figure 7). Thus, data collection associated with inputs and outputs of different processes within the system boundaries becomes the core of life cycle inventory (LCI). Two types of data—foreground and background data—

were required (Clift et al., 2000). The former indicates primary data that is specific to the investigated system or product (such as material input, energy consumption, water, chemical consumables, emission, and waste), whereas the latter refers to the data concerning the generic industrial economy that is non-specific but necessary and sufficient to support the studied system (Kuczenski et al., 2018).

The data used in this study were collected from various sources, depending on the cases, which adopted different goals and scopes (Papers II, III, and IV). Paper I did not apply an LCA; rather, it employed narrative review to provide insight regarding CE and WM, including its practical application throughout the product life cycle stage. Paper II focused on the cost incurred when the stricter legislation concerning source-separated biowaste was applied. The system included waste generation, collection, transportation, and treatment. An optimization problem was employed to generate a collection and transportation route. The economic cost was estimated using CLCC, and the environmental damage cost was estimated using an LCA method called "life cycle impact assessment method based on endpoint" (LIME). Paper III applied LCA, ELCC, and an optimization problem. The assessment was conducted for APW (in this case, a film layer used to wrap hay bales). The collection and transportation routes were solved using an optimization problem, whereas the environmental and economic impacts were estimated using LCA and ELCC, respectively. Paper IV focused on the impacts of mixed waste treatment in the WtE plant when optimization was applied to the operation parameters. The main methods employed in Paper IV were optimization problem, LCA, and CLCC. Various resources were used to gather the data needed, as shown in Table 3.

Table 3. Sources used in the papers

Paper	Data sources
I	• A literature search in Scopus using specific keywords • Reference list of literature retrieved • A manual search of useful information found in the retrieved literature
II	• Data from the waste management company in Kauhajoki municipality • Expert judgment • Data from literature • Data from Ecoinvent 3.6 database
III	• Data from the plastic recycling company • Data from the waste collection company, covering 179 small and medium farms • Expert judgment • Data from literature • Data from Ecoinvent 3.6 database
IV	• Data from WtE equipment producer and operator company • Expert judgment • Data from literature • Data from Ecoinvent 3.6 database

2.2.3 Life cycle impact assessment

Life cycle impact assessment (LCIA) aims to define the impacts of environmental loads caused by the inputs and outputs quantified in the life cycle inventory (Baumann & Tillman, 2004). More relevant and articulate results can be obtained in this phase instead of using the quantity of resources consumed or emission from the LCI. ISO (2006b) describes the sub-phases during LCIA, categorized into mandatory and optional sub-phases. The mandatory sub-phases are:

- Definition of the impact categories: identifying and selecting impact categories.
- Classification: assigning inventory result parameters to their corresponding impact categories (e.g., CO_2 and CH_4 are assigned to climate change impact).
- Characterization: calculating the magnitude of the impact per category.

The optional sub-phases refer to:

- Normalization: transforming the characterization results and giving them greater context by dividing characterization results by the reference value.
- Grouping: sorting and ranking the results.
- Weighting: aggregating normalized results across impact categories by multiplying weighting factors to each category.
- Data quality analysis: conducting assessments to investigate the model reliability (e.g., sensitivity analysis).

There are two different levels of outcomes when calculating the results of an LCA: midpoint indicators and endpoint indicators. These are derived from the difference in the cause–effect chain applied to estimate the impact. The midpoint impacts lie somewhere within the cause–impact corridor, whereas endpoint impacts are at the end of pathways reflecting damage in the area of human health, ecosystem quality, and resource scarcity (Huijbregts, 2016). The midpoint impacts result in lower uncertainty and have a stronger relationship with the elementary flows, whereas the endpoint is easier to communicate although it has higher uncertainty (Hauschild & Huijbregts, 2015).

Presently, there are various ready-made LCIA methods that do not require practitioners to delve rigorously into each different impact category procedure.

Some methods provide only midpoint results (e.g., CML), and others provide both midpoints and endpoints (e.g., ReCiPe). Moreover, LCA software has become more common, making the LCIA phase less cumbersome. Table 4 illustrates the LCIA methods in each relevant paper.

Table 4. Impact assessment methods used in the dissertation

Paper	LCIA methodology	Assessment type	Impact level	Number of impacts assessed	Normalization reference
I	N.A.	N.A.	N.A.	N.A.	N.A.
II	LIME	Economic	Endpoint single score (environmental damage cost)	LIME is based on emission inventory (4 primary inventories)	N.A.*
III	ReCiPe 2016	Environmental, economic	Midpoint	6 (midpoint)	Not applied
IV	ReCiPe 2016	Environmental, economic	Midpoint, endpoint, endpoint single score	18 (midpoint), 22 (endpoint)	World

*LIME applies variables based on different countries; thus, the value for Finland is used.

2.2.4 Interpretation

The final step in conducting LCA is the assessment of results from LCI and LCIA against the goal and scope to produce conclusions and recommendations (ISO, 2006b). The identification of substantial issues and evaluation to determine confidence in the outcomes are carried out at this stage. Interpretations can be presented based on the character of results or analytical purposes (Baumann & Tillman, 2004). The character of results present the outcomes following the LCA phase, such as inventory, characterization, and weighting results. The analytical purposes can be achieved by presenting the results based on the paper's objectives, such as contribution, hotspot, scenario, uncertainty, or sensitivity analysis. LCA is an iterative process. Based on the outcomes of the interpretation, going back to reformulating the goal and scope, as well as refining the model at LCI and LCIA phases, may be necessary.

The papers presented the LCIA results as the contribution, hotspot, scenario, and sensitivity analysis. Conclusions, recommendations, and implications were then elaborated.

2.3 Life cycle costing and optimization problem

Other tools, such as LCC and optimization problems, were employed to obtain more comprehensive results. Paper II focused on the economic assessment of the economic and environmental damage costs obtained using the CLCC and LIME methods. Papers III and IV employed ELCC and CLCC. Furthermore, the optimization problem was also used in Papers II, III, and IV to acquire the best possible solution for the biowaste collection route, APW collection route, and improvement of WtE operation, respectively.

In general, costs can be distinguished into "internal" and "external" (Hunkeler et al., 2008). Internal costs refer to monetary costs incurred inside and outside the system, whereas external costs are incurred outside the economic system without direct monetary value in the market. The cost models comprise budget, transfer, and externality costs (Martinez-Sanchez et al., 2015). Budget costs are costs paid by the stakeholders involved (e.g., costs incurred by waste collectors, including vehicle and labor costs). Transfer costs include taxes, subsidies, and fees, representing income distribution but not resource reallocation. Externality implies welfare effects due to activities that are not otherwise remunerated.

Different cost calculations were applied in Papers II, III, and IV. Paper III employed ELCC, where the boundaries and reference of its calculation follow the LCA. It also included budget and transfer costs. Meanwhile, Papers II and IV can be considered as partial CLCC and ELCC. Paper II did not include LCA, and its cost model for monetary calculation included budget cost alone. Meanwhile, Paper IV included LCA and its economic assessment following the same boundaries as FU; however, it merely considered its budget costs.

Optimization problems are widely used in many sectors to find the most desirable solutions to a variety of challenges. When multiple criteria are considered for attaining an optimized solution, it is known as “multi-objective optimization” (MOO) (Sharma et al., 2012). The objectives may conflict with one another where there are many possible optimal solutions, known as “Pareto-optimal solutions” (Deb, 2001). The optimization problem for collection and transportation was conducted using OpenDoorLogistics (*Open Door Logistics*, 2014), which utilizes a real road network to generate more reliable and accurate results. Meanwhile, an Excel-based MOO program was used in Paper IV to optimize the WtE operation (Sharma et al., 2012; Wong et al., 2016).

3 RESULTS AND DISCUSSIONS

3.1 Circular economy and waste management

CE is a relatively new concept built on other preexisting concepts. Paper I explored CE building blocks within the context of the product life cycle. These building blocks—which concern the resource flow throughout the product life cycle—are industrial ecology (IE), waste management (WM), bioeconomy (BE), product service system (PSS), green supply chain management (GSCM), and cradle-to-cradle (C2C). Paper I explains the concepts above, which are briefly summarized as follows:

- IE can be defined as an economic model connecting to its surrounding system and should be designed to make it harmonious with the natural ecosystem. IE advocates networking among industries to advance the exchange of waste and by-products.

- WM comprises activities to control waste, aiming to protect human health, preserve the environment, and conserve resources. It is guided by the hierarchy that provides priority order in selecting waste management, while deviating from the order through identification of the most suitable options is also permitted.

- BE prioritizes renewable resource utilization, including processing waste into value-added products. The focus is not only on replacing fossil fuels but also on tapping into other sectors, such as pharmaceuticals, food, and feed.

- PSS aims to fulfill customers' needs by combining physical products and intangible services. It does not focus on the product's ownership; instead, it concentrates on delivering the services.

- GSCM is when environmental thinking becomes an integral part of supply chain management (SCM). It also deals with EoL products, whereas regular SCM focuses on the forward logistics until the consumers receive the products.

- C2C aims to decouple economic and environmental conflict, and it is seen as the closest to CE. C2C differentiates biological and technical systems; the former aim to return all the materials into the environment, and the latter intend to keep recirculating material within the system.

Various works have been published on the definition or conceptualization of CE since it became a trendy topic. Many of them equated CE with EoL management. Analysis of 114 CE definitions showed that almost 80% of them defined CE as recycling (Kirchherr et al., 2017). In Paper I, it was argued that CE is more than EoL management, instead encompassing the value chain through which resources or materials flow. As a notion that is built by pre-existing discourses, CE cannot be seen as a merely recycled concept. Its breadth and depth due to the diversity of the building blocks make CE an umbrella concept, as emphasized previously by Blomsma and Brennan (2017). CE can lace connections between its building blocks and construct new paradigms for how we focus on the value retention of products or materials throughout value chain. Although the product life cycle is seen as a whole system (from raw material extraction to EoL management), the adoption models of CE can occur at any stage in the value chain. These adoption models have become one of Paper I's main contributions that contextualized CE within a certain level of value chain to help interested actors or stakeholders navigate their approach toward circularity.

As one of the building blocks of CE, WM is primarily responsible for the EoL stage of the product life cycle, although it also plays a role at other stages (e.g., the manufacturing stage also generates waste). Each CE practice, including those at the EoL stage, can be characterized as resource conservation, narrowing the loop (resource efficiency), slowing the loop (resource prolongation), or closing the loop (resource recirculation). CE creates a paradigm shift in WM. It once aimed to protect human health and the environment; however, it has since become a component of resource management through value retention.

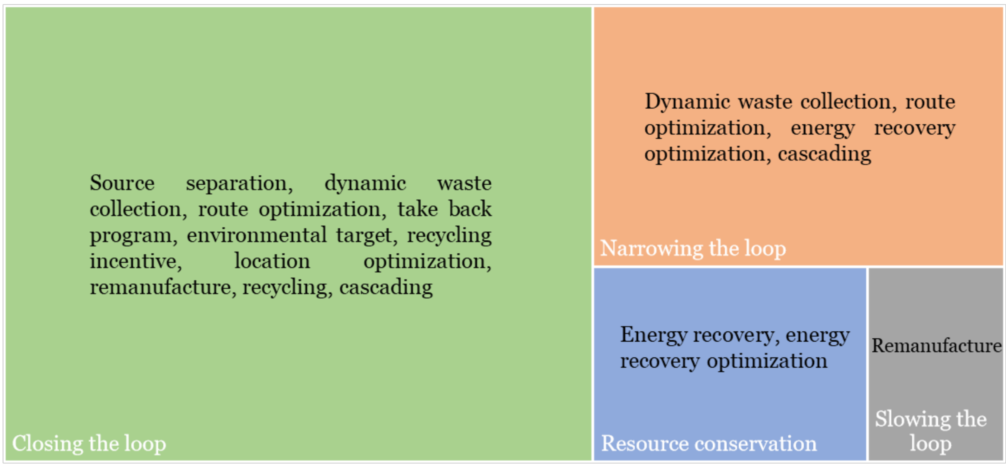

Figure 8. CE practices at the EoL stage

Figure 8 shows how WM is translated into different CE practices classified by circular characteristics at the EoL stage. The bigger the square, the more CE practices fit those characteristics. If these characteristics were deemed equally suitable, one practice could be assigned to more than one characteristic. Most (about 60%) of the practices could be classified as closing the loop, while slowing the loop was the lowest proportion (about 12%) of the practices. This demonstrates that WM primarily retains the value of products through practices that support material recirculation back into the supply chain. Examining CE practices may help expedite CE implementation because the related stakeholders could identify which practices are attainable and possibly combine some of them. In this dissertation, a few practices were investigated further; these are categorized as route optimization, recycling, energy recovery, and energy recovery optimization.

This work began by assessing CE on the conceptual level before narrowing to solve a real case focused on the WM, which is known as the building blocks of CE. CE is often equated with WM, and this association is reasonable considering the global state of WM system and the need to conserve virgin material. However, CE is about value retention; therefore, the following premise is formulated:

Premise 1. CE is beyond EoL management because attempts to retain material or resource can be applied at any point of value chain; thus, different stakeholders throughout the value chain can transition toward circularity by implementing various circular adoption models.

3.2 Environmental and economic impacts of different waste management options

Various other circular waste management options were assessed in both economic and environmental terms. With a combination of economic and environmental assessments, a more comprehensive picture can be obtained to make informed decisions. These cases include source-separated biowaste treated with AD, recycling of APW, and mixed waste treated in waste-to-energy plant. In each paper, the assessments were conducted, and some forms of comparison (not a scenario analysis) were used to contextualize the study. This dissertation found that moving toward more circular practices showed an improvement reported in Papers II, III, and IV. Some caveats may exist, such as the trade-off between economic, environmental, and technological aspects; therefore, prioritization given by decision-makers will be the determinant. Table 5 shows the summary of the main findings of each paper.

Table 5. Summary of main findings

Paper	The impacts of the more circular approach			The impacts of the alternative system			FU
	Environment	Economics	Technology	Environment	Economics	Technology	
II	€ 3.70	€ 146	N.A.	€ 5.13	€ 160	N.A.	Mg waste treated
III	-159.6 kg CO_2-eq (Case 1) -217.6 kg CO_2-eq (Case 2)	€-164 (Case 1) -129 € (Case 2)	N.A.	114 kg CO_2 -eq	N.A.	N.A.	Mg recycled material
IV	-1.4E+08 Pt*	€ 67.80*	18.45%*	-1.2E+08 Pt	€ 75.63	16.30%	Mg waste treated

*These results show the maximized possible benefit of the system; however, it cannot be obtained altogether since objectives are conflicting

In Paper II, the focus was on biowaste, and multiple sub-cases were investigated. Because the emphasis was on the impact of shifting to source-separated collection of biowaste, a fair comparison can be obtained only when the old system (where the biowaste was collected together with other waste as mixed waste) was also assessed. The sub-cases were classified into i) biowaste in the mixed waste under the old legislation (MW-OL), ii) source-separated biowaste under the new legislation (B-NL), and iii) mixed waste without biowaste under new legislation (MW-NL). In Table 5, the circular approach was the sum of B-NL and MW-NL, whereas the alternative system showed results from MW-OL. The assessment of environmental damage cost and the economic cost was conducted in the collection and transportation stage, as well as the treatment phase. The study uses a real case from Kauhajoki municipality where new legislation will create a new cluster of residences that requires a collection system.

Overall results indicated that diverting biowaste separately from mixed waste was economically and environmentally sound. When the results were observed in detail, the collection and transportation of the circular approach showed slightly higher costs. The monetary cost of collection and transportation under circular and alternative approaches were 81.1 €/Mg and 80.7 €/Mg, respectively; meanwhile, the environmental damage costs were 0.24 €/Mg and 0.23 €/Mg. There were significant differences during the treatment phase. The alternative system incinerates all mixed waste. In a more circular practice, the biowaste is treated in AD and the remaining mixed waste goes into incineration. The monetary costs of circular and alternative systems were 64.8€/Mg and 79 €/Mg, respectively, whereas the environmental damage costs were 3.5 €/Mg and 4.9 €/Mg. The case study in Paper II demonstrated that switching to a more source-separated waste was feasible.

The main aim of Paper II was to develop a calculation model for monetary cost and environmental damage cost using case study to implement the applicability of the models. However, these results cannot be generalized to other systems or contexts. Instead of applying the conclusion of this study to other contexts or countries, interested actors need to carry out the calculation, although a similar trend may be found. For instance, the collection and transportation stages contribute significantly to the WM system (e.g., up to 70%). Study in Paper II also emphasized the importance of route optimization. A more realistic approach using a road network rather than the closest distance of a straight line between two points is deemed essential. Differentiating road types can generate more representative results since speed limits are different, affecting travel time. The importance of doing own assessment also comes from the knowledge that source-separated waste will only benefit if the proper infrastructures are available (Daskal et al., 2018). The case study in Paper II also showed that most of the collection points were concentrated instead of scattered. Thus, different outcomes can be expected if the analysis is done in areas of sprawl/low density.

Moving toward circularity through waste separation from the source is deemed expensive because a new collection scheme will be required. This assessment may be accurate because the results show that circular collection and transportation cost more than the alternative case (i.e., no separation). However, the complete picture obtained from including treatment stage shows that the environmental and economic impacts were in favor of the circular case. This implies that a smaller system can ease the assessment and reduce uncertainties, although it provided an incomplete picture of the whole WM system. That deduction leads to the following premise:

Premise 2. The boundaries of the assessment can affect the final conclusions and strategy recommendation.

Paper III deals with untapped potential from APW, particularly wrap plastic used in covering hay bales for livestock. The ELCC covered the collection, transport, and recycling process, including the avoided impact of substituting virgin material with recycled material. Two cases were assessed in the circular practices, in which the collection frequencies were assumed to be once a year and twice a year, respectively. These two cases were studied to aid the WM company in implementing APW collection. Six impacts deemed most significant in plastic recycling were assessed, focusing more on climate change and economic impacts (Table 5). Negative values indicate benefits; thus, the more negative the results, the more beneficial they are.

There were trade-offs between case 1 (once-a-year collection) and case 2 (twice-a-year collection), as the benefit from climate change was higher in case 2, but the economic benefits were superior in case 1. With a yearly collection, it was assumed that more contaminants were mixed into the APW, thereby causing greater loss during sorting, with more waste ending up treated in an incinerator. In case 2, on the other hand, where the collection was more frequent, the contaminants accumulated were deemed less, and more plastic was recycled. Papers III and IV, as well as other studies (e.g., Beylot et al., 2018; Faraca et al., 2019) showed that incinerating plastic had significant impacts on climate change. In contrast, less frequent collection could yield cost savings; thus, the economic benefits were greater in case 1. In both cases, it was found that the collection contributed only about 0.7–1.2% to the climate change impact. Meanwhile, collection accounted for more than 30% of the total costs. The costly nature of the waste collection stage was also confirmed by Paper II.

Contribution analysis was conducted on climate change and five other impacts: fossil resource scarcity (FS), human carcinogenic toxicity (HT-C), human non-carcinogenic toxicity (HT-NC), terrestrial acidification (TA), and water consumption (WC). In both cases 1 and 2, incineration contributed to the greatest environmental impacts on climate change (HT-NC, HT-C), whereas reprocessing (recycling process) was the greatest contributor to WC, TA, and FS. Across all six impact categories, plastic substitution provided the greatest benefits due to the avoided production of virgin material. For the economic impact, plastic substitution offered the highest benefit in cases 1 and 2, whereas the costliest component was reprocessing, followed by collection. A comparison was made between the circular approach and the alternative landfill system. The environmental impacts of plastic landfilling were obtained from Ecoinvent 3.6 with adjustment, equaling 1 Mg of recycled product. The climate change impact of landfilling 1 Mg of plastic is more than 100 kg of CO_2-eq. For the other impacts, recycling provided consistent benefits compared with landfilling. The benefits of recycling were about 2–2.3 times better than landfill for TA, FS, HT-C, and WC in both cases. The greatest benefit was HT-NC, where recycling was 17–19 times better than landfilling in cases 1 and 2.

The significant economic and environmental benefits of recycling plastic were obtained by substituting virgin plastic with the secondary product, regardless of the case in this dissertation. Although the business potential is promising, this situation poses some challenges regarding the fulfillment of standard quality of secondary material and the market acceptance as substitute for virgin plastic. In general, the incorporation of secondary plastic into new products is still relatively low. Therefore, the following premises are formulated:

Premise 3. Collection strategy is driving both the quality of secondary material and total circularity cost.

Premise 4. Policy instruments are needed to gatekeep the quality of secondary material and to support its reintroduction back into the economic system.

Paper IV focused on optimizing operation parameters of WtE to obtain better outcomes. WtE, specifically incineration, is commonly utilized even in countries with established source-separated systems. Paper IV studied the environmental impacts of the WtE plant treating mixed waste and optimized the plant by changing its operating parameter to identify any improvement. The three focal points of the study were the environment, economics, and technology. Different indicators expressed these three aspects: environmental impacts (single score unit), waste treatment cost, and plant thermal efficiency.

For the environmental impact, multiple levels of impacts were assessed, such as midpoint, normalized endpoint, and single score. These approaches were taken to ensure that the results of the alternative system can be compared with other studies because midpoint impacts, especially for climate change impacts, are commonly used. In the alternative system without optimization, the environmental impact, costs, and plant efficiency per FU were -1.2E+08 Pt, € 75.63 and 16.30%, respectively. The midpoint climate change impact from direct emission was 510 kg CO_2-eq, comparable with previous studies (e.g., Beylot, Muller, et al., 2018; Lausselet et al., 2016). For the economic aspect, a similar cost to incinerate waste was shown by Martinez-Sanchez et al. (2016), and the efficiency was also typical for WtE with electricity recovery, which usually shows values of 14–28% (Beylot, Muller, et al., 2018; Martinez-Sanchez et al., 2016).

The baseline operation was optimized by changing six operating parameters, including temperature, pressure, and isentropic efficiency. The optimization problem showed that improving the system was possible where the maximum improvements for each objective were around 13.4%, 10.3%, and 14.8% for thermal, economic, and environmental, respectively. These improvements cannot be achieved in the same optimal solution because the objectives conflict with one another. Related actors must arrange the order of importance of these objectives. A variety of optimal solutions provided by the combination of six operating parameters may yield a consistent improvement in all objectives, or a greater improvement in one objective at the cost of less advancement in other objectives.

Papers II and III discussed circular approaches through changing waste treatment methods. However, this strategy is not always possible, especially within existing economic and infrastructure constraints. Thus, the following premise is advanced, based on findings in Paper IV:

Premise 5. It is possible to be more circular by optimizing the existing waste treatment system when changing it is not an option.

3.3 Assumptions, sensitivity analysis, and scenario analysis

Implementing LCA requires various data as inputs, including assumptions when required data is unavailable, or when simplification is needed. The selection of FU, boundaries, parameters, and methods affect the outcomes of the assessment. This dissertation employed the average condition, common methods, and FU. For example, the investigation in Paper IV applied FU of 1 Mg of incoming waste. The FU of the treated waste quantity is one of the most common FUs in waste management (Laurent et al., 2014). Selecting a common FU or method facilitates equivalent comparison with previous results, which can help to evaluate whether the results are relevant within a particular context and how useful they are in addressing the knowledge gap. Moreover, making comparisons among studies can help direct future research.

A similar phenomenon is observed in the LCIA methods. Various methods available can be selected based on the study's need and primary aims, such as the use of cumulative energy demand (CED) methods when the focus is on the impacts concerning energy resource depletion (Cascone et al., 2020). Papers II, III, and IV used the same combination of data inputs, namely primary data, expert judgment, literature, and the Ecoinvent 3.6 database (Table 3). The full impact assessments were evaluated in Papers III and IV using the ReCiPe method. ReCiPe is one of the most updated assessment methods and quite versatile because the impacts can be calculated at different levels. These levels are midpoint impacts, endpoint impacts, and single score results (RIVM, 2016). Thus, the impacts level can be adjusted based on the need and aim of the study. Paper IV assessed the impacts on different levels, including midpoint impacts, endpoint impacts, and the single score. Midpoint impacts are commonly used (e.g., Beylot et al., 2018; Faraca et al., 2019; Lausselet et al., 2016) to make a comparison. The endpoint impacts with normalization can contextualize all impacts among each other (Paper IV), whereas a single score was deemed useful in simplifying the results generated in multi-objective optimization (MOO) (Paper IV). Instead of presenting the optimized results as separate impact categories (18 and 22 for midpoint and endpoint impacts, respectively), single-score output can streamline the result and render it more articulate.

Another significant assumption was used when calculating the environmental benefit from material produced during plastic recycling. In the waste treatment process, environmental benefits are obtained from products or energy, such as recycled material, biosolids, or energy. Ratio 1:1 is still commonly used to reflect the substitution of virgin material for recycled material (Laurent et al., 2014). It implies that secondary material has the same quality and acceptance as virgin material and could lead to overestimation of the environmental benefit (Gala et al., 2015). Paper III applied a market substitution factor that reflects the acceptance of the recycled material. Even after a careful attempt was made to avoid overestimating the benefits, different studies applied different values of substitution factors, ranging from 50–95% (Faraca et al., 2019; Gu et al., 2017; Rigamonti et al., 2014). Thus, sensitivity analysis becomes important in examining the importance of this parameter relative to the LCA model.

These input parameters, assumptions, formulas, and methodological selection generated uncertainties. The studies were complemented with sensitivity analysis to address the issue. It aims to identify how the outcomes vary as a result of changing the input values (Bisinella et al., 2016). The analysis was conducted by increasing each input parameter by a certain percentage and holding the others the same as the baseline values. The analysis was conducted to gain understanding of ranking parameters corresponding to their sensitivity within a particular LCA or LCC model context. This information conveys insights and can be utilized by interested stakeholders. High-ranking parameters show their relative importance in affecting the outcomes. Thus, reassuring good quality of data regarding sensitive parameters is key. Moreover, when the LCA outcomes need to be adjusted, actors can target a few sensitive parameters. An example can be found in Paper IV, where optimized solutions could be attained by modifying operating parameters in the WtE plants. Six parameters were adjusted to generate optimized solutions. Sensitivity analysis showed that the most sensitive parameter for plant efficiency, environmental impact, and economic impact was steam temperature entering and leaving the high-pressure turbine. Therefore, focusing on these two parameters can simplify the task instead of tuning all six parameters. A summary of the most sensitive parameters is shown in Table 6.

Table 6. The most sensitive parameters of different case studies

Paper	Sub-case	Most sensitive parameter	
		Environmental assessment	Economic assessment
Paper II	Source-separated biowaste	Fuel consumption rate (for the transportation), methane potential in waste (for the treatment)*	Waste quantity (for transportation), labor cost (for treatment)
	Mixed waste (without the biowaste)	Fuel consumption rate (for the transportation), the fossil carbon content in waste (for the treatment)	Waste quantity (for transportation), CAPEX (for treatment)
Paper III	Case 1 and case 2	Market substitution factor	Market substitution factor
Paper IV		The temperature of the steam coming into the high-pressure turbine**	The temperature of the steam leaving the high-pressure turbine

*The environmental assessment in Paper II refers to the environmental damage cost assessment
**The same parameter was also the most sensitive affecting the efficiency

The model robustness was then examined against the background system using scenario analysis; scenario analysis also answers questions regarding multiple alternatives available that should be compared. Furthermore, it can assist in directing future research or making a decision based on the possible pattern produced. Table 7 summarizes the scenarios and their outcomes in Papers III and IV.

Table 7. Scenario analysis in Papers III and IV

Paper	Scenario	Outcomes
III	1. Changing diesel to liquid natural gas (LNG) for the plastic waste collection 2. Changing the marginal energy source to natural gas	1. The results showed relatively small environmental and economic benefits 2. The results varied—some worsening, some improving—depending on the impact categories
IV	1. Changing waste composition by decreasing and increasing the organic and plastic waste composition, respectively 2. Varying the marginal energy source into a more sustainable source (mix of wood, wind, nuclear) and fossil source (mix of nuclear, natural gas, and hard coal)	1. The cost of treating waste increased, the energy produced increased, and the single-score impact decreased (more overall environmental benefit) 2. The impact score increased for more sustainable source increased (less benefit than baseline) and decreased for fossil sources (more benefit than baseline) to LNG provided relatively small environmental and economic benefits

The scenarios in Paper III included changing the fuel type during the collection and switching the marginal energy, for both the electricity and heat source, into natural gas. Paper IV implemented different waste compositions and marginal energy sources for its scenarios. The analysis answered the what-if and offered evidence for more informed decision-making. In Paper III, there were small benefits in environmental and economic performance from LNG compared with diesel. This information can help direct the decision of whether to change fuel based on the interests of affected stakeholders, especially if the vehicle should be modified to run on LNG. Modifying incoming waste in Paper IV presented different results for environmental, energy, and economic aspects. The environmental aspect demonstrated improvement, as shown by the decrease in single-score value. Nevertheless, it is important to obtain more information regarding plastic waste treatment and to contextualize the study. The single score in Paper IV covered all environmental impacts, whereas interested stakeholders may wish to prioritize one impact above the others—specifically, when climate change impact is the highest priority, as Paper III, as well as Hou et al. (2018) and Wäger & Hischier (2015), deduced that recycling was better than incineration.

Papers III and IV included marginal energy analysis. When incineration is accompanied by energy recovery, it is inherently assumed that environmental benefits will be obtained. However, the benefits are relative to the source of marginal energy being substituted. For example, in the case of climate change impact, both Papers III and IV showed that the more sustainable the source of marginal energy, the lower the benefits obtained. The opposite applied when the benefits from fossil energy sources were calculated. The system expansion is performed by subtracting the impacts from the alternative system. Therefore, with a higher climate change impact generated by marginal fossil energy, more value is subtracted from the total impact caused by incineration. This result indicates that we should be cautious in generalizing one result to another, because the source of marginal energy in different contexts or countries tends to vary.

The scenario analysis also provided understanding regarding outsourcing certain processes in different countries because changing marginal energy showed different impacts caused by different energy sources. It could inform stakeholders such as government or private businesses when they plan to recycle waste in other countries or import goods from overseas, because the impact on the manufacturing level can be different, even if the process may be the same. Countries have different energy mix sources, and certain businesses may have greener contracts for their electricity consumption. Comparing several contractors through scenario analysis will produce a more comprehensive understanding.

3.4 Implications for stakeholders

Paper I illustrated the many stakeholders involve in the value chain who have different interests and do not always act in accordance with the circular principle. As mentioned in the introduction, consumers may refuse a certain business model when it does not serve their interest; therefore, business should be aware of all of the circular options that they may implement. The paper provides compact database of circular adoption model in the value chain so that stakeholders with diverse interests and influences can consider alternatives.

In the cases, the broad implication for stakeholders of a more circular practice was found in the case of source-separated biowaste in Paper II and APW recycling in Paper III. Some new players can be found, or the old stakeholders may remain involved but will have new interest when the waste is separated from the source and the treatment is diverted from one method to another. Paper III showed an even larger implication because plastics are versatile materials, and the recycled plastic market is broader beyond the national level. Furthermore, the residuals during the recycling process would be treated in the WtE plant, which shows that the recycling process—at first seemingly isolated—is in fact connected to multiple systems. Table 8 shows the summary of main stakeholders and their interest in each case. These main stakeholders are those who are seen to be involved directly in the system. Extending the boundaries may result in the inclusion of more stakeholders who have less interest and influence in the system. For example, in Papers II and III, the WtE is affected by the treatment option of the respective wastes. When biowaste is diverted to AD, the form of energy produced is changed, and WtE may need to find additional input to maintain its working capacity. In Paper III, the WtE will receive additional waste from the recycling plants. A similar condition can be concluded from Paper IV, where the WtE plant was optimized and could recover more energy. The situation should be assessed against a larger energy system to examine whether the energy recovered from WtE could provide environmental benefits, by analyzing the marginal energy in the study area.

Table 8. Overview of the main stakeholders involved and their interests

Paper II	Paper III	Paper IV
• Consumers: waste disposal • Waste collection company: collect and transport waste efficiently • AD plant: treat waste and generate best products • Buyers of biosolids and energy: products that meet standard and function • Fertilizer producer: maintains market share • Regulator: ensures that WM follows legislation	• Consumers: waste disposal • Waste collection company: collect and transport waste efficiently • Recycling plant: treat waste and generate best products • Buyers of the secondary material: products that meet standard and function • Virgin plastics producer: maintain market share • Regulator: ensures that WM follows legislation	• WtE plant: treat waste and generate best products • Buyers of the energy: products that meet standard and function • Regulator: ensures that WM follows legislation

More circular practices will also demand improved collaboration along the supply chain from cradle to grave, or even from cradle to cradle. As one of the stakeholders, the government has a high interest in directing the course of circular waste management, as well as strong influence to do so. Government should also take an active role beyond setting up new reuse or recycling targets by assessing a new legislation before it takes effect to verify its feasibility. Applying LCA and LCC to assess issues from economic and environmental perspectives may form the basis for an improved decision-making. Additionally, decision-makers should also be aware of the latest adoption models of CE and should assess which models merit more support from the legislation. An example can be taken from the recycling plastic ecosystem, where the recyclers must maintain the output quality and deal with market variation where the price of virgin material can be lower. The relevant government authority may require industry to incorporate secondary material in their products. Collaboration across stakeholders is also essential, not only to ensure that the circular practice can run smoothly, but also to ensure that long-term circular benefits can be maintained through data sharing. All stakeholders across the supply chain must build a collaborative scheme supporting a circular agenda, such as tracing and tracking material and its flow until the EoL stage. Implementing a scheme to share the data could also help when assessment is required (e.g., LCA or LCC), because the data is already available.

4 CONCLUSIONS AND LIMITATIONS

CE is often considered to be the sole solution to environmental problem—a panacea to attain a better future. It is not. Going fully circular will not be possible today or in foreseeable future. However, moving toward more circular practice is nonetheless beneficial, and adopting a semi-circular system may be the best possible option for now. This dissertation discussed CE in a broad sense—from the product life cycle perspective—to paint a picture of the breadth of the concept and to provide understanding that circular approaches can be implemented by different actors at different product stages. The dissertation focused on the final stage of the product, EoL, considering the general state of our waste management system as well as its relationship with resource scarcity issues. Because case-by-case analysis is important in assessing CE benefits, that is what the papers were all about.

The study examines the economic and environmental implications of transitioning toward more circular practices. It was shown that shifting toward a more circular system requires changes at the system level, which in turn requires the participation of multiple stakeholders. The dissertation, which focused on the circular practices at the EoL stages, showed that the shift toward circularity affected other stages in the product life cycle, such as raw material, design, production, and energy system. Moreover, the focus on the WM was found to be broader and intermeshed with other concepts, namely bioeconomy, C2C, and GSCM. The interlinkage and overlap between stages in the product life cycle and CE building blocks show the width and breadth of CE.

On a more specific level, the dissertation employed three cases that showed the economic and environmentally feasibility of implementing more circular practices at the EoL stage, although trade-offs or caveats could be found. Studies on separating biowaste from source showed convergent results in both economic and environmental damage costs. Meanwhile, the study of APW recycling showed trade-offs between the environmental and economic benefits of the two cases. It demonstrated that less frequent collection may save cost but might potentially accumulate contaminants, decreasing the waste's recyclability. In contrast, more frequent collection increased cost but potentially provided more environmental benefits because the condition of the waste collected would be less contaminated. Nevertheless, both cases offered environmental and economic benefits. A different approach was applied in the third case. Although it still employed a more circular practice, there was no shift in the WM option. Instead, it optimized the existing WtE system in light of the consideration that WM diversion is not always possible. The case indicated that increased benefits may be obtained in the thermal, environmental, and economic aspects. Each objective can be optimized maximally, although it would not be obtained from the same solution due to conflict among the objectives.

The trade-offs in plastic recycling and WtE cases illustrate that LCA, LCC, and optimization problems were the only tools to produce comprehensive knowledge. Those trade-offs also demonstrate the specificity of environmental and economic assessment conducted using LCA and LCC, which requires careful consideration when generalizing the study. Related stakeholders must prioritize different aspects or objectives to make the optimal decisions.

As an analytical tool, LCA remains the most suitable instrument to evaluate the shift toward circularity. It is a standardized tool, which means that a variety of organizations can adopt and implement it to produce evidence of the environmental claims regarding their products or services. Larger corporations may begin applying LCA to evaluate their performance and support their decision to pursue certain circular strategies, such as closing the loop or switching to greener suppliers. For smaller and medium organizations, it is also essential to apply LCA and assess their environmental impacts because these types of organization accounts for the majority of businesses worldwide.

Nevertheless, LCA implementation has its limitations, as does this study. LCA, LCC, and optimization problems apply a certain level of simplification, applying assumption, estimation, and secondary data. The primary data was used for the foreground; however, some unavailable data was supplemented by secondary data such as expert judgment, literature, and database information. On a methodological level, system expansion requires the inclusion of an additional system wherein some products or services are substituted by the products or services resulting from the system being studied. This creates uncertainties regarding the environmental benefits obtained from avoidance or system expansion. In the case of APW recycling, there was no single correct value for the substitution factor. One can employ a value based on literature or use a certain approach to generate the value. A similar situation applies to energy substitution, where the environmental benefits depend on the marginal energy sources. This issue becomes acutely relevant when there is an abrupt change in the energy landscape, such as the current disruption caused by Russia's invasion of Ukraine. Countries are scrambling to meet their energy demands after cutting themselves off from the Russian supply. This applies not only to the marginal energy being substituted but also to the average energy mix in different countries. This condition was not captured in the LCA because LCA takes an average value for the marginal energy use in the studied system.

Thus, transparency is always of the utmost importance in conducting LCA because it will allow other, repeated studies to generate a similar result, or a revised study that modifies the sources of the energy mix. The dissertation also shows the importance of conducting sensitivity analysis to deal with uncertainties. The most sensitive parameters may significantly affect the results, even with a small change in their values. It is important to ensure that the data input for the most sensitive parameters is as accurate as possible. In the case of APW, the substitution factor was the most sensitive parameter, yet it did not have a standardized value. Nevertheless, knowledge about its sensitivity could help create awareness in interpreting and using the model. When the substitution factor's value decreases or increases, the model's behavior can be anticipated.

This study has showed how to solve a real-world problem, and its results indicated the viability of going more circular. Due to the breadth of the CE concept, the chosen focus was on the EoL stage for specific types of waste within certain boundaries. Therefore, many avenues can be pursued for future studies. Further studies can look into the impacts of source-separated waste other than biowaste within different geographical areas. Such investigations can produce a fuller picture of the effect of waste type and demographic on environmental and economic aspects. Another possibility is studying the effect of introducing secondary material into the supply chain and the relationship between suppliers and manufacturing. The focus can be expanded beyond environmental and economic impacts—due to the importance of value retention—can be done by applying material flow analysis (MFA) to assess the flows and stocks of materials or substances. Thus, the information regarding how much material can be reintroduced into the economic system, lost, or end up in disposal can be accounted.

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Title page

Circular economy: From conceptualization to database tool of practical implementation

Bening Mayanti [a,*], Petri Helo [b]

[a] Vaasa Energy Business Innovation Centre, University of Vaasa, Fabriikki F455A, Yliopistonranta 10, 65200 Vaasa, Finland (ORCID 0000-0001-5073-7375). E-mail: bening.mayanti@uwasa.fi

[b] Networked Value Systems, Department of Production, University of Vaasa, P.O. Box 700, FI-65100 Vaasa, Finland (ORCID 0000-0002-0501-2727). E-mail: petri.helo@uwasa.fi

*Corresponding author:

Bening Mayanti

Abstract

The study provides an overview of circular economy (CE) theoretical building blocks, conceptualization, and practical implementation within the supply chain. A narrative review is employed to synthesize the knowledge and map research in the CE field. The review results in six major building blocks: waste management, industrial ecology, bioeconomy, cradle to cradle, green supply chain management, and product-service system. These building blocks are mapped to clarify their role and the possible overlap within the supply chain. This study also generates a database containing 43 circular strategies categorized based on different stages in the supply chain. Each strategy is marked by its contribution toward CE, such as resource conservation, narrowing the loop, slowing the loop, or closing the loop. The database serves as a guideline for actors to implement possible circular actions and for policy makers to formulate supportive legislation. This study contributes to the contention of CE and its implementation.

Keywords: circular economy, closed-loop, conceptualization, implementation, resource flow

1 Introduction

The conventional linear model of industrial development called 'take-make-dispose' is becoming irrelevant to our current situation, driving an urgent need to implement a new economic model, namely the circular economy (CE). This linear model entails resource losses throughout its product life cycle. It was reported that around 21 billion tonnes of materials consumed during the manufacturing process were not physically included in the products; moreover, only 40% of waste was reused, recycled or recovered [1]. Implementing CE can decouple economic growth from environmental problems and promote sustainable development. McKinsey [2] predicted a yearly economic gain from CE practice in Europe is about €0.6 trillion for primary resources and €1.2 trillion for non-resource and externality benefits by 2030. It was reported that implementing CE can generate around 50,000 new jobs and up to €12 billion in investment in the UK and Netherlands, making CE a high agenda item at the national and regional levels [3].

The first documented concept of CE was an issue about economic growth and environmental quality raised by Boulding in 1966 [4]. He proposed transitioning from a limitable planet to a closed sphere with a constant resource recirculation. The foundation laid by Boulding on the closed economic model called the "spaceman economy." It was defined as an economy '*in which the earth has become a single spaceship, without unlimited reservoirs of anything, either for extraction or for pollution, and in which, therefore, man must find his place in a cyclical ecological system which is capable of continuous reproduction of material form even though it cannot escape having inputs of energy'* [4]. The model also advocated reducing throughput, resulting in less production and consumption.

As an emerging concept, studies around CE are growing rapidly. Google trends show low interest in CE until 2013, and the search has been consistently increasing until now. On the scientific side, the search for the CE term on Scopus resulting more than 1,000 results since 2018, with the highest outcomes in 2021 resulting in more than 4,800 articles. A broad spectrum of multiple interpretations of CE has emerged. Geissdoerfer, Savaget, Bocken, & Hultink [5] implied that CE played an important role throughout a product life cycle, and it was deemed a condition for sustainability. Kirchherr, Reike, & Hekkert [6] reported that CE is about phasing out waste, so its main role is in the end-of-life (EoL) management through reducing, reusing, recycling, and recovering material. Whereas Smol, Kulczycka, & Avdiushchenko [7] emphasized the importance of innovative technologies to recover valuable material. These examples imply differences regarding CE's core principle, scope, and operationalization. CE could mean many things due to preexisting concepts that provide circular ideas. The situation can create confusion regarding the understanding of CE since those preexisting concepts may overlap with each other. Moreover, it was found that previous studies rarely discussed about the practical implementation of CE within different sectors. The lack of knowledge regarding practical implementation could be caused by a limited understanding of the CE concept and origin [8]. Hence, this article addresses these challenges concerning CE conceptualization and practical implementation. One objective is to contribute toward CE conceptualization based on its building block within the product life cycle. Another objective is highlighting and assembling various strategies to implement CE in different product life cycle stages. These two aims can be achieved by the following research questions (RQ): i. What are the main building blocks of CE? ii. What are the contributions of the CE building block to the product life cycle? iii. How can circularity be approached on a practical level?

This paper is organized as follows. Section 2 outlines the material and methods. Sections 3, 4 and 5 present the building blocks of CE, the conceptualization of CE and practical implementation, respectively. The conclusions are presented in Section 6.

2 Material and methods

A narrative review was used in this study. It is intended for subjects that have been conceptualized diversely and examined by multiple groups of researchers within different fields [9]. It provides general literature overviews, the conceptualization of new studies or reconceptualization of established research [10]. A narrative review can synthesize current knowledge, map a research field, and generate a timeline for a research topic [9]. The results of a narrative review can act as a summary of studies offering a relatively comprehensive overview of the knowledge in that area [11]. These characteristics of narrative review are suitable for reviewing CE, where the concept has existed for quite some time.

The methods section for narrative review is not obligatory; however, it can provide more clarity regarding the main message of the review [11]. The study started with a pre-literature search to help formulate the objectives and obtain a general understanding of what studies have been performed in the area. After the RQs were formulated, the search for literature through the Scopus database was conducted. Scopus was deemed sufficient since it has a wide coverage of journals, including natural science, social science, and interdisciplinary field, which is prominent for CE topics and has a robust exporting feature [12]. The terms used to search the literature were related to the objectives of this study and the "spaceman economy" proposed by Boulding regarding material recirculation in a system. These terms included, but were not limited to, "circular economy," "circular economy AND closed loop," "resource flow," "circular economy AND resource flow," "closed loop supply chain," "material flow," and "waste management."

The next step was article screening to select relevant articles. The screening process was conducted by applying inclusion criteria (language, type of article, subject area, number of citations), followed by abstract screening focusing on history, other pre-existing concepts, relevance to the product life cycle, and practical implementations. Using citation numbers as inclusion criteria has drawbacks since relevant articles can be excluded. Hence, a manual search was applied based on the relevant knowledge and references found in the articles. Next, articles that mainly discuss the concepts and practical implementations regarding Boulding [4] proposition about material recirculation and closed-loop system were gathered, assessed and synthesized. The results would be characterized as CE building blocks (Section 3), CE conceptualization (Section 4) and CE practice within the supply chain (Section 5). The division of these sections and their sub-sections were based upon the RQ posed in this study. The building blocks of CE in sub-section 3 was based on pre-existing concepts predated CE, which contribute to material recirculation/closed-loop system targeting one or more stage in the product life cycle (raw material sourcing until EoL management). Sections 4 and 5 answer RQ 2 and 3 regarding CE conceptualization and practical implementations.

3 CE and related concepts

The following section identifies and describes pre-existing concepts that built CE. This study formulated high-level concepts such as the CE building block that can cover one or more stages in the product supply chain and identify their shared features to conceptualize CE.

3.1 Waste management

Waste management is seen as activities related to waste control in order to protect the environment and human health, as well as conserve resources [13]. The approach to managing waste is guided by the waste management hierarchy that provides a commonly agreed list of desired activities, prioritising waste prevention in the first place [14]. It aims to identify the most suitable option in managing waste that will result in the most ecological environmental outcome. Waste hierarchy emerged because of the urgency to replace open dumping practices that have led to land scarcity, hygiene issues and toxicity from hazardous waste [15].

Later, the waste hierarchy evolved as the paradigm of waste shifted. Waste was once seen as an unwanted matter that needed proper disposal now is deemed as a resource. The hierarchical order shifted to support circularity, and now it consists of refuse, rethink, reduce, repair, refurbish, remanufacture, repurpose, recycle, and recover energy, with landfilling being avoided in CE [16]. The new hierarchy model incorporates the cascade principle, where economic value is considered in managing waste [17]. In this study, prioritization based on environmental and economic considerations in the waste hierarchy is seen as a principle that back the notion of transitioning towards the spaceman economy by avoiding waste disposal in the first place and maximizing its potential through reutilisation in the production process.

Although hierarchical order provides clarity to manage waste the implementation can be hampered by the unclear benefit or unsupportive policy. Unclear benefit can be shown by recycling, which should be prioritized before energy recovery and landfilling, that has its own consequences. Transportation from the collection point to the processing facility can be costly, and product disassembly may require an energy intensive process or use of toxic chemicals, resulting in higher overall environmental impacts than incineration [18,19]. From a policy perspective, a policy that fails to clarify some terms (e.g., what is considered waste or a by-product) can lead to a difficulty in applying a certain measure such as a failure to reuse certain materials directly in the industrial process because the legislation requires certain treatment before it goes back into the process [20]. If the benefit of following the waste hierarchy is uncertain, economic, and environmental impact assessment can be implemented through environmental life cycle costing (ELCC). The results will provide comprehensive information to assist decision-making [21].

3.2 Industrial ecology

Industrial ecology (IE) is seen as an economic model that is not isolated from other surrounding systems; it interacts with the biosphere and should be constructed to make it compatible with the natural ecosystem [22]. It focuses on a closed-loop system through the exchange and cascading of materials and energy covering the fields of industrial ecosystem, industrial metabolism (IM), industrial symbiosis (IS), and legislation to support its implementation [23]. It includes network optimization among industries to enhance the exchange of resources, energy, and capital generated from waste and by-products [24]. IE is also seen as an ecosystem with a certain distribution of materials, energy and information flow that relies on resources and services from the biosphere [22]. The term "industrial ecosystem" itself was conceived by Frosch & Gallopoulos [25]. They argued that industries needed to optimize material and energy use in conjunction with waste minimization and waste utilization as raw material for other processes. At the same time, they also admitted that an ideal industrial ecosystem is not attainable but that shifting toward this principle can reduce adverse environmental impacts. Its emergence was around the time that an alternative to the end-of-pipe approach was thought necessary. Nonetheless, companies that experimented with prevention strategies were limited because it was almost impossible to avoid by-products of certain activities, and they had limited resources to implement the strategies [22]. Broader and comprehensive integration of the end-of-pipe approach and waste prevention was how IE was intended.

Industrial ecology and industrial symbiosis are two terms that are commonly used interchangeably. It is important to distinguish these concepts, especially on the implementation level, where certain requirements and goals must be fulfilled and achieved. Li [23] emphasizes the difference by stating that IE encompasses a broader notion, with IS being one of its parts. IS discovers ways to create a network of knowledge to enable the physical exchange of materials, by-products, and energy within geographic proximity to support higher levels of closed-loop ecosystems [26]. Although until now it is not possible to completely phase out waste, this exchange will reduce the sum of waste generation since the linkage between industrial processes facilitates the uptake of waste generated by the certain industry as an input for others. The Eco-Industrial Park (EIP) is an example of IS implementation that exploits geographical proximity to achieve a nearly closed-loop ecosystem [27]. It is formed through a top-down approach from the government or a bottom-up initiative from industrial consortia. Mo Park et al. [26] explained that top-down approach started with the government supports in the form of regulation or financial to form an EIP, whereas a bottom-up initiative occurs organically because businesses just interact with each other on the basis of economic advantage.

3.3 Bioeconomy

The bioeconomy - also known as the biobased economy- is defined as economic activities that utilize renewable resources, including converting the resources and waste into value-added products [28]. Some scholars argue that economic growth is the priority in the bioeconomy, making it a challenge to promote social and environmental issues [29]. Bioeconomy encompasses wider activity beyond fossil fuel replacement in the use of biobased materials for food, feed, and pharmaceutical sectors derived from biomass resources, including multi-output production process and utilization through a cascading approach [30]. Biomass resources refer to renewable organic materials from animals and plants that can be classified into virgin biomass and biowaste [31]. The concept can be fostered through government support or industry initiative, where research and development are the cornerstones. Although transitioning into a bioeconomy can provide economic benefits, the process also poses some challenges. These challenges include competition in using certain biomass for different purposes, translating research into scalable processes, creating viable business models, and creating socially accepted products [29]. To achieve a sustainable bioeconomy, principles such as safeguarding, avoiding, and prioritizing should be implemented [32]. Safeguards call for production processes that will not exceed the regenerative rate of a renewable resource and the availability of finite resources, whereas avoiding refers to the prevention of the production of nonessential products and the loss of biobased material [32]. Innovation is a prerequisite to ensuring that biobased material can be utilized entirely, whereas biorefineries can help avoid loss through multi-output production. The last principle, prioritizing, is implemented to optimize the utilization of biobased resources and guide the production towards resolving the competition issue, where one resource can produce multiple products. Priority should be given to basic human needs, high-value products, and sectors with no sustainable alternatives [32,33].

Boulding [4] was aware of resource limitations and proposed a closed-loop system to solve the issue. The bioeconomy could contribute to dealing with the resource limitation issue through biorefineries and the cascading principle. The bioeconomy could offer a twofold benefit: it helps with resource limitation by utilizing renewable materials and eliminating environmental risk once the material is returned to nature. Nevertheless, the biobased economy still poses some limitations. The need for additional energy can increase sharply and toxic substances can accumulate after a few cascading processes occurred [33]. They emphasized that the primary aim of the bioeconomy is overall eco-efficiency instead of cascading maximization. Considering the use of more sustainable energy sources and monitoring the output of products and by-products to ensure the optimum of overall outputs are essential. Muscat et al. [32] raised concern about food-feed-fuel competition, highlighting the importance of new metrics that can capture resource and waste efficiency for the entire bio-based system, including a clear definition surrounding the concept.

3.4 Cradle to cradle (C2C)

William McDonough and Michael Braungart argued that the economic model of "take-make-dispose" would not overcome the environmental and economic challenges. They developed the cradle to cradle (C2C) concept as an innovative design based on the natural system's intelligence to delink the conflict between environment and economic growth [34]. The natural system refers to regenerative biological systems such as the closed-loop cycle, where almost all waste goes back into the system as input [35]. Within the C2C notion, two systems keep material in the loop: biological metabolism and technical metabolism [34]. Biological metabolism returns the materials from products to the environment through diffuse pathways. The products should be produced from renewable sources and become nutrients in producing new resources. The technical system expects non-renewable materials to be recirculated within the industrial system and becomes raw materials in manufacturing new products. Another possibility to keep materials in the loop is through cascading. The materials are repeatedly used in the technical cycle while experiencing a reduction in quality, and they finally flow back to the biological system [34].

Unlike eco-efficiency ,which aims for 'less bad' with the main objective situated in the economy, C2C aims to balance economic, environmental, and social goals as triple top lines aiming for 'more good' practice [35]. It consists of three main principles: waste equals food, utilize solar income, and celebrate diversity [34].

- Waste equates to food aims at closing the loop through recirculating nutrients in other product life cycles. It requires separating the technical and biological cycle; otherwise, a product that does not fit into either cycle cannot be reprocessed properly. While eco-efficiency aims to reduce the amount of waste, C2C focuses on designing a system where the waste output can be taken up as input by the other process.

- Utilize solar income implies the use of 'current solar income', interpreted as optimizing solar energy utilization through wind, geothermal, photovoltaic, hydro, and biomass. Currently total dependence on renewable energy is still not possible since the supply can only provide intermediate-load [36]. Hence, further advancement is necessary to realize this principle.
- Celebrating diversity mimics a natural ecosystem with a various organism. It could improve the system's resilience by avoiding one-size-fits-all solutions and designing products and systems based on local environments, cultures, and economics [37].

The cradle-to-cradle concept shares the closest characteristics with CE, and some researchers use the term interchangeably [38]. C2C also aligns with the Boulding principle of having a continuous cycle within the system.

Although it is believed that achieving sustainability is possible through transformation to cradle-to-cradle, criticism also arises. It has been argued that C2C is too focused on upcycling and waste elimination, which might not be relevant to all industries [39]. It also focuses on using infinite renewable energy sources that can neglect the energy efficiency aspect [37]. When renewable energy capacity is sufficient, focusing on effectiveness instead of efficiency will not have consequences. However, the renewable energy capacity is still insufficient; therefore, achieving C2C without considering efficiency can lead to higher energy consumption than the existing practice [37]. Moreover, Bjørn & Hauschild [40] reported that some composites could not be separated thermodynamically or require huge amounts of energy, leading to an increase in the environmental impact of unnecessary recycling processes. Additionally, biological nutrients can have a negative impact on the environment. Assuming that biological nutrients are fundamentally good is misleading, and management of natural materials is always needed [41].

3.5 Green supply chain management

Green Supply Chain Management (GSCM) originates from supply chain management and environmental management [42]. Based on Mentzer et al. [43], a supply chain is a network of actors involved in the movement of services, products, finances, and information to customers on both the upstream and downstream sides. They made a distinction between supply chain and supply chain management, where they argued that the supply chain is an existing phenomenon in business, while supply chain management (SCM) is managed deliberately by actors in the network. SCM is then defined as a holistic strategy in synchronizing traditional business functions with inter-functional coordination within a company and inter-corporate coordination within a supply chain to improve the performance of every single company and its supply chain [43].

The evolution of SCM into GSCM was closely tied to environmental degradation caused by economic activities, resulting in pressure to integrate environmental management into organizational operations [42]. GSCM has overlaps with sustainable supply chain management (SSCM), even these terms are used interchangeably [44]. Nevertheless, GSCM is deemed to be more suitable with CE better than SSCM since its emphasis on balancing the economy and the environment [45]. In the GSCM, environmental thinking is an integral part of each stage in the supply chain, covering design, material sourcing, manufacturing, transportation, consumption, and EoL [42]. Government regulation, company initiative, consumer awareness and supplier requirements are the main drivers of GSCM, while reverse logistics (RL) is a means to bridge collaboration between company and supplier [46]. This supply chain evolution, which combines forward and reverse logistics practices, will enable the system to close the loop [47].

The major feature of GSCM and/or SCM is the network built among all actors. In the ultimate supply chain, the network is rather complex, making an actor have a few different roles, e.g., a company can be a supplier and customer at the same time within a particular supply chain [43]. Inter-functional and inter-corporate coordination in the GSCM is the key in closing the loop through RL [47]. The added word 'reverse' to the term reverse logistics (RL) infers a backward logistics flow from the point of discarded products. The basic flows in RL consist of four different processes [48].

- Product acquisition/gatekeeping. This is where the products from end-users are acquired. These products will be processed further in the next RL stage or fixed and then returned back to the customers.
- Collection. Acquired products that are not returned to customers are then transferred to the collection facility.
- Sorting and inspection. Collected products are sorted based on the inspection result of their appearance and the condition of their contents.
- Disposition. A decision is made as to whether the sorted products will be reutilized or disposed of. Reutilization options include repair, reuse, remanufacturing, and recycling.

Some elements in the GSCM overlap with other CE building blocks. For example, waste hierarchy is part of disposition in RL, where the utilization of discarded products is optimized. Additionally, the broad coverage of GSCM that begins from the design phase correlates with C2C, where design plays a major role in ensuring that the products can be recirculated in either biological or technical metabolisms.

3.6 Product service system

Tukker & Tischner [49] defined product service system (PSS) as *'a mix of tangible products and intangible services designed and combined so that they are jointly capable of fulfilling* *final customer needs.'* The ideal vision of PSS implementation is that the business provides functions for the customers, so they do not need to own the product [50]. Product-oriented companies have the incentive to produce as many goods since the revenue comes from selling the product, whereas service-oriented firms are paid to provide a service through the products so that they have the incentive to prolong the product lifespan to ensure that the products are used intensively. The value proposition emphasizes service delivery instead of ownership [51]. Consumers can experience access to a certain service without taking care of and maintaining the product. This approach allows companies to shift toward circularity by minimizing throughput and the intensive use of products.

There are different types of PSS which can be grouped into four main categories: product-oriented service, use-oriented service, result-oriented service, and demand-side management [50,52].

- Product-oriented service. This model still relies heavily on product sales, complemented with added service. The environmental benefit comes from prolonging the product's life through maintenance so that the overall consumption of materials and energy are less. Examples of product-oriented services including maintenance contracts or take-back agreements.
- Use-oriented service. The provider still owns the product, which is intensively utilized by sharing or renting it out. Environmental benefits are derived from the efficient use of materials and energy since fewer products are needed for more users. Nevertheless, the users sometimes use the product carelessly, knowing they do not own it. Moreover, better planning and infrastructure are needed to implement the model (e.g., car-sharing system, leasing jeans, laundrettes).

- Result-oriented service. The material and energy efficiency in this model comes from selling the ‘result’ required by the consumers. Here, the environmental benefits are expected to be higher than use-oriented and product-oriented services since the providers can come up with entirely new concepts in fulfilling the consumers’ needs while considering sustainability. At the same time, this model poses bigger challenges because business providers sometimes need to change the whole business model. Examples include payment per km driving or company delivering a ‘pleasant atmosphere’ in offices instead of air conditioning equipment.
- Demand side management. This originally comes from the energy sector, where the economic benefit of reducing energy demand is more than increasing capacity. In PSS, this model resembles a result-oriented service, where the most efficient solution will be delivered based on electricity consumers' needs (e.g., a company provides ‘heat’ for an apartment).

Nonetheless, shifting to a complete service-system presents challenges from both the consumer and business sides. Implementing PSS requires a cultural shift since society attaches status to the ownership of goods, and there is still a lack of awareness among consumers of how the system can work [49]. Ownership provides intangible value for consumers, such as experience, brand value, self-esteem, peer acceptance, and a certain level in society [50]. Business also faces challenges such as cultural inertia, where the current business already provides profit, difficulty assessing the trade-off between environmental and economic saving, and lack of regulatory support [53].

4 Toward CE conceptualization and definition

This study deduced that the concepts mentioned in the previous section were developed independently to improve environmental quality due to rapid modern development while maintaining economic gain. Each concept embeds the idea of a closed economic model as Boulding [4] introduced. They focus on the supply chain initiated with raw material extraction and ending when the product is discarded. The concepts have their own features while intermeshing with each other, and altogether they build a new broadly independent concept now known as CE. The breadth and novelty of CE turn it into an umbrella concept where it builds links between independent concepts. An umbrella concept is a notion or idea applied lightly to incorporate and account for various phenomena [54]. An umbrella concept can draw a relation between pre-existing concepts that were initially unrelated by converging on specific, shared characteristics. Not only does the definition of the umbrella concept suits CE, but its trajectory also started with the period of excitement [54]. The excitement sparked because CE can seemingly fill the knowledge gap on maintaining economic growth without risking the environment. Moreover, it is expected to shed light on sustainable development and its goals because some perceive them as vague [6].

The excitement period was followed by a validity challenge, where we are now and in which the concept started to be contested [54]. Many researchers started questioning the theory and practice of the concept which started with the contention of CE itself [6]. The absence of a consistent definition can lead to various implications. First, it leads to confusion in academic and political discussions. A concrete example comes from scientific communities that formulated different definitions in a wide range of directions, which is also reflected in how national governments incorporate CE into their national policy [5,55]. Second, it can delay the transition: the premise brought about by CE is promising for business, but the lack of clarity of the concept, along with the lack of guidelines and examples, deter businesses from making a shift towards implementing CE [38].

4.1 Circular economy throughout product life cycle

CE building blocks are independent, interrelated concepts derived from the spaceman economy model [55]. These concepts have their own distinctive features and serve differently along the product life cycle. This study synthesized these features and resulted in different characteristics showing how practices can contribute toward CE regarding resource flow throughout the product life cycle. These characteristics include resource conservation, narrowing the loop, slowing the loop, and closing the loop [51,56]. Resource conservation ensures sufficient resources for the future and avoids or minimises the environmental impact of extraction and use [57]. When discussing resource flow, narrowing, slowing, and closing the loop are the terms coined by Bocken et al. [51]. Narrowing the loop is about resource efficiency, slowing the loop deals with resource prolongation during the consumption stage, and closing the loop requires the resources to be recirculated back into the supply chain.).

This study drew together the interaction and role among CE building blocks regarding the resource flow throughout the product life cycle, as shown in Fig. 1. As the CE building blocks overlap, the implementation of one concept will result in the indirect implementation of one or more other concepts (McDonough Braungart Design Chemistry, 2013; Tukker, 2015). The overlaps are due to the major motivation of CE in balancing the environment and economy [2], leading to the concepts organically intermeshing with each other. Moreover, as concern about the environment grows, governments are becoming more involved and setting regulations that can tie concepts together. For example, the European Parliament updated waste regulations to strengthen the waste hierarchy by setting new targets for recycling, phasing out landfills and introducing schemes to extend producer responsibility [58]. These three updated regulations can be approached from the post-consumer management perspective using waste management or GSCM. Another example is the industrial ecology application in an eco-industrial park (red box in Fig. 1). Organic waste from one company flows to another company as a resource and is transformed into valuable materials. The concept of exchanging material and energy between industries in a certain system boundary is based on the industrial ecology principle, but waste valorization is a part of the bioeconomy, waste hierarchy, and C2C [33]. It combines economic activity that utilizes renewable resources with the proper treatment of waste so it will not just end up in landfills and can be turned into a regenerative system.

Although these building blocks construct CE, they tend to be not circular as individual concepts. As shown in Fig. 1, GSCM and WM cover a system outside the circular product life cycle boundaries. GSCM and WM deal with reverse logistics and waste management [47], where they follow the hierarchical level of treating discarded products, starting with the highest recovery value up to landfills [15]. Nevertheless, these concepts offer important strategies toward circularity, such as optimization (e.g., for routing or location) and waste regulation (e.g., recycling or recovery target). Therefore, limits are applied to the parts of concepts that are aligned with CE characteristics.

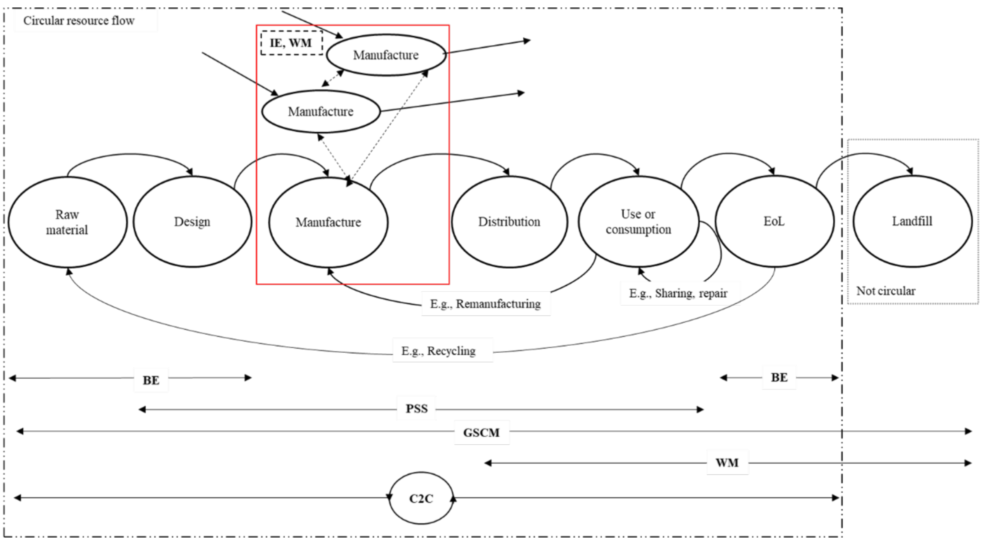

Fig. 1 CE building blocks related to resource flow in a circular product life cycle (IE: industrial ecology, WM: waste management, BE: bioeconomy, PSS: product service system, GSCM: green supply chain management, C2C: cradle to cradle).

This intermeshing is not subject to debate about which practice belongs to which concept. Instead, it should be deemed as an opportunity to flexibly combine one practice with another when there is a gap to fill, or there is a barrier to implementing a particular concept. The diversity of its building blocks and multifaceted approach enable flexibility in its implementation along the material life cycle (Fig. 1). The material life cycle within the supply chain itself can be straightforward or more complex where it involves various parties, and each party has more than one role in the supply chain [43]. The network built may be inside one company among different functions (inter-functional) and outside one company (inter-corporate), stretching from micro up to macro network implementation [43]. This level of implementation is also discussed widely in CE, where it covers three different levels, namely micro (individual firm or consumer), meso (eco-industrial system), and macro (city, province, region, nation) [6].

4.2 CE enablers and definition

This study formulated four enablers: policy, research and technological development (RTD), business model, and consumers. Each enabler plays a key role in one or multiple product life cycle stages. Their interaction is also essential to expedite CE.

It is known that the lack of government support is a substantial barrier for companies to adopt more environmentally oriented practices [59]. It signals the need for better and holistic policies that target and support the actors within the product life cycle. Currently, policies that support CE are focused on the EoL stage based on the notion that in a circular system waste is phased out [55]. Nonetheless, the policies around waste management have ambiguity, such as in the definition of waste and by-products or the variety of targets throughout the waste hierarchy [20]. The policies can also take on different instruments, including administrative (e.g., directive, target), economic (e.g., tax, incentive), and informational (e.g., ecolabelling) [59].

RTD covers a broad range of possible actions to help expedite CE throughout the product life cycle. Industry 4.0 is deemed to be a path of transition toward CE. Implementing more advanced technology (e.g., blockchain, internet of things (IoT), big data, digital twins, additive manufacturing, etc.) and data-driven decision-making can improve design, manufacturing, distribution, logistics, measurement and traceability [45]. Blockchain provides transparency and the history of product flow. Its ledger can assist the industry in handling real-time data so that process design and operation can be improved [60]. Joshi and Gupta [61] explained that IoT and big data enable companies to optimize their processes by using sensors to collect data and control their production lines. Nevertheless, barriers to adopt advanced technologies exist. The concern is about the availability of certain technology and the adoption lag that may result from path dependency. Some technologies are not economically or technically viable on a commercial scale [62]. Other barriers are high investment costs, supply chain integration, lack of skills, resistance to change, and lack of understanding regarding the benefit [63].

Business model is the third enabler for CE. Switching from a linear model will require businesses to rethink how they create, deliver, and capture values. Six circular business models are based on product design through slowing and closing the loop [51]. The circular business model underlines the importance of delivering performance with ownership of the products remaining with the providers instead of selling the products [64] as demonstrated by PSS [50.

- An access and performance model. The consumers do not own physical products of the service they obtain (e.g., car-sharing, leasing phones, launderettes).
- Extension of the product value. Exploiting the residual value from products by returning to the manufacturer (e.g., remanufacturing parts, retailer accepting clothing return).
- Long-life model. Companies deliver long-life products, including maintenance (e.g., luxury products offer longer-lasting products).

- Encourage sufficiency. Companies deliver solutions to reduce consumers' consumption (e.g., energy service companies).
- Extending resource value. Exploiting the residual value of resources through recycling (e.g., collecting fishing net as raw material for rug).
- Industrial symbiosis. Utilizing residual outputs from one process for another within geographic proximity (e.g., EIP Kalundborg).

Companies should rethink the way they obtain revenue while supporting efforts to retain products as long as possible in the consumption stage and close the loop by using secondary materials or parts [50]. Providing reliable and accessible service centers is also important; otherwise, buying new products will be preferred over getting them repaired [51].

Consumers are the last enablers in CE. They play a central role in the use stage of the product life cycle. The success of circular business model implementation relies on how consumers perceive and behave toward the new model. Ownership is still valued more highly; moreover, information about how the new business model works should disperse quickly and be easily understood [50]. Consumers can opt to deliberately reject the product in the form of service and choose the conventional model, where they own the products. Consumers are also responsible for taking care of shared products and repairing them instead of discarding and buying new ones.

This study defined CE based on the literature synthesis regarding its building blocks, characteristics, and enablers. CE can be seen as a regenerative economic model focusing on resource flow and management through the use of renewable resources, resource efficiency, prolonging resources at the consumption stage, and recirculating resources from discarded products into the value chain, enabled by research and technology development, the business model, consumers, and policy. Kirchherr et al. [6] reported that the business model and consumers are CE enablers. We would also like to point out that technology and policy are other enablers worth including in the definition.

5 Circular economy practices throughout the product life cycle

5.1 Circular practices and their related circularity characteristics

Numerous circular practices have been implemented even before the term CE was coined. This study describes those practices based on the literature analysis that was summed up as various approaches throughout different product life cycle stages. Table 1 summarizes these circular approaches that were categorized based on CE characteristics defined in Section 4.1, namely resource conservation, resource efficiency (narrowing the loop), prolonging the resource life span (slowing the loop), and recirculating secondary resources into the production process (closing the loop). The approaches are about circular practices that different actors can adopt, whereas circular characteristics refer to contributions toward CE associated with a certain approach.

This research described six product life cycle stages: raw material sourcing, product design, manufacturing, distribution, use, and EoL management. These stages are suitable for classifying the approaches (Table 1) since they cover the whole phases of resource or material flow, and they could also provide distinction when resource or material moves from one stage to another. Moreover, these stages are considered common in the product life cycle [38]; hence, they could assist actors in the product life cycle in adopting any practice. Raw material and design stages are associated with all activities related to resource extraction and planning for creating objects, respectively. These stages are followed by manufacturing, where the resources taken from the first stage are combined with the plan from the design stage to produce products. The products are then distributed to users for use or consumption. Lastly, the products enter their EoL stage when users are disposing them. Meanwhile, the approaches were various practices resulting from literature analysis that can contribute toward CE through resource conservation, narrowing the loop, slowing the loop, and closing the loop.

Table 1. CE approaches throughout the product life cycle

Product life cycle stage	CE approach: Approach	CE approach: Description	Circularity characteristics	Reference
Raw material	Environmental stewardship	Sourcing raw material from area/nature which is managed responsibly.	1	[56]
	Renewable alternative	Switching to a renewable alternative that is regenerative and available abundantly.	1	[30]
	Green procurement	Activity in purchasing raw materials, goods or services that embed a lower environmental impact.	1	[65]
	Taxation	Taxing virgin natural resources to conserve limited resources and reduce environmental damage by encouraging the use of less harmful alternatives.	1	[66]
	Eliminate distorting subsidies	Reduce subsidies that can harm the environment (e.g. energy subsidies used to extract a dirty energy resource or grow certain crops).	1	[67]
Design	Design for durability	Delivering products with physical and emotional durability supported by maintenance and repair service.	3	[51]
	Design for modularity	Delivering products that are subdivided into smaller independent parts, which can be assembled based on need, repaired and upgraded.	2, 3	[68]
	Design for disassembly	Delivering products that can be separated and reassembled. It also supports ease of repair and recycling.	3, 4	[51]
	Prototyping and design for feedback	Creating product prototypes and testing them to get immediate feedback (through surveys, sensors, digital twin, and interviews) before going to the full scale market	2	[69]
	Dematerialization	Reducing size, weight, or amount of materials incorporated to reduce the environmental impact without compromising the quality of the products.	2	[70]
	Production on demand	Products are made when demand is present, designed and maintained to specifically meet certain customers. Customers are expected to return since a company-customer relationship is built.	2	[64]
	Life cycle assessment	Designing a product by taking into account its environmental impact. Multiple alternatives (e.g., different suppliers, different materials, different lifespans, different disposal, etc.) can be compared.	1	[71]

Product life cycle stage	CE approach		Circularity characteristics	Reference
	Approach	**Description**		
Manufacturing	Energy and material efficiency	Optimize the process so that the energy and material inputs are decreased to produce the same output.	2	[1]
	Energy recovery	Utilizing energy waste or by-product into usable heat or electricity.	4	[71]
	Material recovery	Utilizing the residual value of products or material from the manufacturing process and putting it back into the process.	4	[51]
	Renewable energy	Using renewable energy sources.	1	[1]
	Industrial symbiosis	Building a network with other manufacturers in proximity to exchange knowledge, material, energy.	4	[23]
	Leasing service	Providing service to satisfy users' needs without owning the goods.	3	[50]
	Advanced technology	Switching to more advanced technology to improve efficiency (e.g., automation or additive manufacturing).	2	[72]
	Cascading	Converting biomass in a sequential manner following a biomass pyramid in order to optimize the resources.	2	[30]
	Eco-labelling	Voluntary practice to signify the overall environmental performance of products.	1	[71]
Distribution	Route and schedule optimization	Determining the most efficient route and schedule for distributing goods.	2	[73]
	Reusable packaging or pallets	Using pallets that can be used multiple times before being recycled.	3	[74]
	Leasing pallets	Purchasing service through renting pallets to focus on the operation and leaving maintenance to the leasing companies.	3	[75]
Use or consumption	Sharing	Sharing the use of the products from user-to-user intensifies the use of the product.	3	[50]
	Leasing	Accessing products to fulfil users' needs without having ownership.	3	[50]
	Repair and maintenance	Extending the product lifetime by restoring to an optimal function.	3	[51]
	Reuse	Extending the product lifetime through second hand use.	3	[51]
	Repurpose	Using products for a purpose other than their original function.	3	[76]
	Digitalization	Accessing products in digital form instead of physical form (books, telecommunication, etc.).	2	[77]
	Refurbish	Restoring and updating old products.	3	[76]
End-of-life (EoL)	Source separation	Separating discarded products to optimize recovery of the remaining value.	4	[78]

Product life cycle stage	CE approach		Circularity characteristics	Reference
	Approach	Description		
	Dynamic waste collection	Planning a waste collection schedule based on real-time bin occupancy to improve efficiency.	2, 4	[79]
	Route optimization	Determining the most efficient route to collect waste.	2, 4	[80]
	Take back program	Collecting, sorting, and processing discarded products to be manufactured (e.g., take back under extended producer responsibility scheme)	4	[47]
	Environmental target	Setting policy target to improve EoL management through e.g., collection, recycling or recovery targets.	4	[81]
	Recycling incentive	Providing incentives for recycling behavior (e.g., deposit-refund system).	4	[82]
	Location optimization	Optimizing the location of disposal points to increase the collection rate.	4	[83]
	Remanufacture	Utilizing parts of discarded products for new products with the same function.	3, 4	[76]
	Recycling	Process of recovering and using materials to obtain the same or lower quality.	4	[76]
	Energy recovery	Converting EoL products into usable energy (heat or electricity).	1	[76]
	Energy recovery optimization	Adjust WtE operating parameter to improve energy recovery	1,2	[84]
	Cascading waste	Converting biowaste in a sequential manner following the biomass pyramid in order to maximize the waste.	2, 4	[17]

1: resource conservation, 2: narrowing the loop, 3: slowing the loop, 4: closing the loop

The examples in Table 1 can act as a database to guide any actors to understand CE and its practical actions. The suppliers, manufacturers, transporters, consumers, and waste management companies create the whole product life cycle ecosystem. The database offers choices for different actors to take actions based on what is preferred and possible to implement. Different organizations may approach CE differently: bigger organizations with more resources and capabilities may implement multiple approaches, whereas smaller ones can choose to start with one approach toward circularity. This database could also serve policymakers as a basis for formulating policies. By looking at different approaches, policymakers can identify CE implementations that have not been supported sufficiently by the policies since it is identified that policy is one of the enablers in CE.

The results show the distribution of CE characteristics among all circular approaches. Most approaches have one primary CE characteristic: resource conservation, narrowing the loop, slowing the loop, or closing the loop. Some approaches show two major characteristics: narrowing and slowing the loop, slowing and closing the loop, narrowing and closing the loop. Among 43 approaches found in the literature, the circular characteristics of resource conservation comprised 20%, whereas narrowing, slowing, and closing the loop constituted 26%, 26%, and 28% of the practices. The distribution of CE characteristics was also distinctive at certain stages of the product life cycle.

The higher proportion of closing the loop contribution indicated the close connection between waste management and CE. It could be the result of the long history of waste management implementation in Europe, which started with Ad Lansink proposing a waste hierarchy in 1979 [85]. The concern toward waste management continues to evolve through different principles such as 'polluter pays' and extended producer responsibility where various policy instruments (e.g., recycling target, advanced disposal fee, ecolabeling, deposit-refund system) are employed [86]. The waste management concept also overlaps with the other CE building blocks (Fig. 1), which implies its necessity within the environmental aspect throughout the product life cycle. It also explains the association of CE with EoL management and the notion of phasing out waste brought up by Kirchherr, Reike, and Hekkert [6]. The focus on improving waste management continues, and the government supports it. European Union consistently increases the target for reuse and recycling of municipal solid waste and requires separate hazardous waste collection from household [58]. On country level similar action is taken in Finland to tighten the regulation regarding source-separated waste and increased the recycling target [87].

5.2 Circular economy implementation

It is unlikely for a system to be 100% circular indefinitely; on the other hand, it is also rare that a system is a completely non-circular system. The diverse possible approaches in employing CE show flexibility in CE implementation and the limitation of a system being 100% circular. Implementing a circular economy initiative can be divided into two main perspectives: top-down and bottom-up. A top-down initiative refers to implementation that is characterized by the influence of command and control from the government, while a bottom-up implies that the initiative starts from individuals, organizations or civil society who demand greener actions, products, or legislation [55].

The top-down initiative dominates CE implementation [88]. A top-down initiative starts with a global goal, which can later be specified in more detail in sub-plans at the lower level of the hierarchy. Examples can be found in countries such as China, the Netherlands, Denmark, and many other European countries which have formulated a national plan or road map toward CE. China is known as one of the pioneers of top-down initiatives. Scholars suggested CE to the Chinese government in 1998 and got picked up quickly by the authorities in 2002. It was followed by composing a draft of the CE promotion law and regulation of electronic waste that was preceded by the cleaner production promotion law and the amended law on pollution prevention and control of solid waste [89]. It was then reported that there was an improvement in industrial waste reclamation of about 16-34% in four major Chinese cities, including Dalian, Tianjin, Shanghai, and Beijing [90]. Several European countries have devised a CE roadmap or action plan to move toward circularity. The Netherlands has launched a government program for CE aiming at developing CE by 2050 [91]. The program targets to reduce the use of primary raw materials and set priorities for several sectors, including biomass and food, plastics, manufacturing industries, construction, and consumer goods. Another example can be found in the Danish government, which has developed a strategy to achieve circularity through digitalization, design, consumption, and recycling [92].

At local levels, the top-down initiative includes an EIP. The government supports the EIP through subsidies and supportive policies. This practice is commonly found in South Korea. South Korea has a long history of industrial ecology, and its initiatives are reflected through the transformation from industrial clusters into EIP. Ulsan city was appointed as an industrial cluster to boost the economy in 1962, and later, the Ministry of the Environment imposed more stringent environmental regulations in response to environmental problems caused by industrial cluster activities [26]. They also reported that the approach of industries to this regulation was applying end-of-pipe measures that were believed to be insufficient. the authorities decided to institutionalized cleaner production and environmental management systems for single companies and the entire cluster as one unity while providing funding [26]. It has resulted in symbiosis and allows the reutilisation and treatment of solid waste, wastewater, sludge, energy, boiler water, and other materials.

The bottom-up initiative tends to focus on more specific goals that were initially started by smaller entities. It can lead to the creation of a strong and widespread network as the action gains more traction. The bottom-up initiative will then be integrated into a higher-level goal or initiative. One example can be derived from the repair café movement. Repair cafés started in the Netherlands as groups of people getting together with expert volunteers who helped people to repair everyday items [93]. The practice has been formalized to help the adoption process, and now it has become a global movement. Another bottom-up initiative was companies recognizing that a circular approach can benefit economically. The adoption circular business model can be translated into various circular approaches. For example, the business model of access and performance model can be applied through sharing or leasing approaches, whereas extending resource value can be employed through a take-back program. Werning and Spinler [94] emphasized the importance of reverse logistics in CE and discussed multiple barriers, including its organization, acquisition process, leakages, channel selection, costs, quality, and quantity of products. Companies transitioning to provide services instead of selling their products may face resistance from their consumers [50]. The take-back program requires a solid reverse logistics channel with multiple actors such as companies, collection and recycling operators, consumers, and municipalities.

On a larger scale, the bottom-up initiative is found in EIP. This park formation is due to organic happenstance when different industries are located close to each other. An example of a bottom-up initiative is the industrial park in Kalundborg, Denmark. The industrial area dates back to the 1960s when a symbiosis occurred organically because of the need of a certain company that could be fulfilled by another nearby company. Later, more and more companies entered into agreements to exchange energy and material when they realized the economic gain to be had from this arrangement [95].

Top-down circularity can create significant change when policy interventions are applied effectively. It provides straightforward guidelines and unified goals or targets because the initiative is centralized. On the other hand, the overly broad framework from a top-down initiative can be difficult to translate to lower-level organizations whose varied resources and capabilities lead to unrealistic expectations. Meanwhile, a bottom-up initiative is decentralized. Therefore, an initiative that starts from the concerned parties will tend to be more realistic since it will be adjusted with their vision, resources, and capabilities. However, a decentralized initiative may not be aligned with the central authority's goals and will not be supported by the government. Nonetheless, these two types of initiatives are not mutually exclusive, and both can complement and assist the transition toward CE.

6 Conclusion

The circular economy is gaining momentum, making it an especially popular concept. The widespread discussions of CE and frequent use of the term can obscure its meaning. This article aims to contribute to CE conceptualization and its implementation by identifying its building blocks, characteristics, enablers, and role in the resource flow throughout the product life cycle. The building blocks used to conceptualize CE include waste management, industrial ecology, bioeconomy, cradle to cradle, green supply chain management, and product service system. The synthesis of CE from preexisting concepts that intermesh together makes it an umbrella concept that helps to connect diverse ideas. Based on our findings, we define CE as *a regenerative economic model focusing on resource flow and management through the use of renewable resources, resource efficiency, prolonging resources at the consumption stage, recirculating resources from discarded products back into the value chain, enabled by research and technology development, business model, consumers, and policy*.

Furthermore, a database tool has been developed to assist CE implementation. The database contains 43 circular approaches covering different stages of the supply chain. The specific circular contribution such as resource conservation, resource efficiency (narrowing the loop), resource prolongation (slowing the loop) and resource recirculation (closing the loop) was determined for each approach. The research findings have shown that the most common circular contribution throughout the supply chain was closing the loop.

Given the current prominence of CE, many implications can be inferred from the findings for all actors involved directly or indirectly in any stage of the supply chain. The actors can be companies, governments, NGOs, and scholars. The government can take top-down strategy by composing supportive policies based on the possible circular approaches in the database, while NGO or scholars can assist through a bottom-up scheme. The database tool will allow the company to initiate the most viable actions to be more circular.

This study has some limitations concerning the method and the topic. A narrative review lacks systematic literature selection that could result in a biased outcome. An additional search was employed in this study based on a reference list of retrieved literature and knowledge found in the retrieved literature. Improvement can be applied by conducting systematic reviews covering a similar topic or using this study to navigate topic selection within the CE area. Since CE covers a broad range of areas, certain boundaries were selected in this paper that might exclude other aspects. The current study analyzed CE building blocks and their role throughout the general supply chain. Future studies can expand similar research to understand the role of CE building blocks by incorporating specific products or systems such as plastic or nutrients cycle. The applicability of the database can be tested by using real cases from different sectors, including primary, manufacturing or services. The expected outcome will be a more refined database showing the applicability of different circular strategies within a specific sector.

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Author Contributions

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Monetary and environmental damage cost assessment of source-separated biowaste collection: Implications of new waste regulation in Finland

Bening Mayanti[1] and Petri Helo[2]

Abstract

This study develops a cost model covering monetary and environmental damage costs for source-separated biowaste collection. The model provides an improved basis for decision-making by including environmental damage costs compared to the assessment that considers the only monetary cost. The monetary cost calculation integrated route optimisation using existing road networks, while the environmental damage cost was estimated using the life cycle impact assessment method based on the endpoint (LIME) model. The model was tested in the Finnish case where the new law implements the stricter requirement for source-separated biowaste. The costs of collection, transportation and treatment of three different scenarios were assessed: mixed waste under the old law (MW-OL), biowaste under the new law (B-NL) and mixed waste without biowaste under the new law (MW-NL). The results showed the economic and environmental benefits of sourced separated biowaste. The overall cost of collection and transportation (CT) under the old law and new laws were 80.7 € Mg^{-1} and 81.1 € Mg^{-1}, respectively. Treatment costs were 79 € Mg^{-1} and 64.8 € Mg^{-1} under the old and new laws, respectively. The damage costs for CT under the old and new laws were 0.23 € Mg^{-1} and 0.24 € Mg^{-1}, respectively. At the same time, the damage costs from the treatment stage were 4.9 € Mg^{-1} and 3.5 € Mg^{-1} under the old law and new law, respectively. The model supports decision-making when the collection scheme requires a change. Failing to plan an optimised solution and cost will lead to inefficient systems.

Keywords

Waste management, monetary cost, damage cost, route optimisation, emission inventory

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Introduction

Waste generation is an unavoidable consequence of anthropogenic activities that may damage human health and the environment (Logan and Visvanathan, 2019). Consumption patterns, population growth, income and climate affect the quantity and composition of municipal solid waste (MSW) generation (Kinobe et al., 2015; Ragossnig and Schneider, 2019). Consequently, MSW varies among regions, while at the same time, it becomes crucial to manage MSW properly to protect human health and the environment. With the new paradigm of the circular economy (CE), waste is seen as a resource, so the end-of-life (EoL) notion should be phased out by recovering the remaining value of the waste (Geissdoerfer et al., 2017).

Policies are the key driver to achieve circular waste management in Europe (Wilts et al., 2016). European Commission formulates directives that should be achieved by the members without specifying the laws or means to fulfil the target. An example is the waste directive framework that requires the country members to recycle biowaste up to 70% by 2030 (European Commission, 2021). This affects how waste management policy is applied in each country member. In Estonia, the biowaste collection is a source-separated for at least 10 apartments (Tallinn, 2022). Sweden has a different approach where the target is set on the national level, guiding the municipalities in structuring the waste collection and treatment. Household biowaste can be collected separately or as a fraction of mixed waste, with about two-thirds of all municipalities collecting source-separated food waste at varying degrees (Avfall Sverige, 2018). Denmark applies a similar approach as Sweden regarding biowaste collection (State of Green, 2017). Finland amended the Waste Act (646/2011), which sets the goal for the reuse and recycling of 55% of MSW in 2025, 60% in 2030 and 65% in 2035. The new

[1]Vaasa Energy Business Innovation Centre, University of Vaasa, Vaasa, Finland
[2]Networked Value Systems, Department of Production, University of Vaasa, Vaasa, Finland

Corresponding author:
Bening Mayanti, Vaasa Energy Business Innovation Centre, University of Vaasa, Yliopistonranta 10, Vaasa 65200, Finland.
Email: bening.mayanti@uwasa.fi

legislation also tightens the obligation of biowaste collection. Biowaste should be collected separately if the residential property has at least five apartments which is stricter than 10 apartments in the previous legislation. Source-separated municipal waste effectively optimises resource recovery and avoids landfilling as long as proper infrastructures are available (Daskal et al., 2018; Kawai and Huong, 2017). A source separation system is generally more complex and costly than a mixed waste system, and it involves more stakeholders and alternatives for collection routes (Lavee and Nardiya, 2013). It also requires higher collection costs, more labour, new infrastructure and vehicles (Groot et al., 2014). Therefore, estimating the economic implication of implementing a source-separated collection system is imperative to formulate a charging method to cover the entire cost. It is especially significant since the collection and transportation (CT) stage may contribute up to 70% of the total cost of waste management (Rathore and Sarmah, 2019).

Previous studies about waste CT focused mainly on investigating the shortest distance and fuel consumption (e.g. Edwards et al, 2016; Kinobe et al., 2015; Nguyen and Wilson, 2010) as well as the monetary costs incurred (e.g. Boskovic et al., 2016; Larsen et al., 2010). Meanwhile, the environmental damage cost in waste CT is still underrepresented. Therefore, more accurate analysis employing real road networks to optimise collection routes as well as estimate monetary costs and damage cost is important.

This research aims to develop a monetary and environmental damage cost calculation model for waste CT. The calculation of monetary cost and environmental damage cost can provide a more comprehensive insight into which trade-offs may occur. We also utilised a tool that allowed the use of an actual road network. The applicability of the cost model is demonstrated through a real case study of Kauhajoki municipality in Finland. The amended Waste Act (646/2011) resulting in new legislation that requires biowaste separation for residential property with at least five apartments, compared with 10 apartments in the previous legislation. The amended legislation creates new clusters of properties that necessitate a separate organic waste collection; hence, new route planning is needed. Specific objectives are then formulated to achieve the aim of the study by focusing on: (i) optimisation of the waste collection route, (ii) identification of the monetary and environmental damage cost (€ Mg^{-1}-waste) for CT, (iii) comparison of the monetary and damage cost of biowaste treatment in waste-to-energy (WtE) and anaerobic digestion (AD) facility and (iv) investigation of sensitive parameters in the cost model. The research can denote how to deal with the challenge of transitioning toward CE, where a source-separated system will be the norm.

Materials and methods

Study area

The study area comprises Kauhajoki municipality in the Southern Ostrobothnia region of Finland, which covers the area of 1315 km^2 with 13,172 inhabitants (Kauhajoki, 2020). The temperature varies from −11 °C up to 21 °C throughout the years. The warm season lasts for about 3 months from June to August with an average temperature of 16 °C, whereas the cold season lasts for almost 4 months, from December to March, with an average temperature below 1 °C (Weather Spark, 2021). The growing session starts at the end of April and lasts for about 140–175 days in the area of study (Finnish Meteorological Institute, 2020).

The publicly owned waste management company provided data concerning the study area where the law will affect the collection scheme. The data included the location of collection points (CPs) and the total number of households being served. The number of CPs is 198, and it comprises 2202 individual households. The collected waste is transferred to the transfer station (TS) in Teuva, a nearby municipality, and gathered collectively with waste from neighbouring areas. The waste is then transported to the treatment facility in Vaasa region for about 82 km from the TS where WtE and AD are situated at the same area. Figure 1 displays the location of the CPs, garage (G), TS and AD.

Some of the CPs are scattered and located near the bounding coordinates. However, most CPs are concentrated in the built-in area (Figure 1). Each CP has a different number of waste bins (wheelie bin) with a total volume of 240 L. One bin is used for 5–19 households, whereas two bins cover 20–45 households.

The amended waste act commits to improving collection and recycling by increasing new targets for waste separation and recycling. The previous waste act required source-separated biowaste for a minimum of 10 dwellings clustering together. With the amended version, a municipality must collect biowaste separately from residential properties with minimum dwellings of five no later than May 2024 (Botniarosk, 2020). Containers to separate waste, including mixed waste (for incineration), biowaste, paper, metal packaging, glass packaging, cardboard packaging and plastic packaging, are provided in the residential properties with at least 10 dwellings. For the dwellings of 5–9, it is compulsory to provide containers for mixed waste and biowaste, whereas the other types of containers are optional. Lastly, for dwellings between 1 and 4, biowaste separation is not required, and it can be collected together in the mixed waste container. In addition to curbside containers, a bring-in scheme is also applied through ecopoints and recycling stations. There are more than 2500 ecopoints throughout Finland where the inhabitants can bring their packaging and paper waste. Recycling stations are facilities similar to ecopoints that accept a wider variety of waste, including stone material, electrical and electronic equipment, wood or hazardous waste.

The new policy will create a cluster where biowaste should be collected separately. The old policy collects mixed waste once a week. Whereas the new one collects biowaste weekly during summer (12 weeks) and fortnightly for the rest of the year (40 weeks). The remaining mixed waste is also collected in a fortnightly manner throughout the year. The collection starts with the vehicle leaving the G, driving to dwellings, and collecting the waste. After completing the collection, the vehicle goes to the TS, where the waste collected from Kauhajoki is combined with waste collected from neighbouring areas. A bigger vehicle then collectively

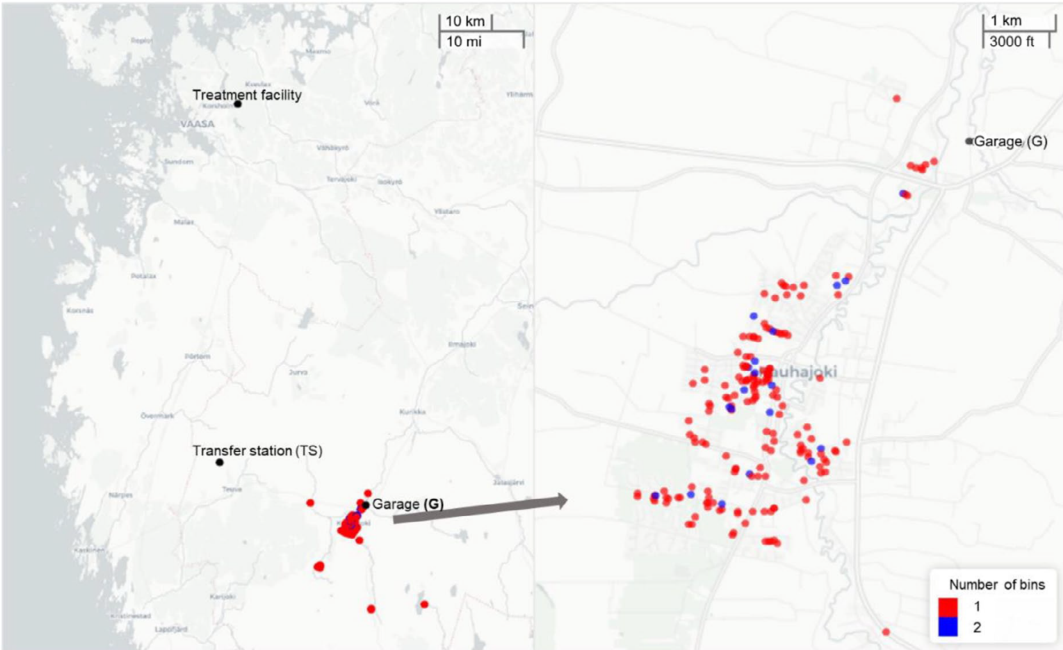

Figure 1. Left: Location of the collection points, garage, transfer station and anaerobic digestion. Right: The concentrated CPs.

transports the waste to the treatment facility area, where AD and incineration are located within the vicinity area. In this study, different terms are used to describe the activities of transporting waste. Collection refers to the activity where the waste in each CP is picked up, whereas waste transportation concerns transporting waste from TS to the final treatment facility AD.

The scenario covered by this study includes mixed waste under the old law (MW-OL), biowaste under the new law (B-NL) and mixed waste without biowaste under the new law (MW-NL). These three scenarios applied different collection frequencies where MW-OL (16 Mg truck) and MW-NL (20 Mg truck) collect the waste weekly and fortnightly throughout the year, respectively. Meanwhile, B-NL collects the waste weekly during the summer period (8 Mg truck) and fortnightly for the remainder of the year (16 Mg truck). The waste generation was estimated at around 6.6 kg household^{-1} week^{-1} (MW-OL), 2.6 kg household^{-1} week^{-1} (B-NL), and 4 kg household^{-1} week^{-1} (MW-NL) (Botniarosk, 2020; HSY, 2021).

Route optimisation

Waste CT can be seen as a conventional vehicle routing problem with a few caveats, such as the limited capacity of the vehicle and different quantities of waste in each CP (Nambiar and Idicula, 2014). In this study, there were two routes of optimisation for waste CT: (i) waste collection where the vehicle (vehicle 1) leaves the G, collects the waste, drives to TS and goes back to the G, (ii) waste transportation where the vehicle (vehicle 2) transports waste from the TS to the treatment facility such as WtE or AD plant and drives back to the TS. Vehicle 2 has a bigger capacity since it transports waste gathered from multiple collection areas. The sequence of waste CT are presented in Figure 2.

Route optimisation was executed using Open Door Logistics (ODL) software. ODL is a geographic information system (GIS) software built on Graphhopper and Jsprit library, whereby the optimisation uses real road networks (Welch, 2017). The user inputs information such as service time, waste quantity in each CP, the geocode of CP, time constraint and capacity constraint. It will generate the optimal route, including travel time, distance and waste quantity in each road segment.

Cost calculation

The costs consist of monetary costs and damage costs. Monetary costs were vehicle cost, bin cost, fuel cost, labour cost and treatment cost. Damage cost indicates the monetary value of welfare losses due to the impacts of emissions caused by anthropogenic activity on the environment (Liu et al., 2021). In this case, the emission is caused by collection, transportation and treatment. Usage rate was applied to the annualised fixed cost of truck and bin so that the cost incurred is associated with the use of the infrastructure. The results are expressed as € Mg^{-1}-biowaste as well as the annual sum cost in €. The analysis was conducted to estimate the total cost before and after the new law is applied, hence the implication of the new law could be inferred by comparing these two types of costs.

Vehicle cost. The vehicle cost refers to the fixed annual capital costs, insurance, maintenance, licence, oil and miscellaneous costs (COWI, 2004). Capital cost is a one-time expense incurred

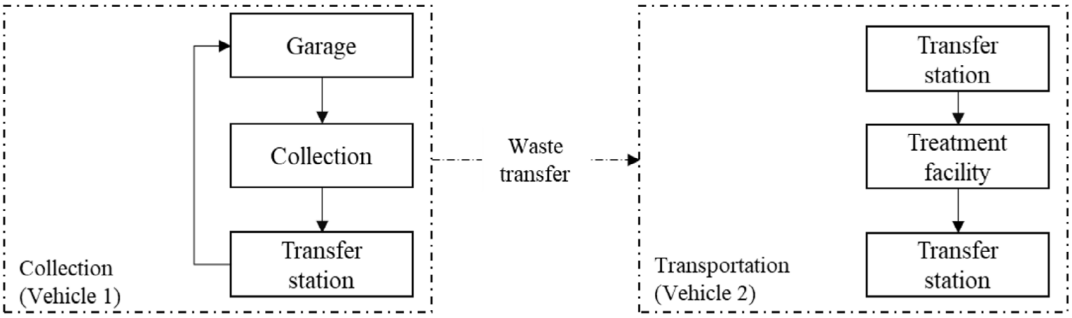

Figure 2. The sequence of waste CT.

due to the purchase of the vehicle and allocated equally throughout the vehicle's lifetime. Other cost components were estimated using a percentage of the capital cost (detail is provided in the Supplemental Table 1).

To calculate annualised capital cost (A), equation (1) was applied:

$$A = \frac{P}{\left(\frac{(1+r)^n - 1}{r \cdot (1+r)^n}\right)} \tag{1}$$

where P, r and n refer to lump-sum cost, interest rate and a vehicle's lifetime, respectively. When the available information concerning the cost of the waste truck did not represent the reference year used in this study, adjustment to the year 2019 using Marshall and Swift index (see Supplemental Table 3) was applied by using equation (2):

$$\frac{\mathrm{Pr}_1}{\mathrm{Pr}_2} = \left(\frac{I_1}{I_2}\right) \tag{2}$$

where I_1 and I_2 are the index in the year 2019 and the original year, respectively, while Pr_1 and Pr_2, show the calculated price for 2019 and the original price, respectively.

Price adjustment was also required when the available information showed a different truck capacity than the desired one. The calculation to approximate capital cost for different capacities follows the rule of six-tenths, as shown by equation (3) (Serna, 2018):

$$C_B = C_A \cdot \left(\frac{S_B}{S_A}\right)^{0.6} \tag{3}$$

where C_A and C_B are the cost of the known and desired equipment, respectively, whereas S_A and S_B are the capacity of the known and desired equipment, respectively. The total annual fixed cost was divided by the usage rate. For the truck, the annual usage rate was estimated using equation (4) (Martinez-Sanchez et al., 2015):

$$\mathrm{AUR}_{\mathrm{truck}} = \frac{\mathrm{AT} \cdot \mathrm{AL}}{T_r} \tag{4}$$

where $\mathrm{AUR}_{\mathrm{truck}}$ is the annual usage rate of the truck, AT the annual time that the truck can be used. AL refers to the average load of the truck, and T_r is time per round of collection or transportation. A summary of vehicle costs is shown by equation (5):

$$\mathrm{Vehicle}_{\mathrm{cost}} = \frac{A + \mathrm{Insurance} + \mathrm{Miantenance} + \mathrm{License} + \mathrm{Oil} + \mathrm{Miscellaneous}}{\mathrm{AUR}_{\mathrm{truck}}} \tag{5}$$

Bin cost. Bin cost was calculated based on the annualised investment divided by the usage rate. The cost estimation concerning bin cost per Mg biowaste was calculated using equation (6). The price of a single bin (€), the lifetime of a bin (year), the number of bins and the annual usage rate (Mg year^{-1}) are represented by C_{bin}, $\mathrm{Ls}_{\mathrm{bin}}$, n_{bin} and $\mathrm{AUR}_{\mathrm{bin}}$, respectively. For bin, the annual usage rate equals the waste generation of the biowaste.

$$\mathrm{Bin}_{\mathrm{cost}} = \frac{\frac{C_{\mathrm{bin}}}{\mathrm{Ls}_{\mathrm{bin}}} . n_{\mathrm{bin}}}{\mathrm{AUR}_{\mathrm{bin}}} \tag{6}$$

Fuel cost. The cost of fuel consumption was the sum of service consumption and travel consumption then divided by waste quantity (m_{biow}). Service consumption occurs in CPs, TS and AD. The costs from service were categorised as idle cost and lift cost equation (7).

$$\begin{aligned} \mathrm{Fuel}_{\mathrm{cost,service}} &= \frac{\mathrm{Fuel}_{\mathrm{cost,idle}} + \mathrm{Fuel}_{\mathrm{cost,lift}}}{m_{\mathrm{biow}}} \\ \mathrm{Fuel}_{\mathrm{cost,idle}} &= \left(\mathrm{ST}_{\mathrm{TS}} + \mathrm{ST}_{\mathrm{AD}} + \sum(\mathrm{ST}_{\mathrm{CP}} - T_{\mathrm{lift}})\right) \cdot \mathrm{FC}_{\mathrm{idle}} \cdot C_{\mathrm{diesel}} \\ \mathrm{Fuel}_{\mathrm{cost,lift}} &= \mathrm{FC}_{\mathrm{lift}} \cdot T_{\mathrm{lift}} \cdot \mathrm{CP}_{\mathrm{s}} \cdot C_{\mathrm{diesel}} \end{aligned} \tag{7}$$

ST_{TS} is service time in the transfer station (hour), ST_{AD} refers to service time in an anaerobic digestion facility (hour), ST_{CP} is service time in each collection point (hour), T_{lift} is time per lift (h), FC_{idle} is fuel consumption when idling ($l\ h^{-1}$), and C_{diesel} refers to diesel price (€ l^{-1}). For the cost related to fuel consumption when lifting the bin, FC_{lift} is fuel consumption per lift ($l\ h^{-1}$), T_{lift} is the time required per lift (h) and CP_s is the number of CPs.

Fuel consumption from travelling (FC_{LU}) was calculated based on load weight and truck velocity in each segment of the collection route. The segment of the collection route was generated by ODL and represented the actual distance. Fuel consumption during travelling was then calculated using equation (8) (NTM, 2020).

$$\mathrm{Fuel}_{\mathrm{cost,\,travel}} = \frac{\sum(FC_{empty} + (FC_{full} - FC_{empty}) \cdot LCU_{weight})}{m_{biow}}$$
$$LCU_{weight} = \frac{\text{load weight}}{\text{maximum weight capacity}} \tag{8}$$

where FC_{LU}, FC_{empty} and FC_{full} are fuel consumption at load capacity utilisation (LCU), empty load and full load in $l\ km^{-1}$ (see Supplemental Table 1 for the values), whereas LCU_{weight} refers to LCU. Information about the values of fuel consumption when the load is empty (FC_{empty}) and full (FC_{full}) are required to estimate fuel consumption (FC_{LU}). These values vary for different vehicle capacities and roads, such as highway/rural and urban (Swahn, 2008). The type of road was distinguished based on the calculated velocity based on information generated by ODL concerning the distance and travel time in each segment of the roads. Urban roads allow a maximum speed of 50 km h^{-1}, whereas nonurban roads permit speeds up to 80 km h^{-1} (European Commission, 2017).

Labour cost. Labour cost refers to the wage of the driver and bin loader for waste collection, whereas waste transportation requires only a driver. The labour cost was about 25 € $hour^{-1}$ (see Supplemental Table 1). Equation (9) is used to calculate the labour cost.

$$\mathrm{Labour}_{\mathrm{cost}} = \frac{\mathrm{Wage} \cdot t_{tot} \cdot n_{labour}}{m_{biow}} \tag{9}$$

where Wage, t_{tot}, n_{labour} and m_{biow} represent the driver's wage (€ $hour^{-1}$), the total time to collect or transport waste (the sum of travel time and service time) in hours, the number of workers and biowaste mass in Mg, respectively.

Treatment cost. Information regarding treatment costs in WtE and AD were calculated using tools that were developed for WtE optimisation (Mayanti et al., 2021b) and marginal cost estimation of an AD (Mayanti and Helo, 2021). The calculation for WtE was modified based on the waste composition in this study (Supplemental Table 2), whereas the AD in this study was assumed to treat kitchen waste. The lower heating value (LHV) and life cycle inventory (LCI) of waste treated in WtE were estimated by a tool called waste incineration life cycle inventory tool (WILCI) (Beylot et al., 2017). The methane potential, which determined the total methane production and its fugitive, was estimated using stoichiometry based on waste chemical composition (Supplemental Table 1). The focus of the study was on the CT since they will be affected directly by the new law. However, including the treatment stage could provide more comprehensive knowledge of whether a trade-off exists between treatment schemes.

Damage cost. Damage cost was calculated for CT as well as the treatment phase. The method to calculate damage cost was the Life Cycle Impact Assessment Method based on Endpoint 3 (LIME3) model (Inaba and Itsubo, 2018; Liu et al., 2021). The model was initially established by Japan's Ministry of Economy, Trade and Industry to portray the environmental and social situation in Japan (Itsubo and Inaba, 2003). It has been undergoing development, and in 2016 LIME 3 model was developed with various coefficients applicable to different countries (Inaba and Itsubo, 2018). Applying LIME to obtain environmental damage costs starts with building inventories, followed by multiplying the inventories with the factor/coefficient. The framework used to calculate damage cost in waste treatment is shown by Figure 3 (Liu et al., 2021). Three coefficients can be used to obtain different results, namely damage factor (DF), weighting factor (WF) and integration factor (IF). The last coefficient is a product of multiplying DF with WF. The environmental damage costs can be obtained by multiplying emissions in the inventory with the IF.

Emission inventory. Emission inventories were collected from the collection, transportation, and treatment phase. The emission from the CT phase included CO_2, and it was calculated using the emission factor (EF) issued by the Finnish Standards Association (2013), as shown by equation (10). Em_i^{CT} shows emission (kg) *i* from collection and transportation, EF_i^{CT} is tank-to-wheel diesel emission factor of *i* from collection and transportation (kg l^{-1}), CF_i is the characterisation factor of emission *i* if applicable, whereas $FC_{service}$ and FC_{travel} are fuel consumption during service and travel (l), respectively.

$$Em_i^{CT} = EF_i^{CT} \cdot (FC_{service} + FC_{travel}) \cdot CF_i \tag{10}$$

For the treatment phase, CO_2 was not considered. It is assumed that CO_2 from the biowaste during treatment is equal to CO_2 absorbed during biomass production; therefore, it should not be reported (IPCC, 2019). Sulphur dioxide (SO_2), nitrogen oxides (NO_x) and non-methane volatile organic compounds (NMVOC) were considered for damage cost estimation. Other emissions were not included due to the limited coefficient in the LIME database. Moreover, other emissions can be considered marginal (Liu et al., 2021). The emission inventories from the incineration process were obtained using the WILCI (Beylot et al., 2017). Meanwhile, methane (CH_4) was the emission considered from

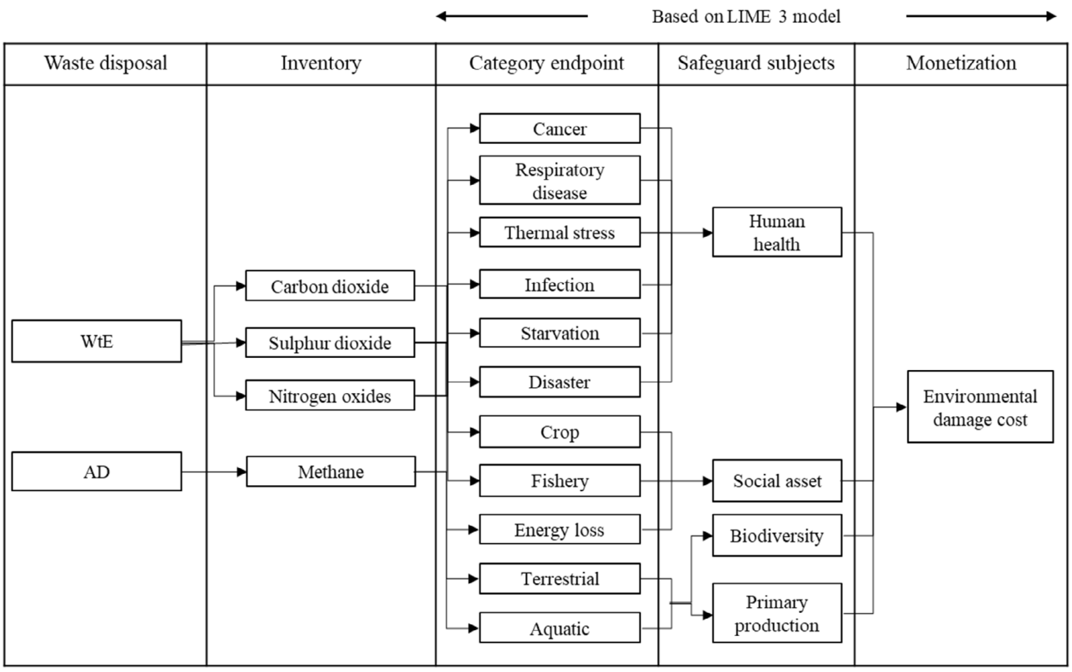

Figure 3. Framework of environmental damage cost of WtE and AD.

AD due to unintentional leaks, accounting for 1% of total CH_4 production (Liu et al., 2021). The general emission equation from AD was calculated using equation (11).

$$\mathrm{Em}_i^{\mathrm{AD}} = \mathrm{EF}_i^{\mathrm{AD}} \cdot m_{\mathrm{biow}} \cdot \mathrm{CF}_i \tag{11}$$

where $\mathrm{Em}_i^{\mathrm{AD}}$ and $\mathrm{EF}_i^{\mathrm{AD}}$ are emission i from anaerobic digestion (kg) and emission factor i in anaerobic digestion (kg Mg^{-1}-biowaste), respectively.

Damage cost calculation. After collecting emission inventories, the damage cost calculation can be estimated using the LIME database (LCA Society of Japan, 2018). The general equation to estimate the environmental damage cost is shown by equation (12).

$$\mathrm{DC}_i^h = \mathrm{Em}_i^h \cdot \mathrm{IF}_i \tag{12}$$

where DC_i^h is the damage cost of emission i from waste management h (€ Mg^{-1}-biowaste), Em_i^h represents the emission i from waste management h (kg), and IF_i refers to the monetary-based integration factor of emission i (€ kg^{-1}-emission) at which the value can be obtained from the coefficient list (LCA Society of Japan, 2018).

Sensitivity analysis

Sensitivity analysis examines the results as an effect of modifying the inputs. Perturbation analysis was applied by changing the input parameters by 10% one-at-a-time while maintaining all other parameters the same as the baseline values. The outputs from perturbation analysis were then used to calculate the sensitivity ratio (SR), as shown in equation (13). SR is the ratio between two relative changes, namely the relative change of result and input parameter (Bisinella et al., 2016).

$$\mathrm{SR}_i^j = \frac{\left(\dfrac{\Delta\,\text{result}}{\text{initial result}}\right)^j}{\left(\dfrac{\Delta\,\text{parameter}}{\text{initial parameter}}\right)_j} \approx \frac{\partial z_j}{\partial x_i}\frac{x_i}{z_j} \tag{13}$$

Results

Route optimisation

Route optimisation was done for CT. The same routing was obtained for MW-OL, B-NL, and MW-NL. The total waste generated ranged from 5.7 Mg up to 17.6 Mg in each collection. During the collection stage, different results are displayed by different scenarios. The travel time and service time required for MW-OL and MW-NL were 4.12 and 4.51 hours, respectively. The B-NL scenario needed 4.12 hours of travel and 3.71 hours of service. In between CPs, the velocity varied in each road segment, showing the minimum and maximum values of 30 km h^{-1} and 59.5 km h^{-1}, respectively. Total fuel consumption for MW-OL and MW-NL was not much different, as shown by 64.5 L and 65.9 L in each collection, respectively. For B-NL, fuel consumption during summer collection would consume 57 L, and the rest

Table 1. Summary of route optimisation of waste CT.

Item	Unit	Collection				Transportation[a]			
		MW-OL (w)	B-NL (w)	B-NL (f)	MW-NL (f)	MW-OL (w)	B-NL (w)	B-NL (f)	MW-NL (f)
Time	Hour	8.6	7.8	7.8	8.6	3.0	3.0	3.0	3.0
Waste	Mg	14.5	5.7	11.5	17.6	14.5	5.7	11.5	17.6
Fuel usage	Litre	64.5	57.0	59.7	65.9	23.5	9.3	18.5	28.5
Average speed	km hour^{-1}	33.0	33.0	33.0	33.0	73.2	73.2	73.2	73.2
Distance	km	203.9	203.9	203.9	203.9	165.4	165.4	165.4	165.4
Truck capacity	Mg	16	8	16	20	32	32	32	32

w: weekly collection; f: fortnightly collection.
[a]The transportation phase applied a usage rate so that the fuel consumption is proportional to the waste generated from Kauhajoki to the total capacity of a 32-Mg truck.

of the year would require 59.7 L. The Supplemental (Figures 1 and 2) presents the CT route results.

Route optimisation for waste transportation was simple since it only had one stop in the treatment facility. The transportation used a truck with a 32-Mg capacity, transporting waste gathered from multiple locations. The travel and service time required to complete the transportation was about 2.26 h and 0.75 h, respectively, with a total distance around 165 km. Table 1 displays the summary results of route optimisation for both CT.

Monetary and damage costs of CT

The monetary and damage costs results will be presented as absolute annual value and relative cost per Mg waste. The assessment resulted from CT costs for about 80.7 € Mg^{-1} total waste under the MW-OL scenario. Under the new law, B-NL and MW-NL would cost 91 € Mg^{-1} and 74.7 € Mg^{-1}, respectively. On an annual basis, the cost under the new law will be equal to 81.1 € Mg^{-1} of total waste. Figure 4 shows the annual cost items for waste CT under the old and new laws.

A similar pattern was observed in the old law and new law scenarios where the highest contributor to the total cost was vehicle costs ranging between 44 and 47%, followed by wage contribution for about 37–39%. Contribution from bin and fuel consumption was relatively low and did not exceed 16%. The analysis showed that separating biowaste caused a slight increase of about 359 € annually, translating into 0.5 € Mg^{-1} total waste.

The results of damage costs under the new law showed a marginal increase compared to the old law, corresponding to the monetary costs results contributed by the fuel consumption. The annual damage costs difference between old and new laws was about 9.1 €. The damage costs of the old law and new law that resulted from CT were 0.23€ Mg^{-1} and 0.24€ Mg^{-1}, respectively. Figure 5 shows annual damage costs under the old and new laws for waste CT.

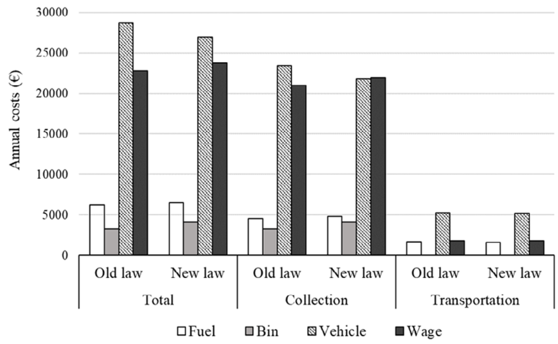

Figure 4. Annual monetary costs of waste CT.

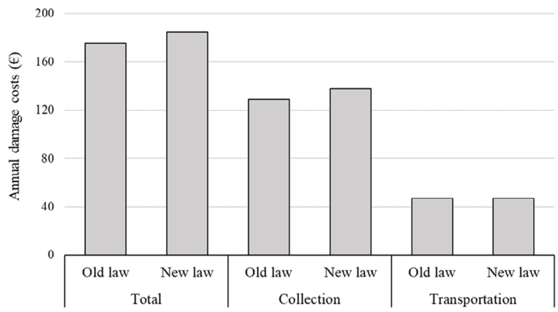

Figure 5. Annual damage costs of waste CT.

Monetary and damage costs of waste treatment

Cost calculation of waste treatment was applied using the system expansion principle. The products generated from waste treatment were deemed as credits for being an alternative in substituting the original products (e.g. biosolids could substitute artificial fertiliser). It brought revenue in the monetary cost calculation; meanwhile, it was considered environmental credits instead of damage in the damage cost calculation. Before the new law takes effect, mixed waste is treated in WtE. The new law will treat the biowaste in AD and the remaining mixed waste in WtE. The treatment cost using WtE associated with MW-OL was 79 € Mg^{-1} total waste. Under the new law, the cost was around 64.8 € Mg^{-1} total waste. It consisted of B-NL treatment using AD and MW-NL treatment using WtE, which each of them costed for about 23.3 € Mg^{-1} biowaste and 91.7 € Mg^{-1} mixed waste, respectively.

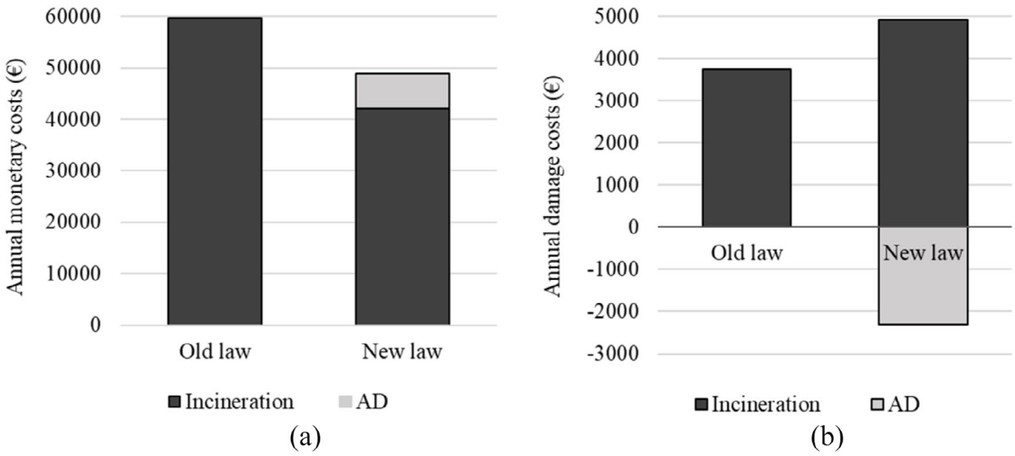

Figure 6. Annual monetary costs (a) and damage costs (b) under different laws.

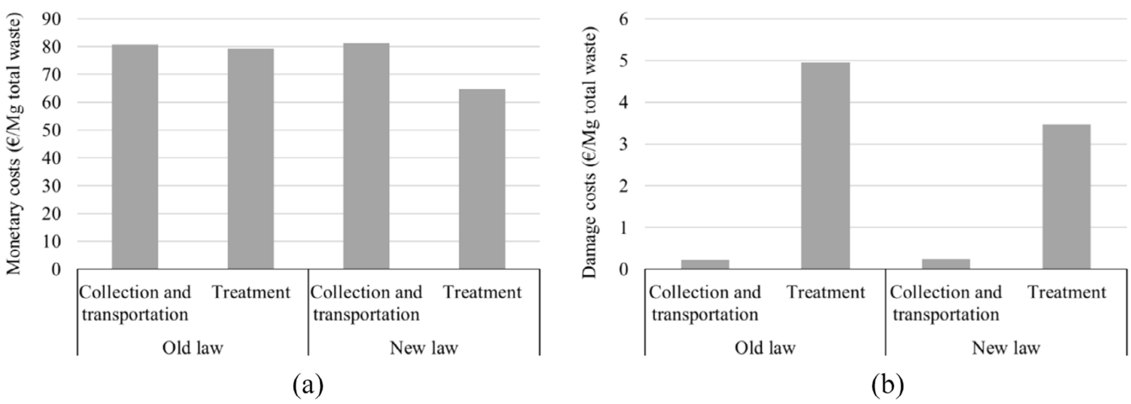

Figure 7. Monetary costs (a) and damage costs (b) per Mg waste under different laws.

Under the old and new laws, the environmental damage costs were 4.9 € Mg^{-1} total waste and 3.5 € Mg^{-1} total waste, respectively. The total damage costs under the new law consist of treatment costs using WtE and AD for about 6.5 € Mg^{-1} mixed waste and −3 € Mg^{-1} biowaste, respectively. The damage cost due to incineration was more expensive under the new law; nevertheless AD showed negative damage cost resulting in more benefit compared with the old law. Figure 6 shows annual monetary and damage costs of waste treatment under the old and new laws.

Comparison between the old law and new law

This study emphasised the monetary cost and damage cost implications of a waste diversion strategy where source-separated biowaste is implemented. Due to the new law, this diversion strategy required a new collection infrastructure that was translated into additional cost. A cost comparison between the old and the new laws was conducted to obtain a more comprehensive perspective. The old law scenario is the baseline case before implementing the source-separated biowaste strategy. It will collect mixed waste (burnable and biowaste) from the household and treat it in incineration. With the diversion strategy, biowaste is collected separately and is treated in AD. In general, the results suggested that implementing the new law would benefit both from monetary and damage costs perspectives. The overall costs of collection, transportation and treatment under the old law and new law would be around 160 € Mg^{-1} and 146 € Mg^{-1}, respectively. The same pattern was found in the damage costs assessment, where the old and new law results were 5.2€ Mg^{-1} and 3.7€ Mg^{-1}, respectively. Figure 7 displays the costs comparison between the implementation of the old and new laws.

Sensitivity analysis

The SR was calculated to measure the most sensitive parameters for monetary and damage costs. If a change of parameter results in an SR value of 2.5, it indicates that a 10% increase of the parameter will increase the results by 25%. The tested parameters differed between monetary and damage costs because some parameters only affected monetary costs, such as bin price, discount rate, etc. Overall, there were 12 separate sensitivity analyses. These comprised the sensitivity analysis for the monetary cost of collection and treatment as well as damage cost for collection and treatment applied to MW-OL, B-NL and MW-NW. The complete results can be found in Supplemental Figures 3 and 4.

The SR of monetary costs applied to CT showed certain similarities. In all three scenarios, the most sensitive parameters to cost decrease were an increase in truck operating hours and lifetime. Meanwhile, waste quantity and vehicle cost were most sensitive to an increase of the costs. Overall, the SR values ranged from −0.66 up to 0.7. For the treatment using WtE and AD, different parameters were tested against different treatment methods. The similarity was found in MW-OL and MW-NL, where the treatment costs using WtE were most sensitive to the change of discount period (SR −0.4) and capital cost (SR 1). For the B-NL scenario, the most sensitive parameters were methane LHV (SR −0.5) and labour cost (SR 0.85).

For the damage costs, a similar trend showed in the results of SR on the CT. The most sensitive parameters that caused a decrease and increase in the overall damage costs were waste quantity and fuel consumption rate, respectively. The SR itself ranged for about −0.6 to 1. The most sensitive parameters for WtE treatment for both MW-OL and MW-NL were similar. The damage cost decrease was most sensitive to an increase of net efficiency of the WtE (SR −0.4 up to −0.3), whereas the damage cost increase was most sensitive to an increase of the fossil (SR 1 up to 1.4). Biowaste treatment using AD showed a difference where the overall damage cost was negative. Therefore, the positive value of SR indicates more benefit (the overall damage cost will be more negative), and the negative value of SR suggests damage (the overall damage cost will be less negative). For the B-NL scenario, the most sensitive parameters were fugitive methane (SR −0.03) and methane potential in the waste (SR 0.93).

Discussion

The importance of waste composition and quantity

Municipal waste composition and generation are affected by income, climate and demography (Kinobe et al., 2015). Hence, it varies among regions while at the same time, it is a crucial component in planning for collection, transportation and treatment. It affects the infrastructure needed, collection frequency and collection route. Waste composition is associated with the monetary costs and the environmental impact, especially during the treatment process. This study indicated the effect of waste composition on the monetary and damage costs of treatment using WtE. There was an increase in the monetary and damage costs of treating mixed waste under scenarios MW-OL and MW-NL. Although the LHV of waste in the MW-NL scenario was higher due to the removal of biowaste, the cost became higher. The damage costs were higher as the fossil carbon content in each Mg of waste was increasing. The increase of fossil carbon was mainly caused by plastic waste, which contributed to about a quarter of waste composition. Incinerating plastic waste and its association with the high environmental impact was demonstrated in a previous study done by Beylot and Villeneuve (2013). Simultaneously, the overall monetary cost was also increasing under the MW-NL scenario. WtE facility has limited thermal capacity; hence the increase of waste LHV will reduce the amount of waste being treated. The result confirms the guide provided by the waste hierarchy regarding the importance of material recovery before utilising WtE for energy recovery.

The knowledge regarding waste quantity also affects the optimisation of the collection, transportation and treatment. It determines the required infrastructures such as bin size, the number of bins, truck capacity, waste storage, feed preparation and treatment facility. This study used a value of 6.6 kg total mixed waste per household per week, consisting of 2.6 kg biowaste and 4 kg mixed remaining waste. The number was adopted from the report from waste management in Kauhajoki (Botniarosk, 2020), with an additional 5% as a buffer. The quantity of waste in the report shows the overall municipal waste, including household, private sector, public sector and other similar waste. The calculation was done using the assumption that the household generated 54% of total municipal waste (HSY, 2021).

Adding buffer value in calculation can help deal with some waste generation uncertainties. Denafas et al. (2014) reported that seasonality affects MSW composition and quantity. Statistics also show that was generation can change from year to year (Botniarosk, 2020). At the same time, too much buffer can lead to overestimation, which causes inefficient and costly systems. Evaluating the system periodically can prevent inefficiency so that collection frequency, route or vehicle capacity can be adjusted. The sensitivity analysis results also displayed the importance of waste quantity as the most sensitive parameter for the waste CT cost model. Ensuring that the data regarding waste quantity is as accurate as possible will result in reliable cost estimation. For comparison, waste generation in Kauhajoki was compared to other cities. Helsinki generates a higher household than Kauhajoki, as shown by the value of 7.1 kg per household (7.5 kg with 5% buffer), whereas household biowaste was about 2.8 kg (2.9 kg with 5% buffer) (HSY, 2021). In Copenhagen, the average household generates 3.5 kg of biowaste (3.7 kg with 5% buffer) in a week (State of Green, 2017).

Assumptions used and uncertainties of the results

Uncertainties in the results are generated by the accumulation of uncertainty present in the data input, methodologies, assumptions and formulas. The input values represent the average condition; therefore, unusual events or irregularities cannot be captured. Seasonality in the waste generation or situations that result in the vehicle moving slower can generate different outcomes. Other sources of uncertainties were the assumptions used to assess treatment costs. For the WtE facility, assumptions such as the emission factor of the air pollution control unit, the efficiency, and the amount of ash generation will affect the calculation results. Meanwhile, parameters in AD including the rate of fugitive methane, methane potential, efficiency and biosolids generation, also influence the outcomes. This study utilised typical WtE and AD used by previous studies (Mayanti and Helo, 2021; Mayanti et al., 2021a), since these studies used the Finnish context in building their calculation tools. Since it is not possible to test every possible assumption, sensitivity analysis was applied to understand the effect of each parameter. The assessment was conducted by applying perturbation analysis to obtain information concerning the magnitude of change in the results regarding shifting the input value. Knowledge about the most sensitive parameter could improve decision making and help assess the risk of a particular strategy associated with a particular input value. Moreover, it assists in predicting the results of a decision if a situation turns out to be different from the baseline prediction.

The results are expressed in € Mg^{-1} total waste; however, the overall annual cost was also assessed to provide a more comprehensive picture regarding the implications of changing waste management policy. Different studies used different measuring units, and implementing both could be useful to compare results. Martinez-Sanchez et al. (2015) reported that the collection cost of separated biowaste in Denmark was 96.3 € Mg^{-1}, whereas this study showed a result of 91 € Mg^{-1} biowaste. Seyring et al.(2015) reported the collection fee of separated biowaste in Tallin was charged between 1.5 and 4.2 € bin^{-1} emptying for a bin size of 240 L. The same bin size was utilised in this study, which resulted the cost of 2.4 € bin^{-1} emptying. In general, some results of this study correlated with others. The difference could be caused by the size of the study area, input parameters or assumptions. Comparing results can provide useful information concerning the possibility to generalise the study to another context.

Implications and limitations

The results provide information regarding the optimised route, monetary cost, and damage cost regarding new waste management regulations. It will have implications for different actors such as waste management companies owned by several neighbouring municipalities, households, property owners, waste treatment operators and the government. Monetary cost aimed to quantify the real and internal waste CT costs in order to prepare the infrastructure for the transition toward a source-separated waste system. The real cost is what the waste management company needs to be aware of in order to determine the fee that the household will bear. The environmental damage cost was applied to provide a comprehensive perspective from an environmental perspective. The overall results indicated that the new law would benefit from economic and environmental perspectives. A good planning becomes the key for all stakeholders involved in a waste management system. For example, some of the most sensitive parameters during waste collection were the waste quantity and vehicle cost. It pointed out that estimating waste generation was important since it would affect the choice of truck capacity.

For the authorities, the findings on monetary cost can be used as a reference in organising the CT operation (tender process and awarding contracts). The costs are split into two parts, where different vehicles are in charge of the CT. This separation helps clarify the cost incurred in each part, especially when the authorities plan to hire different waste transporters. This study provides preliminary knowledge for the authorities regarding the route, cost and emission. They will be able to fine-tune what the waste transporters offer to avoid an inefficient system. They can also start communication with property owners and households to inform the approximate additional waste bin and fee.

The study requires various inputs along with the choice of boundary, type of parameters and method that can lead to uncertainty. Primary data provided by waste management companies can reflect the actual situation; however, the remaining inputs must be supplemented by secondary data from the literature. The secondary data used in this study reflected either the Finnish or European context to produce a realistic result.

Conclusions

A model to estimate the overall waste management cost, including collection, transportation, and treatment, was developed. The costs consist of vehicle, labour, fuel, bin and treatment costs. The CT model was constructed through route optimisation based on real road network. The route optimisation generates the time and distance used to calculate costs related to fuel consumption and labour. This cost model helps stakeholders understand the economic implications of implementing different waste management strategies where more CPs and new infrastructure are required. The assessment of damage cost and comparison between different waste strategies provides further insights from an environmental perspective by presenting the trade-offs between monetary cost and environmental damage cost. The authorities are more informed and could weigh on different aspects before making decisions.

The applicability of the cost model is demonstrated through a real case where new legislation requires more source-separated organic waste. The model behaviour and sensitivity are also examined through sensitivity analysis. This study shows the significance of comprehensive assessment when the waste management policy changes the current practice. Separate assessment can indicate a higher cost when biowaste is collected and treated separately. The CT of source-separated biowaste could cost 91 € Mg^{-1} biowaste (B-NL scenario), whereas mixed collection before the new law was applied was around 80.7 € Mg^{-1} total waste (MW-OL scenario). However, a different conclusion was obtained when a comprehensive approach was applied. The analysis was done not only on the MW-OL and B-NL, but also on the MW-NL (74.7 € Mg^{-1} mixed waste). A similar conclusion was also derived from the treatment phase, showing the benefit of separating biowaste from the source and treating it in AD instead of WtE plant.

This research also emphasises the importance of sensitivity analysis in handling uncertainties, improving decision making and predicting outcomes. Various parameters affected the outcomes of monetary and damage costs differently throughout waste management phases (transportation, collection and treatment). Careful planning and realistic estimation of waste generation can create an efficient system. All actors and decision making can utilise this study to assist them in making a decision regarding waste management. The trade-offs between monetary cost and environmental damage costs can be considered so that the decisions could reflect beyond monetary benefit. There may be subjectivity, especially when there are multiple criteria in making a decision, and stakeholders can assign weight to each criterion. Nevertheless, careful deliberation is needed when generalising this study and applying it in other contexts because the research covered a distinct study area with specific parameters such as waste generation, collection frequency, service time and road network, which mainly reflect the Finnish context.

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ORCID iD

Bening Mayanti https://orcid.org/0000-0001-5073-7375

Supplemental material

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Closed-loop supply chain potential of agricultural plastic waste: Economic and environmental assessment of bale wrap waste recycling in Finland

Bening Mayanti [a,*], Petri Helo [b]

[a] Vaasa Energy Business Innovation Centre, University of Vaasa, Yliopistonranta 10, 65200, Vaasa, Finland
[b] Networked Value Systems, Department of Production, University of Vaasa, P.O. Box 700, FI, 65100, Vaasa, Finland

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ABSTRACT

It is estimated that 12000 tons of plastic waste is annually generated from the agricultural sector in Finland, and more than half of it comprises bale wrap films. Up to 70% of plastic film waste from the agricultural sector in Finland goes into landfills, and only around 10% is recycled. Recycling plastic material is desirable in order to close the loop in achieving a circular economy. This paper aims to assess the environmental and economic implications of bale wrap collection and recycling within the Finnish context. Two different collection scenarios, S1 (once a year collection) and S2 (twice a year collection), covering 179 farms, were assessed. The research applied vehicle routing problem and environmental life cycle costing to quantify the cost and environmental impact per ton of granulate recycled material produced. It took a consequential approach, where the system boundary was expanded, and product substitution was considered. Overall, S1 offers 27% more economic savings with 36% less global warming potential (GWP) than S2. The collection phase, which has not commonly been included in existing recycling studies, shows significance in both scenarios. Although it only contributed about 0.7-1.2% to GWP, collection accounted for 32-36% of the total economic cost. Critical parameters were primarily associated with the market substitution factor and material loss during the recycling process. This study demonstrates that recycling bale wrap can provide environmental and economic savings. Furthermore, it shows the importance of decision-makers in prioritizing goals to balance environmental and economic objectives.

1. Introduction

Plastic is a versatile material used in various applications due to its mechanical and chemical properties. In 2018, plastic converter demand in Europe was 52.2 million tons, covering various sectors such as packaging, building and construction, automotive, electrical and electronic, and agriculture (PlasticsEurope, 2019). In the agricultural sector, plastic is categorized into packaging and non-packaging (film), which constitutes 3.4% of the demand (Erälinna and Järvenpää, 2018; PlasticsEurope, 2019). Most agricultural plastic is of the film type, mainly used for greenhouses, mulching and low tunnels in Southern Europe, and silage and bale wrap in Northern Europe (Briassoulis et al., 2012).

The intensive use of plastic in the agricultural sector creates a waste problem. Finland generates 12,000 tons of agricultural plastic waste annually, dominated by bale wrap comprising around 7000 tons (Alenius, 2016). These plastic film wastes are disposed of in various ways such as landfilling (70%), open field burning (10%), energy recovery (10%), and recycling (10%) (Briassoulis et al., 2013; Erälinna and Järvenpää, 2018). The lack of proper management can be caused by difficulties in handling different types of plastic waste, impurities, and contaminants. Thickness is the main parameter for recycling plastic films, and multiple types of waste plastic films can result in mixed thickness that risks compositional uniformity (Briassoulis et al., 2013). Thus, source-separated collection or commingled collection with a proper sorting system becomes necessary. A thorough washing process is required to deal with impurities such as soil, organic material, dirt or metal (Briassoulis et al., 2013).

The handling of bale wrap waste in Finland goes against sustaining the production system and achieving sustainable development. Inability to capture the opportunity of recycling means financial loss and environmental damage. The former implies the loss of potential financial gain from recycled materials, while the latter is due to the increased demand for raw material and landfilling practice. Moreover, agricultural plastic film is composed of a limited range of resins such as low-density polyethylene (LDPE) or linear-LDPE (LLDPE), making it a good input for mechanical recycling (Borreani and Tabacco, 2017; Scarascia-Mugnozza et al., 2012). Implementing the circular economy (CE) model can

* Corresponding author.
E-mail addresses: bening.mayanti@uwasa.fi (B. Mayanti), petri.helo@uwasa.fi (P. Helo).

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Acronym list

APW	agricultural plastic waste
CE	circular economy
CLSC	closed-loop supply chain
EoL	end-of-life;
ELCC	environmental life cycle costing
FS	fossil resource scarcity
FU	functional unit
GWP	global warming potential
HDPE	high-density polyethylene
HT-C	human carcinogenic toxicity
HT-NC	human non-carcinogenic toxicity
LCA	life cycle assessment
LCC	life cycle costing
LDPE	low-density polyethylene
LLDPE	linear low-density polyethylene
MRF	material recovery facility
PE	polyethylene
PP	polypropylene
SC	sensitivity coefficient
SR	sensitivity ratio
TA	terrestrial acidification
WC	water consumption

improve the production system economically and environmentally.

CE advocates resource recirculation, and its implementation will decrease the need for virgin material in the production system (Ellen MacArthur Foundation, 2013; Genovese et al., 2017; Nasir et al., 2017). Transitioning toward CE requires the closed-loop supply chain (CLSC) to maximize product value in its entire life cycle through product acquisition, collection, sorting and recovery (Guide and Van Wassenhove, 2009). Product recovery allows post-consumer products to re-enter the supply chain as input through reuse, remanufacturing or recycling (Cannella et al., 2016; Hicks et al., 2004; Nasir et al., 2017).

Recycling enables CLSC through material recovery. Evaluating the environmental consequences of plastic recycling became prominent and has been commonly done through life cycle assessment (LCA). Most of the environmental assessment of plastic waste recycling has focused on municipal waste, as shown in various studies (e.g. Al-Salem et al., 2014; Faraca et al., 2019; Hou et al., 2018; Rigamonti et al., 2014; Shonfield, 2008) and very few have addressed agricultural plastic waste (APW) (Cascone et al., 2020; Gu et al., 2017). However, the environmental assessment needs to be integrated with an economic assessment to obtain comprehensive results to improve decision-making. Currently, the economic assessment of plastic recycling is still rare (Faraca et al., 2019). Furthermore, based on the authors' knowledge, none of the studies has addressed bale wrap waste collected at the source. In this case, the impact and benefits from recycling bale wrap waste have not been addressed sufficiently (Cascone et al., 2020; Horodytska et al., 2018). This creates a knowledge gap that can hinder the implementation of bale wrap recycling.

To assess whether CLSC for bale wrap is attainable, we investigated the environmental and economic impacts concerning the mechanical recycling of bale wrap waste within the Finnish context. This case study covers bale wrap waste generated by cattle farms in the Southern part of Finland and assesses collection and mechanical recycling as a recovery option for bale wrap waste by applying environmental life cycle costing (ELCC). ELCC offers comprehensive environmental and economic performance assessment to improve decision-making (Lichtenvort et al., 2008). It extends conventional life cycle costing (LCC) to be consistent with the system boundaries and functional unit (FU) of life cycle assessment (LCA) (Martinez-Sanchez et al., 2015). The main goals of this study are achieved by focusing on the following targets: i) quantifying the potential environmental and economic impacts of bale wrap waste recycling, ii) examining the contributions of key processes to environmental and economic performance, and iii) identifying critical parameters through sensitivity analysis.

The rest of the paper is organized as follows. Section 2 presents a selected literature overview of previous research work that is considered relatable to this study. Section 3 describes the case study as well as the material and methods. Section 4 reports the environmental and economic analysis results, and Section 5 presents conclusions and suggestions for further research.

2. Literature overview

In recent years, there has been a growing interest in CLSC to improve sustainability. One of the ways to attain it is through recycling and recovery of end-of-life products (EoL) (Das and Rao Posinasetti, 2015). As a versatile material used in various products, recycling plastic has a significant potential to close the material loop and divert it from landfills. The success of recycling depends not only on the recycling technology but also on other factors such as citizens' participation, segregation method (mixed or separated), contaminants and impurities, collection scheme (curbside or bring-in), collection frequency, and sorting process.

Collection schemes and their frequency can either support or hinder plastic waste recycling. They affect citizens' participation, recycling rate, and quality (Cole et al., 2014; Dahlbo et al., 2018; Hahladakis et al., 2018). Curbside collection is the preferred scheme for attaining a higher waste recovery than bring-in schemes for household plastic waste. Hahladakis et al. (2018) showed the recovery rate of curbside collection for household plastic waste was up to 90%, ten times higher than bring-in schemes. Jenkins et al. (2003) found that curbside collection increased citizens' participation in disposing of recyclable waste properly by 20% compared with bring-in schemes. Larsen et al. (2010) also reported a higher recycling rate for household recyclables when implemented curbside collection. Nonetheless, there will be trade-offs in each selected scheme. Curbside collection will require more trips, which translates into higher fuel consumption, emission, and cost. The LCA study showed that drop-off schemes generated less environmental impact than curbside schemes (Iriarte et al., 2008). However, there is no consensus about the most suitable collection method. The decision is specific and mainly based on the financial aspect, citizens' participation, and collection logistics (Iriarte et al., 2008).

The collection is also deemed to influence the quality of the recycled material. Hahladakis et al. (2018) concluded that collection schemes affected the contamination level of the waste and the quality of recycled material. WRAP (2010) and WRAP (2009a) reported similar results, showing the quality of waste collected through the curbside collection was better than other schemes, and they were less likely to be rejected in the material recovery facility (MRF). Meanwhile, Luijsterburg and Goossens (2014) reported the insignificance effect of collection schemes on the recycled material and emphasized the importance of sorting and reprocessing.

Besides addressing the collection issue, CLSC on plastics also focused on the environmental impact of recycling and the environmental benefit obtained from recycled material. Wäger and Hischier (2015) and Hou et al. (2018) evaluated the environmental performance of plastic recycling compared to other plastic waste treatments. Both studies showed that recycling was a better option than incineration or landfilling; moreover, recycling provided environmental savings due to avoiding virgin material production. The extent of environmental savings derived from recycling depends on the substitution factor. Simões, Xará and

Bernardo (2011) compared the environmental impact of anti-glare lamella (AGL) made of virgin high-density polyethylene (HDPE) and recycled HDPE generated from packaging recycling. It was found that AGL made of recycled material had more environmental advantage shown by a reduction in the fossil fuel impact category. Rajendran, Scelsi, Hodzic, Soutis and Al-Maadeed (2012) conducted LCA to compare the environmental performance of composite made of virgin plastics and recycled plastics. The result showed that the environmental benefit of recycled material could be different depending on the product application. For non-automotive applications, recycled material provided environmental benefits, whilst virgin material performed better in an automotive application. Gu, Guo, Zhang, Summers and Hall (2017) applied LCA to evaluate the environmental performance of different mechanical recycling routes for different types of plastics. They applied various substitution factors depending on plastic type, ranging from 10% to 50%, resulting in considerable environmental benefit.

Substitution factor is essential when quantifying the benefit derived from secondary material or comparing the consequence of using virgin material and recycled material. The majority of studies in LCA applied a 1:1 substitution factor of recycled material to virgin material (Laurent et al., 2014), implying that recycled material could substitute the same amount of virgin material with the same quality and acceptance (Gala et al., 2015). This practice could lead to an overestimated result, especially when some studies on agricultural plastics showed the weathering effect on the degradation of plastic properties (Basfar and Idriss Ali, 2006; La Mantia, 2002; Tuasikal et al., 2014).

A more comprehensive approach was found in studies that combined environmental and economic aspects through LCA and LCC. Martinez-Sanchez et al. (2015) provided a comprehensive model in performing an environmental and economic assessment of solid waste management using different types of LCC, namely conventional LCC (CLCC), environmental LCC (ELCC) and social LCC (SLCC). Each type of LCC has a different relationship with an LCA, and it consists of different types of cost items such as budget costs, transfer costs and externality costs (Edwards et al., 2018). Presenting the basic principles of cost analysis in harmony with LCA and examples of applying different types of LCC in waste management systems can provide comprehensive results to improve decision-making (De Menna et al., 2018). Accorsi et al. (2014) combined LCA and CLCC to evaluate the environmental and economic performance of food packaging throughout its entire lifecycle. Their results indicated that the best environmental performance depended on the EoL management. Simões et al. (2013) integrated LCA and SLCC to assess anti-glare lamella made of virgin and recycled HDPE. The environmental and economic consequences of anti-glare lamella production favoured the use of recycled material. Faraca, Martinez-Sanchez and Astrup (2019) performed ELCC of hard plastic recycling in Denmark. They combined the environmental and economic aspects to assess three different recycling systems for hard plastic collected at the recycling centre without considering the collection stage. The study showed the importance of integrated environmental and financial assessment as a key to improve decision-making.

This literature overview demonstrates that CLSC for plastic products has become an essential topic for sustainability. However, there is still a lack of studies on agricultural plastic despite the multiple uses of plastic films in the sector and EU priority to reduce the impact from the agricultural sector. Furthermore, the collection stage and economic assessment are not always included in LCA studies. Few studies on agricultural plastic have focused on the characteristics of agricultural plastic waste and specifications for mechanical recycling (Briassoulis et al., 2012, 2013). At the same time, the environmental impacts and benefits obtained from agricultural plastic recycling have been shown only by Gu et al. (2017) and Cascone et al. (2020). The former applied LCA to quantify the environmental impacts of recycling various plastics, including from the agricultural sector, whereas the latter assessed the environmental impacts of collecting and recycling greenhouse films. This paper attempts to fill the gap by conducting a more comprehensive study concerning agricultural plastic films by combining environmental and economic aspects using ELCC while considering the collection phase.

3. Materials and method

This paper aims to contribute to the current debate around CLSC as a strategy to shift toward CE by implementing ELCC to bale wrap recycling. Hunkeler et al. (2008) and Swarr et al. (2011) published a handbook and code of practice to apply ELCC by a parallel combination of LCA and LCC to consistently evaluate economic and environmental dimensions. Therefore, identical system boundary, FU, goal and scope must be adopted (Martinez-Sanchez et al., 2015).

3.1. Study area

The study area covered 179 small and medium farms in 51 municipalities in Finland. Finland has a large landmass and sparse population, making the collection a challenging task, especially given the company's plan to implement curbside collection. The study area included six regions, namely Southern Ostrobothnia, Tavastia Proper, Central Finland, Pirkanmaa, Ostrobothnia and Satakunta (Fig. 1). The information about farms' location was provided by a company that treats animal by-products and plans to expand its service into bale wrap waste collection for recycling. Bale wrap is a stretching film made of low-density polyethylene (LDPE) or linear-LDPE (LLDPE) and used to preserve and store forage to maintain feed quality for cattle that can only graze during the summer period. This resin results in minimum film thickness while providing maximum protection due to its mechanical properties (Borreani and Tabacco, 2017; Scarascia-Mugnozza et al., 2012).

There was a lack of studies and data regarding agricultural plastic waste in Finland; consequently, bale wrap waste generation from 179 farms was estimated using farm size and annual bale wrap waste (Alenius, 2016; Erälinna and Järvenpää, 2018; Naturresursinstitutet, 2020). The average number of cattle per farm in a municipality was estimated by dividing the cattle population by the number of farms; hence, all farms in the same municipality are assumed to have the same cattle numbers. This approach aligned with the pattern of farm distribution in Finland (Naturresursinstitutet, 2020b, 2020c) and was confirmed by the company that treats animal by-products. Each farm had cattle numbers ranging from 38 to 139, and their distribution is shown in Fig. 2.

The bale wrap waste per cattle was calculated by dividing annual bale wrap by the total number of cattle so that the waste generated in

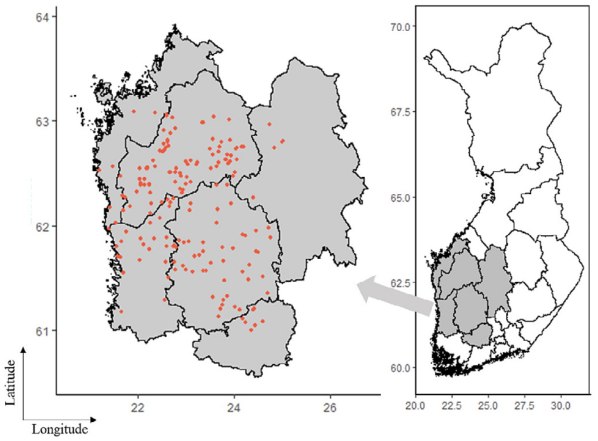

Fig. 1. Study area including the regions (grey color) and farm locations (coral dot). (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

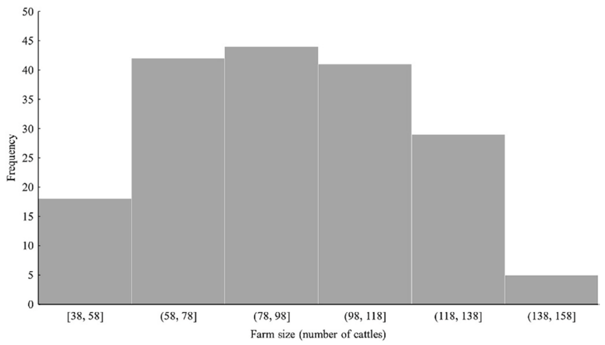

Fig. 2. Farm size distribution in study area.

each farm can be estimated using the product of waste per cattle and the number of cattle. The study resulted in around 137 tons of bale wrap waste annually, corresponding to 300–1100 kg per farm per year. Two collection scenarios were studied, namely S1 for once a year collection and S2 for twice a year collection, assuming that waste was not generated during the summer due to the grazing period. The collected bale wrap waste is transported to the recycling plant for reprocessing into recycled granulates (rPE).

3.2. Collection route

The optimized vehicle routing for S1 and S2 was analyzed using Open Door Logistics software ("Open Door Logistics," 2014). It calculated distance and time using Finland's road network graph, where the vehicle velocity was varied based on different types of road in the network. The truck capacity was specified to identify how many trips were required to collect all the bale wrap waste. The information concerning distance and times was used to calculate the cost and environmental impact in the collection stage.

3.3. Goal and scope definition

ELCC aims to quantify the environmental and economic impacts of collection and recycling bale wrap waste and assess the contribution of each process on the total impacts. The FU is 1 ton recycled granulates (rPE) with a waste composition consisting of 100% LLDPE. In both scenarios, the bale wrap waste underwent the same recycling process. Information about the recycling process and its auxiliary inputs was gathered from personal communication with the recycling company and its environmental permit (Ympäristö- ja terveyslautakunta, 2019). The difference between S1 and S2 was in the frequency of collection, which affects the annual distance, collection time, and the solid contaminants in the bale wrap waste. It was assumed that S1, where the collection was arranged once a year, had more solid contaminants due to the longer storage time. Solid contaminants were assumed to be a mix of garden type waste such as wood or fiber and contaminants from other plastic types. The rate of solid contamination was estimated based on the manual sorting efficiency (Pressley et al., 2015; WRAP, 2009b) since there was a lack of knowledge on solid contaminants in bale wrap waste.

It was estimated that S1 and S2 have 7% and 3% solid contaminants, respectively. These contaminants would be removed by manual sorting in the recycling facility. Automatic sorting was not necessary since the plastic is source-separated (Briassoulis et al., 2013). The plastic went through size reduction, separation, washing, dewatering, drying, extrusion and pelletizing (Fig. 3). The material loss occurred during the recycling process due to shredding, separation, washing and extrusion. In both scenarios, the loss was assumed to be 15% of the weight after being manually sorted (Shonfield, 2008). Used lubricating oil, material loss, and waste from manual sorting were treated in an incinerator with energy recovery.

The impurity was estimated to be around 6% of the weight after being manually sorted (Briassoulis et al., 2012). It comprised dirt, soil or other organic material attached to the plastic film and was removed by the washing process, resulting in wastewater. The wastewater was treated in an on-site wastewater treatment plant (WWTP). The treated water was recirculated along with additional water (1.6 m^3/h) to replace water loss during evaporation in thermal drying, whereas the sludge was transported into a nearby composting site (Ympäristö- ja terveyslautakunta, 2019). Product substitution, including rPE, electricity, heat, and compost, would benefit both environmentally and economically.

The assessment considered the collection and recycling of bale wrap

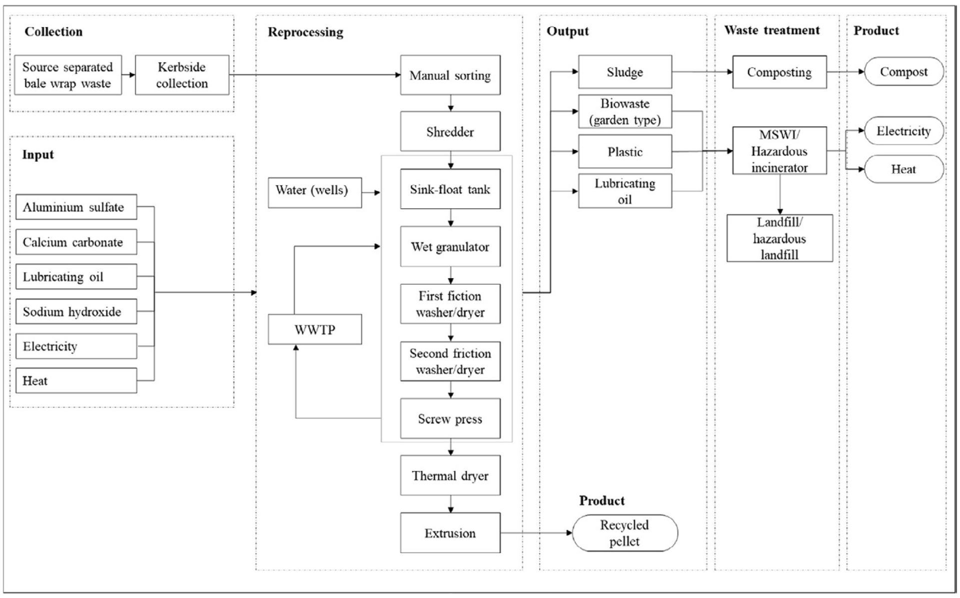

Fig. 3. System boundaries of bale wrap waste recycling.

waste (see Fig. 3), which resulted in system boundaries of: 1) bale wrap waste collection, 2) acquisition of auxiliaries and energy (input), 3) recycling process, 4) the waste treatment of all waste generated during recycling, and 5) product substitution.

3.4. Life cycle inventory

One of the key phases in the LCA is collecting all input and output data related to the process. This study gathered various primary and secondary data within the Finnish context. When the information was not available, the European context was applied.

3.4.1. Operational data

Inventory data concerning farm location and fuel consumption were provided by a company that treats animal by-products. Meanwhile, information about the technology and processes in the recycling plant was obtained from the recycling company. The remaining operational parameters such as material loss, contaminant removal, impurity, material, and energy consumption were gathered from the literature and Ecoinvent v.3 database. A summary of the annual input and output of bale wrap recycling and the operating parameters is shown in Table 1. Detailed information concerning the input and output data is presented in the supplementary material (see Table 1).

The market substitution factor for recycled granulates of LLDPE was 0.47–0.62, and it was derived from the price of recycled material (Plasteurope, 2020a, 2020b). The median value of the market substitution factor was used in this study to quantify the environmental benefit obtained from the avoided production of virgin PE. The environmental benefit was a multiplication product of the market substitution factor and environmental impact of virgin PE production. A market substitution factor was used because recycled material could not completely substitute virgin material (Rigamonti et al., 2014).

The marginal electricity was coal because it had the lowest operational cost and could respond to changes in demand (Mathiesen et al., 2009). It was argued that coal and gas would be marginal electricity in Nordic countries until 2050 (Dotzauer, 2009). Woodchip was the marginal for heat because of the current and future trend of heating in Finland (Finnish Energy, 2019) as well as unconstrained sources and technologies.

3.4.2. Cost inventory

Cost inventory data was obtained from scientific studies and reports. It was expressed in monetary value per FU (€/ton-rPE) using 2018 as the reference year. Economic and physical parameters were used to calculate the cost. The economic parameter presents the monetary value of an item (e.g. 0.07 €/kWh), whereas the physical parameter describes the quantity of items required to perform an activity under the system boundary (e.g. 593 kWh/ton-input). The cost structure in ELCC consists of budget cost and transfer cost, as described by Martinez-Sanchez et al. (2015).

Budget costs were annualized to net present value based on plant capacity of 19,000 ton/year (Ympäristö- ja terveyslautakunta, 2019) with 5% discount rate and 15 year discount period. They included capital cost, fixed operation and maintenance cost, variable cost related to operation and maintenance, and the sale of products generated from the treatment process (rPE, heat, electricity, compost). The equipment costs were normalized based on the usage rate (Martinez-Sanchez et al., 2015). The price of recycled LLDPE granulates ranged between 0.6 and 0.8 €/kg and was used as the basis to calculate the market substitution factor (Plasteurope, 2020a, 2020b). When the costs of equipment were known, but the capacity differed from the required one, an adjustment could be performed using equation (1) (Serna, 2018):

$$\frac{C_1}{C_2} = \left(\frac{Cap_1}{Cap_2}\right)^{0.6} \tag{1}$$

Table 1
Summary of input, output and operational parameters of bale wrap recycling.

Items	S1 Value	S1 Unit	S2 Value	S2 Unit	Note
Collection-input					
• Bale wrap waste	137.64	ton	137.64	ton	
• Diesel	0.35	l/km	0.35	l/km	
• Distance	5731.5	km	8492.4	km	
Collection-output					
• Bale wrap waste	137.64	ton	137.64	ton	
Recycling-input					
• Electricity	539.57	kWh/ton-input	539.57	kWh/ton-input	
• Heat	300	kWh/ton-input	300	kWh/ton-input	
• Water	12.5	ton/ton-input	12.5	ton/ton-input	
• Sodium hydroxide	0.00032	ton/ton-input	0.00032	ton/ton-input	
• Aluminum sulfate	0.00032	ton/ton-input	0.00032	ton/ton-input	
• Lubricating oil	0.000005	ton/ton-input	0.000005	ton/ton-input	
• Calcium carbonate	0.053	ton/ton-input	0.053	ton/ton-input	
Recycling-output					
• rPE	1	ton	1	ton	
Operational parameters					
• Manual removal	7	%	3	%	Percentage of waste collected.
• Material loss	15	%	15	%	Percentage of waste after being sorted manually.
• Impurity	6	%	6	%	Percentage of waste after being sorted manually.
• Market substitution factor	54.5	%	54.5	%	
• Heat efficiency (incineration)	63	%	63	%	
• Electricity efficiency (incineration)	18	%	18	%	

where C_1 and C_2 are the cost of first and second equipment, respectively, whereas Cap_1 and Cap_2 are the capacity of first and second equipment, respectively. Furthermore, Marshall and Swift index was applied to adjust the costs into the reference year 2018 using equation (2):

$$\frac{P_1}{P_2} = \left(\frac{I_1}{I_2}\right) \tag{2}$$

where $P_{1,}$ and $P_{2,}$ show the calculated price for the year 2018 and the original price, respectively, whilst I_1 and I_2 are the index for the year 2018 and the original year, respectively.

Transfer cost can be defined as income distribution among different actors without resource allocation, typically in subsidies or taxes (Martinez-Sanchez et al., 2015). Transfer cost in this study consisted of landfill tax, labor tax, energy tax (applied to natural gas, diesel and electricity) and CO_2 tax (applied to diesel and natural gas). Taxes concerning company operations are commonly excluded, mainly due to the difficulties in calculating them since they depend on various factors and principles (Møller and Martinsen, 2013).

3.5. Assessment method

A consequential approach was applied to reflect the change in cost and emission resulting from modification in bale wrap recycling practice. Hence, the allocation was avoided by product substitution and system expansion (Gala et al., 2015). The modelling was carried out using OpenLCA software ("OpenLCA," 2007). Contribution analysis was applied to identify the relative contribution of each key process to the total environmental and economic impacts. The key processes included collection, reprocessing, electricity substitution, heat substitution, compost substitution, PE substitution, incineration, composting, and transport. Transportation was divided into two key processes, namely collection and transport. The collection is defined as gathering bale wrap waste from farmers to take to the recycling plant, whereas transport indicates the transfer of waste generated into the waste treatment facility and auxiliary input to the recycling plant.

The environmental assessment followed the ReCiPe midpoint (hierarchist) (RIVM, 2016). There are 18 impact categories generated by ReCiPe midpoint (H); however, we focused on six impacts that were considered significant in plastic recycling and APW. These impacts were global warming potential (GWP), fossil resource scarcity (FS), human carcinogenic toxicity (HT-C), human non-carcinogenic toxicity (HT-NC), terrestrial acidification (TA), and water consumption (WC). The use of fossil fuel as raw material in plastic production shows the importance of addressing GWP and FS, whereas WC is a concern because agricultural activity and APW recycling require a high quantity of water. Cascone et al. (2020) reported that water consumption, fossil resource, and climate change are the primary agenda for the impact reduction under EU agricultural policy. Furthermore, GWP, FS, and TA were commonly assessed in plastic recycling, implying the importance of these impacts (Lazarevic et al., 2010; Meys et al., 2020). We added HT-C and HT-NC since these impacts are related to the effect of toxic compounds on the human environment. HT can also be useful as the initial phase of risk assessment when the full assessment is costly and the full data set is not available (Chen et al., 2017; Hertwich et al., 2006).

Normalization is an optional step in the LCA, and its application is related to the goal and scope of a study. Normalization is employed to evaluate the relative magnitude among the impacts within a study or to compare the impacts with a reference situation (e.g. total impacts in a particular region) (Baumann and Tillman, 2004; Pizzol et al., 2017). This work quantified the environmental impacts and compared different scenarios without focusing on the contrast of relative magnitude within the study or reference situation. Therefore, the results were interpreted without normalization since the application would not provide added value in this context.

The economic assessment evaluated the cost associated with producing 1 ton of rPE and its potential change when the modification in recycling occurred. The monetary flow between actors involved in bale wrap recycling (e.g. farmers, collection company, recycling company) was not identified. Total cost was calculated by summing up the cost of collection, reprocessing, incineration, composting and transport, and subtracting by electricity substitution, heat substitution, compost substitution and PE substitution. Hence, a negative result in economic assessment, as with environmental assessment, indicates a benefit.

3.6. Sensitivity and scenario analysis

Scenario and sensitivity analysis identify how the model behaves due to the uncertainty of the input in both the foreground and background systems. Faraca et al. (2019) reasoned that uncertainty in the foreground system could be addressed using sensitivity analysis, whereas uncertainty in the background system could be handled by scenario analysis.

3.6.1. Sensitivity analysis

Global sensitivity analysis was applied to identify how the outputs differed because of the change in the inputs (Bisinella et al., 2016). It consists of a contribution analysis, perturbation analysis and quantification of sensitivity coefficients. In perturbation analysis, each parameter is increased by 10% while maintaining all other parameters fixed at their original value. It is followed by calculation of sensitivity ratio (SR) and sensitivity coefficient (SC) for each parameter using equations (3) and (4):

$$SR_i^j = \frac{\left(\frac{\Delta\ result}{initial\ result}\right)^j}{\left(\frac{\Delta\ parameter}{initial\ parameter}\right)_j} \approx \frac{\partial z_j}{\partial x_i}\frac{x_i}{z_j} \tag{3}$$

$$SC_i^j = \frac{(\Delta\ result)^j}{(\Delta\ parameter)_j} \approx \frac{\partial z_j}{\partial x_i} \tag{4}$$

where $i = 1, \ldots, n$ are tested parameters and $j = 1, \ldots, m$ are the impact categories. SR shows the model's sensitivity related to each parameter, and SC is used to determine the contribution of every parameter to the total variance (Clavreul et al., 2012). The analytical uncertainty of each parameter i associated with impact category j is calculated by equation (5):

$$V_i = V(Y)_i^j = \left(SC_i^j\right)^2 . V_{input}(X_i) \tag{5}$$

with V_{input} describing the initial uncertainty related to parameter X_i. Accordingly, the relative contribution of the uncertainty in X_i to the total parametrical variance is shown by equation (6):

$$S_i = \frac{V_i}{V(Y)} \approx \frac{\left(SC_i^j\right)^2 . V_{input}(X_i)}{\sum_{i=1}^{n}\left[\left(SC_i^j\right)^2 . V_{input}(X_i)\right]} \tag{6}$$

with S_i index being used to sort individual parameters according to their prominence for the result (Bisinella et al., 2016; Faraca et al., 2019).

3.6.2. Scenario analysis

The robustness of the LCA related to its background system is tested by scenario analysis (Rigamonti et al., 2014). We varied the type of marginal energy and fuel type for the collection and transport. The initial marginal electricity and heat were coal and woodchip, respectively. They were modified into natural gas in the scenario analysis. The use of diesel for collection and transport was modified into LNG in the scenario analysis.

4. Results

4.1. Vehicle routing problem

Vehicle routing in S1 required 15 trips to collect all bale wrap waste, whereas S2 needed 9 trips in each collection, resulting in 18 trips annually (Fig. 4). The total distances for S1 and S2 were 5731.5 km and 8492.4 km, respectively, which translated into annual diesel consumption for S1 and S2 of around 2006 L and 2972 L, respectively. The collection times, including travel time and loading time (20 min per farm), were 159.72 h and 274.56 h for S1 and S2, respectively.

4.2. Environmental assessment

Fig. 5 presents the environmental impacts of six impact categories based on the relative contribution of individual key processes, with the net result as the sum of various impacts and benefits. The impacts can be seen as benefits when represented in negative values. The environmental benefit from recycling was primarily acquired from avoided environmental impact due to the substitution of virgin plastic with

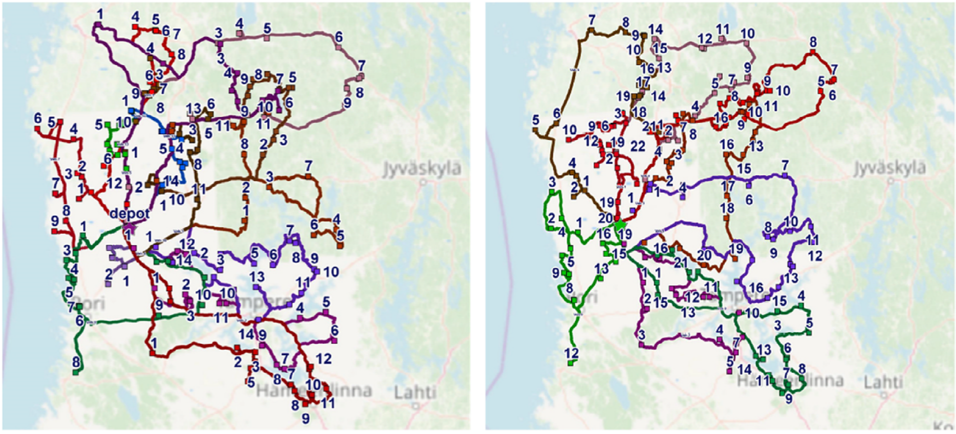

Fig. 4. Vehicle routing of bale wrap collection for S1 (left) and S2 (right).

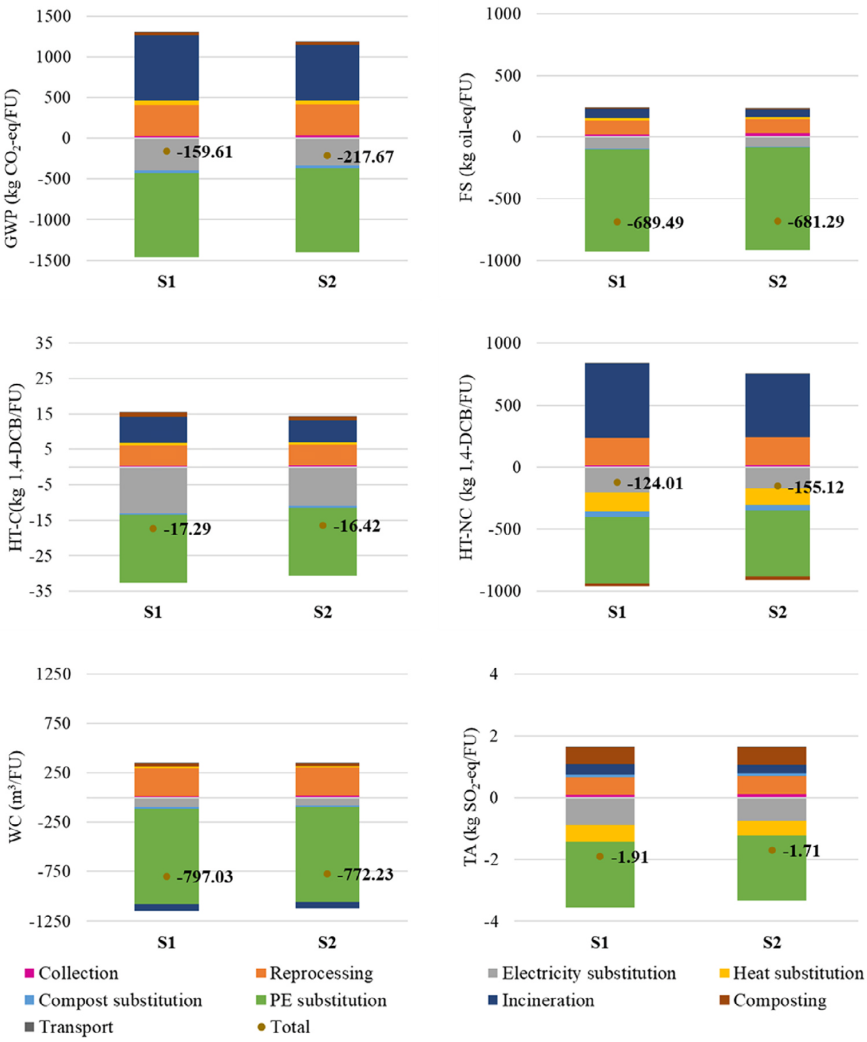

Fig. 5. Environmental impacts of bale wrap recycling based on its key-process for S1 and S2. GWP: global warming potential, FS: fossil resource scarcity, HT-C: human carcinogenic toxicity, HT-NC: human non-carcinogenic toxicity, TA: terrestrial acidification, WC: water consumption.

recycled material. Moreover, the electricity and heat recovered from incineration as well as compost from composting the wastewater sludge also offered environmental benefits. On the other hand, impact on the environment occurred during collection, transportation, plastic reprocessing, incineration, and composting.

Among these processes, incineration caused the highest impact. Hence, minimizing plastic waste that goes into incineration becomes important. It can be achieved through an advanced recycling system that can sort material well and minimize material loss during reprocessing (Faraca et al., 2019) and a source-separated waste system. Electricity and heat substitution are environmental savings that are expected from incineration. In this study, electricity substitution provided benefit across all impact categories; meanwhile, heat substitution provided benefit only in TA and HT-NC. The outcome from energy recovery in the consequential LCA depends on the marginal energy source (Faraca et al., 2019; Mathiesen et al., 2009; Rigamonti et al., 2014). In this study, a positive value from heat substitution indicated that the heat generated from plastic incineration performed worse than the marginal generated from woodchip.

The collection phase was the key difference between S1 and S2. Two collections within a year in S2 caused an increased traveled distance of 48% compared with S1. The results showed that collection was not one of the main contributors to the environmental impacts since it contributed only 0.74% and 1.12% of the total GWP in S1 and S2. As for other impact categories, collection contributed around 0.7–2.7%. This trend was confirmed by Cascone et al. (2020) and Larsen et al. (2010), who showed that the contribution of the collection phase in the recycling system was less than 5% of the overall impact.

The overall results showed net environmental benefit in each impact category for both scenarios. S1 performed better in FS, HT-C, WC, TA, whereas S2 provided more benefit for GWP and HT-NC. The similarity between S1 and S2 was that PE substitution provided the highest environmental credit in all impact categories, making it the key in closing the loop by avoiding virgin material production.

4.2.1. Global warming potential (GWP)

GWP is used to measure greenhouse gas potential in trapping heat in the atmosphere relative to CO2, expressed in kg CO_2-eq (Huijbregts, 2016). The larger the value of GWP, the higher its ability to trap the heat. The overall GWP of S1 and S2 per FU were −159.61 kg CO_2-eq and −217.67 kg CO_2-eq, respectively, and it indicated that S2 provided about 36% more benefit than S1. The highest contribution to GWP was incineration, as the results showed the values of 803.86 kg CO_2-eq/FU and 684.66 kg CO_2-eq/FU in S1 and S2, respectively. As for the collection, the difference was not significant as the results for S1 and S2 per FU were 20.47 kg CO_2-eq and 29.08 kg CO_2-eq, respectively.

The net benefit was higher in S2 than S1, although the travel distance for the collection was longer in S2. Fewer collection frequency in S1 was assumed to accumulate higher solid contaminants (e.g., other types of plastic and garden waste) and have more plastic film unintentionally mixed with other municipal waste streams, causing a lower collection rate. These situations lead to a lower quantity of plastic going into recycling and more materials are incinerated, resulting in more CO_2 emissions in S1.

4.2.2. Fossil resource scarcity (FS)

This impact category refers to the depletion of fossil resources. It is determined as the energy content ratio between a particular fossil resource and crude oil, expressed in kg oil-eq (Huijbregts, 2016). Recycling bale wrap can avoid the production of virgin LLDPE that is mainly derived from fossil fuel. For FS, processing contributed to the highest environmental impact, followed by incineration. While electricity substitution provided benefits, the heat caused an impact. It indicated that from an FS perspective, heat derived from woodchip was more sustainable than WtE. The overall performance of both scenarios was not significantly different, where S1 showed around 1.2% more benefit than S2. This benefit was obtained from electricity substitution, which was higher in S1 compared to S2. The marginal electricity that was sourced from coal made WtE a more sustainable choice.

4.2.3. Human carcinogenic toxicity (HT-C)

Human toxicity potential indicates the impact on humans caused by toxic substances released into the environment. The toxicity potentials are quantified considering the toxicity's fate, exposure, intake, and effect (Baumann and Tillman, 2004). Calculating human toxicity potential in LCA is complex because people respond differently to chemical exposure, and the causation effect may be poorly understood (Shonfield, 2008). Human toxicity potentials are categorized into carcinogenic and non-carcinogenic and expressed as 1,4-dichlorobenzene equivalent (1, 4-DCB) (Huijbregts, 2016). Incineration gave the highest contribution to HT-C of about 7.34 kg 1,4-DCB/FU and 6.22 kg 1,4-DCB/FU for S1 and S2, respectively. S1 performed about 5% better than S2 in HT-C due to the higher electricity substitution which replaced marginal electricity derived from coal. The spoil from coal mining is a major contributor to the emission of a carcinogenic toxic substance such as chromium-VI into the water.

4.2.4. Human non-carcinogenic toxicity (HT-NC)

For HT-NC, S2 provided 25% more benefit compared with S1. Other than PE substitution, the environmental savings were obtained from electricity and heat substitution, which provided total benefit of about −358.81 kg 1,4-DCB/FU and −303.67 kg 1,4-DCB/FU for S1 and S2, respectively. Nevertheless, the direct impact from incineration in S1 counterbalanced the benefit derived from energy substitution, making the overall HT-NC in S2 better than S1. Incineration directly emits ionic zinc as the main cause of HT-NC.

4.2.5. Water consumption (WC)

Water consumption (m^3) implies water use incorporated into the products or losses through evaporation, discharge, and transfer into other water bodies (Huijbregts, 2016). Plastic recycling requires a large amount of water for washing, especially in agricultural plastics, where certain types of plastic can contain a high level of impurities (Briassoulis et al., 2012). In both scenarios, 282.9 m^3 of water was needed to produce 1 ton of recycled PE. Nevertheless, recycling benefits WC compared with virgin material production by avoiding water consumption of about −960.89 m^3/FU. Between the scenarios, S1 showed better performance than S2 by slightly more than 2% in WC. Total plastic being recycled in S1 was less than S2, causing lower water consumption.

4.2.6. Terrestrial acidification (TA)

TA (kg SO_2-eq) reflects the maximum potential to acidify soil relative to SO_2 (Baumann and Tillman, 2004). This study showed TA per FU of −1.91 kg SO_2-eq and −1.71 kg SO_2-eq for S1 and S2, respectively. Reprocessing contributed the highest impact, with about 0.579 kg SO_2-eq/FU in S1 and S2, followed by composting, with 0.562 kg SO_2-eq/FU in both scenarios. S1 provided a higher benefit of about 11% compared with S2. S1 could perform better because it generated higher electricity substitution from the incineration process. The marginal electricity in this study was coal, known as the primary source of sulfur dioxide, which contributes to acid rain formation and affects the terrestrial ecosystem.

4.3. Economic assessment

The result of the economic assessment is expressed in €/FU. Fig. 6 presents the total cost per FU based on the contribution of individual key processes. Both scenarios provided overall economic benefit (indicated by negative value), although S1 was a more profitable scenario than S2. Transfer costs in S1 and S2 were almost identical, showing results of around 100.98 €/FU and 100.68 €/FU, respectively, whilst budget costs were −265 €/FU and −237.92 €/FU for S1 and S2, respectively.

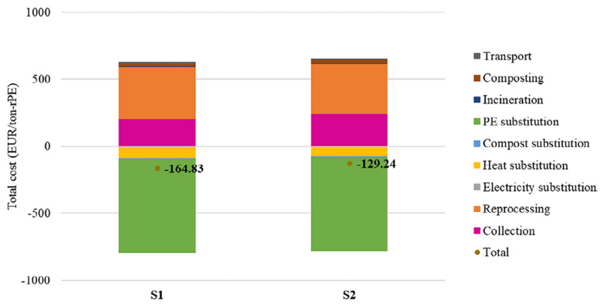

Fig. 6. Cost of bale wrap recycling based on its key processes for S1 and S2.

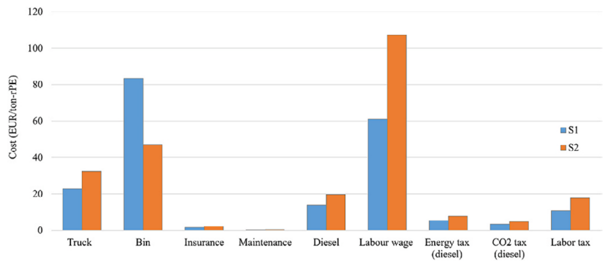

Fig. 8. The cost breakdown in the collection stage for S1 and S2.

Contribution analysis in Fig. 6 shows three major key processes in S1 and S2, namely PE substitution, reprocessing, and collection. In both scenarios, PE substitution generated financial savings of about 88–89% of total revenue. Reprocessing costs in S1 and S2 were 388.55 €/FU and 378.22 €/FU, while collection costs were 202.61 €/FU and 239.24 €/FU, respectively. In contrast to the environmental assessment, collection was one of the most crucial key processes in terms of economic impact.

Fig. 7 displays the costs incurred based on their sequence along the recycling chain. The collection process was divided into farm and bale wrap transport. The latter incurred the highest expense in recycling, particularly in S2. A collection might have an insignificant contribution in the instance of a recycling center (Faraca et al., 2019); however, in the case of the curbside collection of recyclables, the financial cost can reach 300 €/ton (Groot et al., 2014). Given the significance of the collection stage, we further analyzed its cost itemization (Fig. 8). Since the curbside scheme was applied, specific bins were needed to have a source-separated waste system. The costs of bins and labor wages dominated the expenses in the collection phase. The costs of bins can contribute significantly to curbside collection (e.g., Edwards et al., 2018). S1 displayed a 30% higher cost of bins than S2 because the annual collection requires farmers to provide more bins to store a larger quantity of waste. In contrast, the labor wage of S2 was 139% higher than S1 due to the more frequent collection.

4.4. Sensitivity and scenario analysis

4.4.1. Sensitivity analysis

Sensitivity ratio (SR) was used to express the model's sensitivity related to each input parameter. Perturbation analysis was applied by increasing the value of the input parameter by 10% one at a time; hence, if the SR value equals 2, a 10% increase in that parameter will result in a 20% increase in the model's result. Sensitivity analysis was conducted using 13 and 45 individual parameters for environmental and economic assessment, respectively. Fig. 9 shows the SR results for GWP and economic assessment in both scenarios. All parameters for environmental

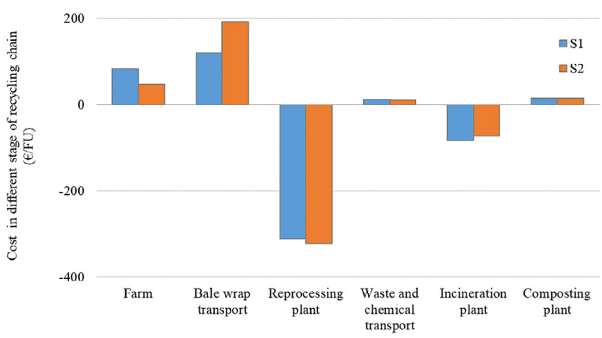

Fig. 7. Costs incurred in different stages of the recycling chain.

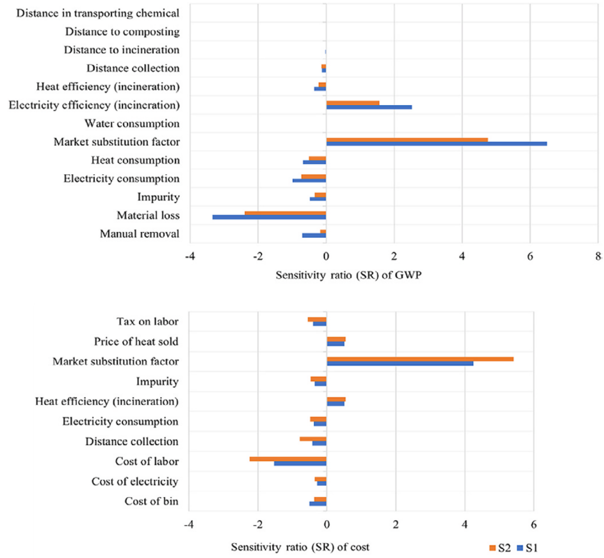

Fig. 9. Sensitivity ratio (SR) of global warming potential (GWP) and costs for S1 and S2.

assessment were presented, whilst the 10 most sensitive parameters in the economic assessment that were overlapping in both scenarios were shown (see supplementary material in Table 4 and Fig. 1 for SR in all impact categories).

Similar behavior was found for environmental and economic assessment in S1 and S2, although the magnitude of sensitivity was different. For instance, SR for GWP showed that market substitution factor, material loss and electricity efficiency in incineration were the three most sensitive parameters in both scenarios. However, the sensitivity was higher in S1. The market substitution factor and labor cost were the most sensitive parameters in both scenarios for the economic analysis, although S2 showed more sensitivity than S1. For other impact categories, the market substitution factor was also the most sensitive parameter.

Following the perturbation analysis, the SC value was used to rank the relative contribution of each parameter to the total variance of each impact category, as illustrated in Fig. 10. The y axis displays the percentage of total variance related to the number of parameters included in the calculation. Overall, three parameters were sufficient to achieve 90% of the total variance in economic and environmental assessment except for TA, which required four parameters.

Fig. 11 summarizes the three most crucial parameters and their associated contribution to the overall variance. The value indicates the share of variance covered by the related parameter. A similar pattern was found in S1 and S2, except for TA. Electricity efficiency in incineration, which was one of the highest contributors to the total variance

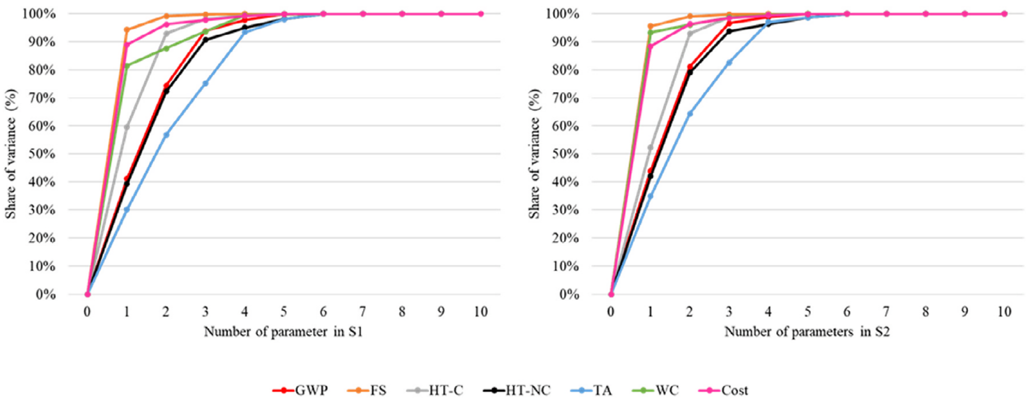

Fig. 10. Share of uncertainty contribution analysis for S1 and S2.

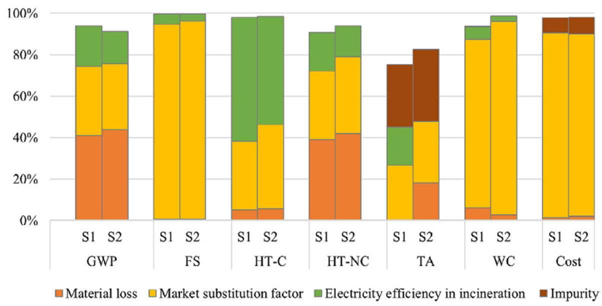

Fig. 11. Ranking of the three most important parameters associated with their percentages for S1 and S2.

of TA in S1, was not found in S2. Meanwhile, material loss was found only in S2.

4.4.2. *Scenario analysis*

Scenario analysis presented the model behavior related to its background system. Fig. 12 presents the scenario analysis results by changing the fuel to LNG and marginal energy source to natural gas. The orange dot and black square indicate the results obtained from scenario analysis; meanwhile, the blue bar shows the baseline. Shifting the fuel provided insignificant savings in both environmental and economic assessment. The improvement for all categories in both scenarios ranged from 0.9 to 7%.

In contrast, shifting marginal energy sources brought considerable change across impact categories except for WC. Trade-offs among the impact categories were also observed. Improvements were obtained in GWP and FS; meanwhile, HT-C, HT-NC, and TA deteriorated. For example, in S1, 242% improvement compared with the baseline was found for GWP; conversely, TA showed a 274% decline compared with the baseline scenario. The change in results was mainly caused by the shift of marginal heat from woodchip to natural gas. Using woodchip as marginal heat generated environmental impacts (positive result) for GWP, HT-C, FS and WC, and environmental savings only for TA and HT-NC. This implies that the environmental benefit from energy substitution in incineration is relative to the marginal energy source. For economic assessment, the change in marginal energy source did not affect the result due to the assumption that the marginal energy source did not affect the energy price.

5. Discussion

The discussion will focus on GWP and cost assessment due to their importance. Moreover, previous research commonly investigated GWP so that a comparison across studies is possible.

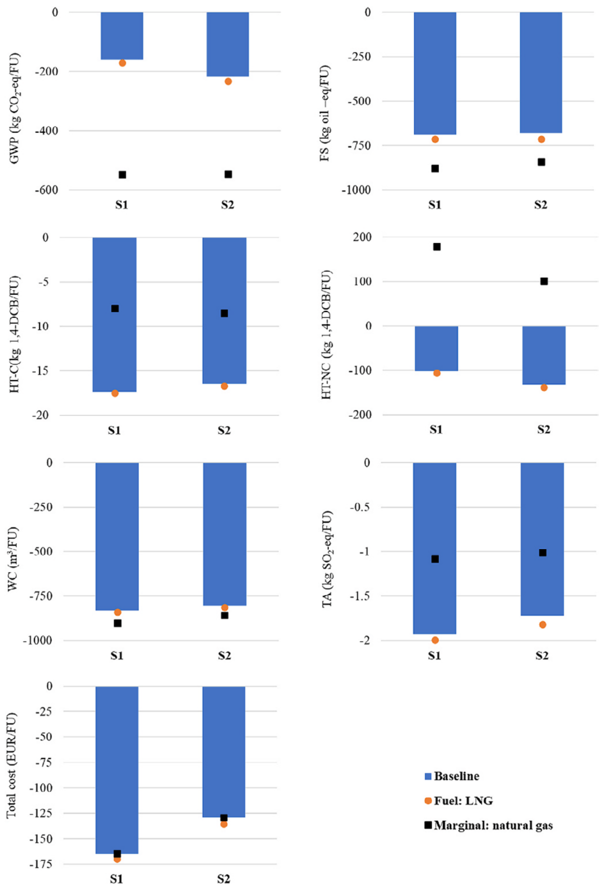

Fig. 12. Results of scenario analysis by changing fuel type and marginal energy for S1 and S2.

5.1. *Overall result*

Total environmental savings for GWP in both scenarios ranged from around −160 kg CO_2-eq to −217 kg CO_2-eq per ton rPE, indicating much less savings compared with studies on plastic recycling as performed by Faraca et al. (2019), Rigamonti et al. (2014) and Shonfield (2008). Even when the FU in this research was adjusted from ton rPE to ton plastic waste to match these studies, the environmental saving was still 50% lower. Previous studies showed that recycling PP, PE, PET, PS, and PVC would avoid GWP to around 500–700 kg CO_2.eq per ton of plastic waste. The difference was heavily affected by operational data. Faraca et al. (2019) and Rigamonti et al. (2014) used a higher market substitution value, resulting in higher avoided virgin material production. Specifically for PE in an advanced mechanical recycling scenario, Faraca et al. (2019) applied a value of 91% for market substitution, which contributed to higher environmental saving compared with the value of 54.5% in this study. The difference of operational data across various studies was caused not only by the types of plastic but also by the source of plastic waste and its impurity. Recycled material derived from a single polymer with little organic impurity can replace virgin material with almost a 1:1 ratio (Lazarevic et al., 2010). Even though agricultural plastic waste is a good input for recycling due to its limited resin type, it is commonly impure due to organic contaminants (Briassoulis et al., 2012). This study applied a 54.5% market substitution factor based on the selling price of recycled LLDPE in the market. LLDPE is the material used in agricultural applications, and the value chosen in this study showed similarity with Gu et al. (2017), who used 50% as the substitution factor for recycled material derived from agricultural plastic.

We then compared the environmental consequences between recycling and landfilling, as most of the APW is still disposed of in landfill. The impact characterizations of the landfill are acquired from Ecoinvent v.3 database. Across all impact categories, recycling showed better environmental performance. GWP, FS, HT-C, WC and TA in recycling offered superior environmental performance, ranging from 2 to 2.3 times better than landfill. The highest environmental saving was from HT-NC, where recycling performed 17.4 and 19 times better than landfill for S2 and S1. Similar patterns were reported by Hou et al. (2018), who compared recycling and landfilling of post-consumer plastic film. Landfill emits ionic zinc that seeps into the water as a major contributor to HT-NC. Consequently, proper treatment and diversion from landfills become important. Policy instruments such as landfill tax or landfill ban play an important role in waste diversion, especially considering the low cost of landfilling, which is about 23 €/ton excluding tax (WRAP, 2018).

In the economic assessment, the financial saving was around −165 and −129 €/ton-rPE for S1 and S2, respectively. Faraca et al. (2019) showed the economic benefit of −90 €/ton plastic waste for an advanced recycling scenario, in which a 50% contribution was derived from avoided virgin material. Similarity was found in this study, where avoided virgin material contributed about 48–49% to the total cost. Our results provided more financial savings, even though we included the collection phase, and the value of the market substitution factor and electricity efficiency were lower. This could be caused by applying a discount rate and discount period of 5% for 15 years, whereas Faraca et al. (2019) did not apply the discounting of future cost and benefit.

5.2. Influence of process parameter and assessment methods

The parameters, FU, boundaries, and methods affect the outcome of LCA and LCC. We used average conditions and a common method to compare and assess the results across studies to accommodate this. The results from perturbation analysis showed that LCA and LCC are more sensitive to a few parameters such as market substitution factor, material loss, and cost of labor. By knowing this information, all actors in the recycling chain know how to anticipate any disruption or improve the process by concentrating on a few parameters.

The market price of recycled material was the basis for determining the market substitution factor, which will affect the environmental benefit of the recycled plastic. Hence, the price of recycled plastic was not directly tied to the price of virgin material. This study showed that recycling could provide financial savings if the price of the recycled material is higher than 0.535 €/kg and 0.570 €/kg for S1 and S2, respectively. Different factors affect the price of recycled material, including the loss of quality during reprocessing, difference properties between virgin and recycled material, market acceptance of recycled material, and public pressure to incorporate a minimum amount of recycled material in products (Gu et al., 2017; Holmvik et al., 2019; Rigamonti et al., 2014). Faraca et al. (2019) determined the market substitution factor from the literature to calculate the price of recycled material (the product of market substitution factor times the price of virgin material). It was argued that the increase or decrease of recycled material follows the trend of virgin material. However, it is not always the case because the mismatch of supply and demand of recycled material has driven the price of recycled material higher than virgin products (Holmvik et al., 2019). This issue becomes especially important if the government plans to impose a minimum amount of recycled plastic in new products. The policy should guarantee that the demand for recycled material should not exceed the current capacity to produce it.

Scenario analysis provided information on the interaction between the model and the background system. Fig. 12 depicts the total environmental and economic impacts caused by the change of marginal energy and fuel. Although shifting to LNG showed improvement in all categories, it was not significant. This implied that fuel type was not crucial in this study and might not encourage change in the use of diesel as is an established practice. However, the marginal energy source modification showed significant change in LCA results involving trade-offs across different categories. Even within individual impact categories, a different trend was found in heat and electricity substitution. Shifting from woodchip to natural gas as marginal heat created a remarkable improvement of about 87% in GWP; however, the shift from coal to natural gas as marginal electricity worsened the saving from electricity substitution by around 14%. This indicates that the benefit of energy recovery from incinerators depends relatively on how sustainable the existing marginal energy source is.

5.3. Shortcomings

The primary shortcoming in this study was its reliance on secondary data for most of the processes. Data uncertainty was also seen as a limitation. There was a lack of research on non-packaging agricultural plastic waste, especially focusing on bale wrap, as shown by previous studies on greenhouse plastic (e.g., Briassoulis et al., 2013; Cascone et al., 2020; Gu et al., 2017). Hence, the data used in this study was adopted from the recycling of other types of plastic (e.g. post-consumer plastic film, greenhouse covering, etc.). Moreover, this study was part of the planning phase for bale wrap recycling so that the real-world applications and challenges in the recycling of bale wrap were still unknown. For example, the stretch and clinging characteristics of the wrap may cause the plastic to curl during the process (Briassoulis et al., 2013), or the quality of recycled pellets from bale wrap is unclear.

To overcome the shortcoming in data uncertainty, using distribution instead of single numbers is recommended. However, as shown by this study, in the case of information about distribution being unavailable, the use of a single number accompanied by sensitivity analysis and uncertainty contribution analysis can be applied. It will provide information about the source of uncertainty and the most sensitive parameters. Consequently, more attention can be paid to the most crucial parameters when decision-making is required or a future study is conducted.

5.4. Managerial implications and policy recommendations

Moving from current practice - where there is no clear and unified guideline in handling non-packaging agricultural plastic waste - to establishing a recycling scheme will require change that involves many actors. Through the recycling process, starting from the collection phase to the production of recycled pellets, this practice will greatly affect the actors in the collection phase. The sorting and reprocessing phases are established already. They may require a small adjustment which depends on plastic type and condition (e.g. dry or wet granulation, one or two washings, manual or automatic sorting). In contrast, a strategy in the collection phase is crucial to ensure that the financial burden is distributed fairly among the actors so that farmers are willing to participate.

The collection company becomes a key actor in devising a collection strategy (e.g. collection scheme, frequency, fee) that the farmers need to agree on. A financial assessment will play a more important role than an environmental assessment in devising a collection strategy since the results show the significance of collection to the total cost. Although the default plan is applying curbside collection to ensure a high collection rate, the collection company must consider the bring-in scheme as an alternative. The bring-in scheme will require farmers to bring their waste to the reception points, reducing the collection cost due to the shorter collection distance. Collection companies and farmers must agree on cost structuring where pay-as-you-throw (PAYT) or annual membership can be alternatives. The former is a typical cost structuring in waste management where the cost will be based on the quantity of waste, whereas the latter is a fixed cost for a year with unlimited collection quantity.

An agreement must be made between the collection company and the recycling operator concerning the collection frequency. One-time collection requires a longer storing period, which can increase solid contamination and weathering effect. These issues can reduce the quantity that goes into recycling and the quality of the recycled material, although various studies showed inconclusive results regarding weathering effect. A study performed in Finland showed no significant weathering effect (Erälinna and Järvenpää, 2018), whereas the weather played a central role in hotter regions such as Italy or the Middle East (Basfar and Idriss Ali, 2006; La Mantia, 2002; Tuasikal et al., 2014).

Governments can play an important role in APW recycling by implementing extended producer responsibility (EPR). It is especially essential in a country where a national scheme does not exist yet. Farmers will still bear the cost of collection and recycling through the integration of the EoL management fee into the price of the plastics. Nevertheless, there will be coordination and clarity regarding the fee, reception points, collection frequency, and organizations in charge. Furthermore, EPR will require reporting and targets that can improve the transparency and performance of APW recycling. This policy approach can be combined with the regulatory instrument and financial instruments such as landfill ban or landfill tax.

6. Conclusions

The mechanical recycling of plastic waste is not a new technology. However, the emergence of the circular economy that demands closing the loop of material flow increases the urgency of recycling practice. This study evaluates the mechanical recycling of bale wrap waste in the Finnish context using 2018 as a reference year. Two scenarios are constructed based on collection frequency: one collection (S1) and two collections (S2) per year. The analysis covers the cost and environmental assessment as well as sensitivity analysis to assess the impacts and benefits of applying a closed-loop supply chain for bale wrap. These are the conclusions derived from this study:

- The quantification of environmental and economic performance in S1 and S2 show a trade-off between GWP and cost. The trade-off indicates that it is not possible to maximize both environmental savings and economic benefits. The scenario that offers more economic benefits will provide fewer environmental benefits when compared with the other scenario. In this case, S1 provides 27% more economic savings with 36% less GWP savings compared with S2. Hence, decision-makers must prioritize using weighting criteria to achieve the balance between economic and environmental goals.
- The collection contributes little to the environmental impact; however, it is one of the key processes for economic performance. It covers around 32–36% of the total cost for both scenarios, with S2 incurring 18% higher cost than S1. This cost is borne by the farmers, whose willingness to participate will determine the success of bale wrap recycling.
- Material substitution is the primary key process for economic and environmental saving by avoiding virgin material production, whilst the incineration of waste generated in reprocessing causes the highest impact on GWP. It implies the importance of efficient reprocessing, where material loss should be minimized.
- The market substitution factor is the most sensitive parameter for both GWP and financial assessment. It results from the price of recycled material, which is affected by its supply and demand, quality, and acceptability. The highest uncertainty in GWP is generated from material loss, and in financial assessment it is derived from the market substitution factor. The results of sensitivity analysis are particularly important for the actors involved in CLSC and for decision-makers. When actors or decision-makers decide to adjust the recycling process or impose a certain policy, even a small change can significantly affect the output if the action affects the sensitivity parameter.

Future direction can still focus on EoL management by assessing different recycling methods and collection strategies such as feedstock recycling or bring-in collection schemes. Furthermore, future studies can also explore the effect of recycled material on the supply chain and the relationship between suppliers.

Declaration of competing interest

None.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ijpe.2021.108347.

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Multi-objective optimization to improve energy, economic and, environmental life cycle assessment in waste-to-energy plant

Bening Mayanti [a,*], Joel Songok [b], Petri Helo [c]

[a] Vaasa Energy Business Innovation Centre, University of Vaasa, Yliopistonranta 10, 65200 Vaasa, Finland
[b] Electrical Engineering, School of Technology and Innovations, University of Vaasa, Yliopistonranta 10, 65200 Vaasa, Finland
[c] Networked Value Systems, Department of Production, University of Vaasa, P.O. Box 700, FI-65100 Vaasa, Finland

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ABSTRACT

This paper presents a multi-objective optimization (MOO) of waste-to-energy (WtE) to investigate optimized solutions for thermal, economic, and environmental objectives. These objectives are represented by net efficiency, total cost in treating waste, and environmental impact. Integration of the environmental objective is conducted using life cycle assessment (LCA) with endpoint single score method covering direct combustion, reagent production and infrastructure, ash management, and energy recovery. Initial net efficiency of the plant was 16.27% whereas the cost and environmental impacts were 75.63 €/ton-waste and -1.21×10^8 Pt/ton-waste, respectively. A non-dominated sorting genetic algorithm (NSGA-II) is applied to maximize efficiency, minimize cost, and minimize environmental impact. Highest improvement for single objective is about 13.4%, 10.3%, and 14.8% for thermal, economic, and environmental, respectively. These improvements cannot be made at once since the objectives are conflicting. These findings highlight the significance role of decision makers in assigning weight to each objective function to obtain the optimal solution. The study also reveals different influence among decision variable, waste input, and marginal energy sources. Finally, this paper underlines the versatility of using MOO to improve WtE performance regarding the thermal, economic, and environmental aspects without requiring additional investment.

1. Introduction

Unsustainable production and consumption drive an increase in waste generation. Currently, waste-to-energy (WtE) is the most common technology to deal with a variety of municipal waste as well as part of industrial solid waste (Arena and Di Gregorio, 2013; Lausselet et al., 2016). In 2018, Europe treated approximately 70 million ton of municipal solid waste in WtE, showing a 117% increase compared to 1995, and this trend is predicted to rise (Birgen et al., 2021; Eurostat, 2019; Scarlat et al., 2019). Incineration technology in the WtE plant not only is robust, but also can significantly reduce the waste volume that goes to landfill and generate heat and electricity (Arena, 2012; Fruergaard and Astrup, 2011). However, WtE is regarded expensive since the payback period can take about 10–30 years, and the cost in treating waste per ton can range from 53 to 150 € (Assamoi and Lawryshyn, 2012; Fernández-González et al., 2017; Zabaniotou and Giannoulidis, 2002).

To ensure the benefit from WtE, its operation must be optimized to increase energy efficiency so that the electricity or heat obtained from the process can be maximized. In the optimization of thermal power generation, the thermo-economic objectives are combined to maximize energy efficiency and minimize the cost by applying multi-objective optimization (MOO). MOO, which can utilize different algorithms, becomes the main solution to optimize the power generation system. NSGA-II was commonly used to maximize thermal efficiency and minimize the cost of steam cycle, organic Rankine cycle, Kalina cycle in cogeneration plant, and WtE (Behzadi et al., 2018; Hajabdollahi et al., 2012; Hajabdollahi and Fu, 2017; Özahi and Tozlu, 2020). The results showed an increase in thermal efficiency and decrease in the cost rate. Optimization using other types of algorithms, such as genetic diversity evaluation method or modified differential evolution, also showed improvement of thermal efficiency and cost for different types of power generation (Baghernejad and Yaghoubi, 2011; Naserabad et al., 2018; Wang et al., 2014).

* Corresponding author.
E-mail addresses: bening.mayanti@uwasa.fi (B. Mayanti), joel.songok@uwasa.fi (J. Songok), petri.helo@uwasa.fi (P. Helo).

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Nomenclature

APC	air pollution control
C_{el}	electricity price (€/MWh)
C_p	cost of treating the waste (€/ton-waste)
C_{labor}	total annual salaries (€/year)
DFCI	Direct fixed-capital investment
FCI	Fixed-capital investment
FEP	Fossil energy provision
FU	functional unit
h	enthalpy (kJ/kg)
HPT	high pressure turbine
IFCI	Indirect fixed-capital investment
LCA	life cycle assessment
LCI	life cycle inventory
LCIA	life cycle impact assessment
LHV	lower heating value (kJ/kg)
LPT	low pressure turbine
$\dot{m}$	mass flow rate (kg/s)
MNG	maximum number generation
MOO	multi-objective optimization
nGD	normalized generational distance
nSP	normalized spread
NSGA-II	non-dominated sorting genetic algorithm
PEC	purchased-equipment cost
$\dot{Q}$	heat (kW)
r	interest rate
SEP	Sustainable energy provision
SNCR	selective non-catalytic reduction
SSDTC	steady-state detection
SSI	single score impact (Pt/ton-waste)
t_a	annual plant operation (hours)
$\dot{W}$	power (kW)
WPD	weighted percentage deviation factor
WtE	waste-to-energy
y	discount period (years)

Greek

ε_{el}	electric efficiency
η_{pb}	boiler pump isentropic efficiency
η_{pc}	condenser pump isentropic efficiency
$\eta_{T,s}$	turbine isentropic efficiency
χ	vapor quality

Subscripts

i	inlet
o	outlet

However, with growing concern about sustainability, there is still a lack of integration of environmental impact in the optimization problem of power generation. Some of existing studies integrated environmental objective into MOO on power generation as total damage cost (Mahmoodabadi et al., 2015; Sayyaadi, 2009) or CO_2 emission (Ahmadi et al., 2011; Javadi et al., 2019). Few studies applied comprehensive approach by integrating environmental objective through life cycle assessment (LCA). Gerber et al. (2010) and Nguyen et al. (2014) integrated the environmental objective to optimize biomass power generation as well as oil and gas platforms using LCA. Hence, their included a broad range of emissions and impact categories from the product's life cycle to produce comprehensive assessment, prevent burden-shifting, and identify activities that cause the highest impact.

Currently, to the authors' knowledge, there seems to have been no study regarding the integration of the environmental objective using LCA and MOO in the WtE system to evaluate energy, cost, and environmental impact. This creates a gap concerning assessment of the environmental performance of an improved WtE plant. Therefore, this paper presents the study of WtE optimization that considers energy efficiency, cost, and environmental life cycle assessment. The aim is achieved by focusing on several objectives, such as (i) assessing the cost, environmental impact, and energy efficiency of the system, (ii) applying NSGA-II to improve WtE performance taking environmental, thermal, and economic aspects as objective functions, (iii) applying scenario and sensitivity analysis to evaluate the behavior of the model and the influence of each decision variable in the steam cycle operation.

2. Material and methods

2.1. System description

This illustrative case was a scenario built on an actual incinerator with electricity recovery. The information concerning the WtE specification and its operating condition were obtained from a company which operates a small-scale incinerator, then supplemented by Ecoinvent database.

Fig. 1 displays a scheme of the WtE with annual throughput of 36,208 ton-waste. Bottom ash and fly ash are transported to the landfill and hazardous landfill, respectively, without any material recovery. The plant recovers energy in the form of electricity for self-consumption and sale, and heat for self-consumption. Energy recovery that is shown in dashed boxes can avoid conventional production of electricity and heat. The cycle in the center of Fig. 1 are the simplified version of steam cycle consisting of boiler, turbine, feed pump, and condenser. Heat from combusting waste is used by boiler to convert water into steam. Thermal energy in the steam is extracted by turbine to rotate generator and produce electricity. The steam outflow from turbine is then transformed back into water by the condenser and being cycled back to the boiler by using feed pump. More detail process in steam cycle is shown by Fig. 2.

Apparatus 1 and 2 are high-pressure turbine (HPT) and low-pressure turbine (LPT), respectively. Both will extract energy out of steam generated by boiler. However, HPT works for higher pressure steam and LPT is designed to recover exhaust energy from lower pressure steam that comes out of HPT. The symbol 'G' next to HPT and LPT are generators that convert rotary motion into electricity. Apparatus 3 and 8 are principally heat exchanger. The former is a steam condenser that recirculates water (from source 10 to sink 11) to condense the steam into water, and the latter utilizes steam to preheat the air that is used in the combustion process (apparatus 14 and 15 represent source of air and heated air, respectively). Steam (line 7), water (line 12 and 19), and make-up water (line 20) flow to the deaerator (apparatus 5). Deaerator removes dissolve gases from water to prevent corrosion in the system. The steam (line 7) will heat up the water so that the dissolved gases are released and can be vented out. Excess water is drained to sink 12, while the feedwater is being pumped and recirculated to boiler. Line 3, a steam bleed from HPT, has zero flow presently. It is illustrated in Fig. 2 because the WtE operator considers a possibility to reuse the steam. (e.g., supplying to other company).

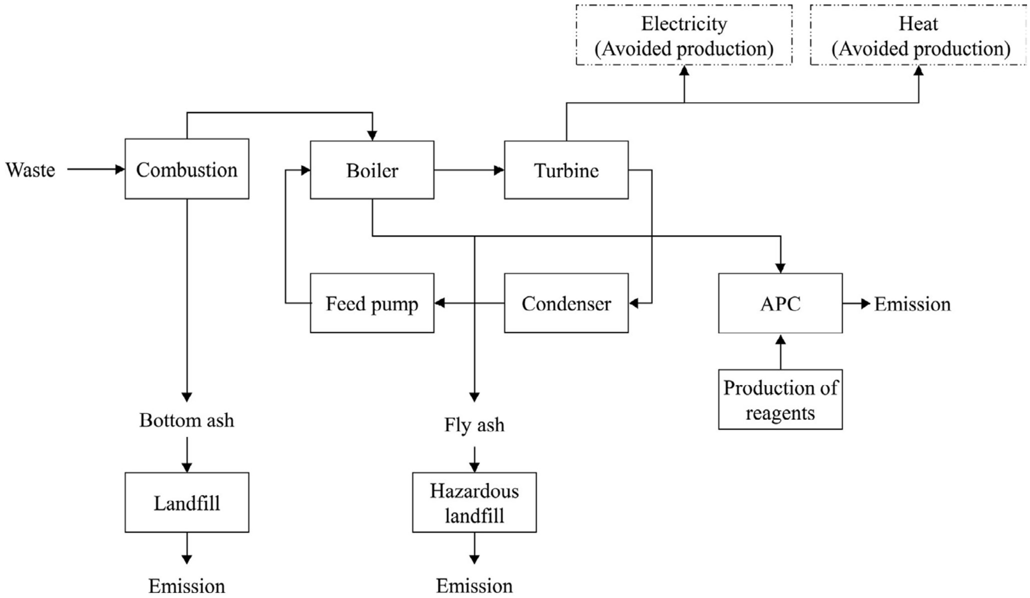

Fig. 1. System description of WtE plant.

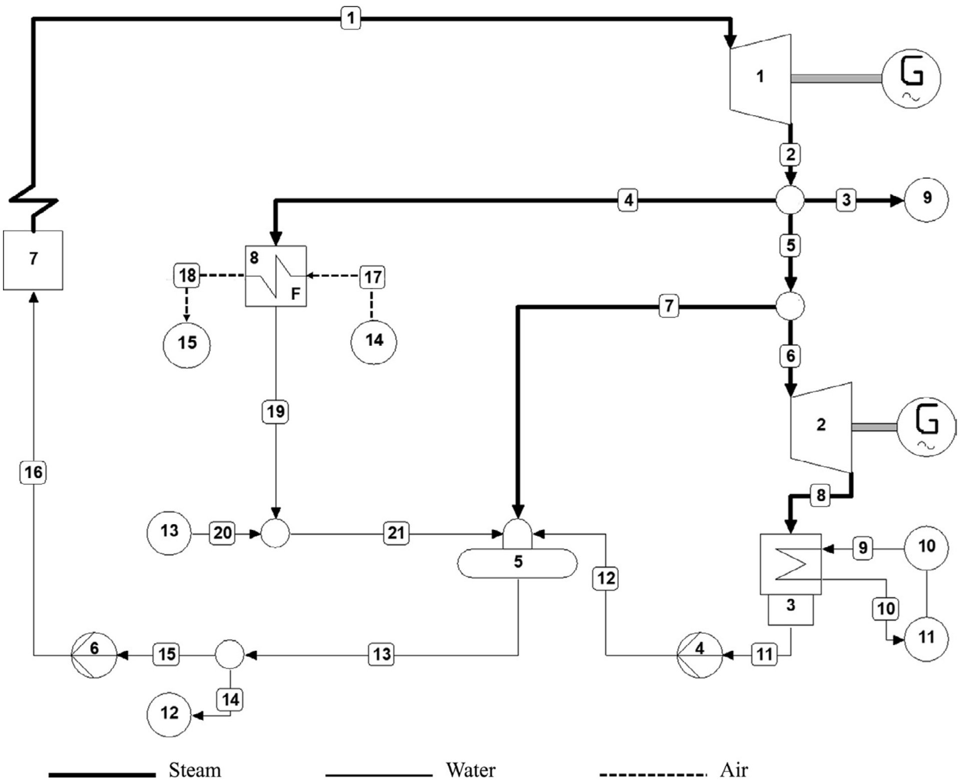

Fig. 2. Schematic of steam turbine cycle studied in this paper. The steam cycle consists of apparatus such as: high-pressure turbine (HPT) (1), low-pressure turbine (LPT) (2), steam condenser (3), condensate pump (4), deaerator (5), feedwater pump (6), boiler (7), heat exchanger (8), source (10, 13, 14), sink (9, 11, 12, 15), and generator (G).

2.2. Energy assessment

For the energy assessment, mass and energy balance are utilized to model the mass flow rate and energy transfer rate among unit operations using the assumption that there is no loss during the operation. The performance indicator for energy assessment is electric efficiency delivered to the grid (ε_{el}) derived from the total electricity recovered from the combusted waste subtracted by the amount for self-consumption. The formula to calculate mass and energy balance are expressed by equations (1) and (2):

$$\sum \dot{m}_i = \sum \dot{m}_o \quad (1)$$

$$\dot{Q} - \dot{W} = \sum \dot{m}_o h_o - \sum \dot{m}_i h_i \quad (2)$$

where $\dot{m}$ is the mass flow rate (kg/s), subscripts i and o indicate the incoming and outgoing stream, respectively,$\dot{Q}$,$\dot{W}$, h are heat (kW), power (kW), and enthalpy (kJ/kg), respectively.

The net energy efficiency is calculated using equation (3):

$$\varepsilon_{el} = \frac{\dot{W}_{net}}{\dot{m}_{waste} \cdot LHV_{waste}} \quad (3)$$

where $\dot{m}_{waste}$, LHV_{waste}, and $\dot{W}_{net}$ are waste mass flow rate (kg/s), waste lower heating value (kJ/kg), and net power (kW), respectively. The net power is determined by using equation (4):

$$\dot{W}_{net} = (\dot{W}_{HPT} + \dot{W}_{LPT}) - (\dot{W}_{pump\ 4} + \dot{W}_{pump\ 6}) - \dot{W}_{self\ \ consumption} \quad (4)$$

where $\dot{W}_{HPT}$and $\dot{W}_{LPT}$ are power generated (kW) by HPT and LPT, respectively, $\dot{W}_{pump\ 4}$ and $\dot{W}_{pump\ 6}$, are power consumed (kW) by pump 4 and pump 6, respectively, and $\dot{W}_{self\ \ consumption}$ is the amount of electricity consumed by the plant (kW) that is generated by the plant. Thermal modeling is initially simulated using Cycle Tempo software which is later compared to the actual system to ensure it is correct. The model is then reconstructed using thermotables, a Ms. Excel thermodynamics add-in (University of Alabama, 2011) since the optimization was performed using an Excel-based MOO program (Sharma et al., 2012; Wong et al., 2016).

2.3. Economic assessment

The economic assessment determines the associated cost of treating the waste, C_p (€/ton-waste). The cost was calculated as the sum of annualized fixed-capital investment (FCI), insurance and maintenance, labor cost, cost of flue gas cleaning and ash disposal, and revenue from electricity sale, as shown in equation (5).

$$C_p = \sum\nolimits_{t=1}^{y} \times \frac{r/(1-(1+r)^{-y}) \cdot FCI + C_{IM} + C_{labor} + C_{FGA} - (\varepsilon_{el} \cdot C_{el} \cdot t_a \cdot \dot{m}_{waste} \cdot LHV_{waste})}{Plant\ \ capacity} \quad (5)$$

where r and$y$ correspond to interest rate and discount period, respectively. C_{IM} indicates the cost of insurance and maintenance, C_{labor} implies the total annual salaries of the personnel (€/year), whereas C_{FGA} refers to the cost of flue gas cleaning and ash management. The revenue is associated with net efficiency (ε_{el}), the price of selling electricity (C_{el}), annual operating hours (t_a), waste flowrate ($\dot{m}_{waste}$), and lower heating value of the waste (LHV_{waste}).

FCI consists of different cost items, including purchased-equipment cost (PEC). PEC was calculated as a function of thermodynamics, where the results will be used to estimate total investment cost. To perform the calculation of PEC, the cost coefficient was adjusted to the year 2018 using the chemical engineering plant cost index (CEPCI, 2018). A percentage of PEC was used to estimate the total investment as a sum of various cost items, such as equipment installation, piping, instrumentation, legal cost, etc. Information concerning parameters and equation used to calculate the cost is given in the Supplementary material (see Tables 2–4).

2.4. Environmental assessment

Environmental assessment was carried out using life cycle assessment (LCA). LCA is commonly used for the environmental accounting of a system or comparing the performance of two or more systems. The methodology in this study follows the procedure provided by the ISO (ISO, 2006a, 2006b). The LCA in this study is used to assess the environmental performance of WtE within the Finnish context. The functional unit (FU) is 1 ton of incoming waste treated in the WtE plant. System boundaries cover direct emission resulted from waste combustion and indirect emission from upstream and downstream activities concerning waste treatment in the WtE. Upstream activities include reagent production and WtE infrastructure, whereas downstream activities comprise ash management and electricity recovery. Other than treating waste, WtE provides a function as electricity and heat producer. This multifunctionality issue was resolved by applying system expansion, where the conventional electricity and heat production system was considered. The electricity from WtE was assumed to substitute the average electricity consumption mix whilst the heat will supersede the average heat consumption by the plant.

A WtE plant recovers energy in the form of electricity for self-consumption and sale, heat for self-consumption, while bottom ash is sent to landfill, and the APC residue is assumed to be sent to hazardous waste landfill. The waste composition for municipal solid waste in Finland was modified from Liikanen et al. (Liikanen et al., 2016) since there is a difference in waste categorization between their study and the present one. The waste composition consists of 45.9% organic waste, 16.8% plastics, 8.8% cardboard, 8% paper, 5.5% textiles, 5.4% composite waste, 3% sanitary textiles, 2% non-combustible (e.g., ceramics), 1.95% metals, 1.55% glass, 0.9% combustible (e.g., wood), and 0.2% hazardous waste.

WtE specification and waste composition were used as inputs for the analysis, and it resulted life cycle inventory (LCI). LCI was quantified using the waste incineration life cycle inventory (WILCI), a tool developed based on the incineration sector in France (Beylot et al., 2018, 2017). This tool was used because it provided a seamless way to define the input, output, as well as the management options for air pollution and ash. Moreover, the results of LCI from WILCI can be modified as an input to perform life cycle impact assessment (LCIA) in OpenLCA software. WILCI also provides results on flue gas volume, which is used to estimate the cost of APC unit.

LCIA was conducted using ReCiPe methodology for the midpoint and endpoint single score result, taking a hierarchist perspective (RIVM, 2016). Hierarchist (H) is rooted from the most common policy approach that uses medium time horizon of 100 years. In this study, the single score impact (SSI) is the indicator of environmental performance that is utilized as the environmental objective in the MOO. The optimized system has to minimize the environmental impact, or in the other words, the system needs to maximize the environmental benefit. To avoid confusion, environmental benefit here refers to the environmental impacts avoided from conventional electricity and heat production, and it was later indicated by a minus sign. Primary data from the plant was used in combination with Ecoinvent database. The temporal scope was 2018–2038, and the geographical scope was Finland.

2.5. Multi-objective optimization

This section describes the methodology for multi-objective optimization, which consists of the objective functions, decision variables, and non-dominated sorting genetic algorithm (NSGA-II).

2.5.1. Formulation of the objective functions

Three objective functions in the WtE system were considered as the optimization problem. They covered the energy, environment, and economic aspects represented by energy efficiency, LCA single score impact (SSI), and cost, respectively.

The objective function of the energy aspect represented by net efficiency (%) is displayed by equation (6):

$$Max\ \ \varepsilon_{el} = \frac{\dot{W}_{net}}{\dot{m}_{waste} \cdot LHV_{waste}}$$

$$subject\ \ to\ \ x = 0.9\ \ and\ \ \eta_{T,s} \leq 0.9; \tag{6}$$

where x and $\eta_{T,s}$ are steam quality in pipe 8 and isentropic efficiency for both turbines (see Fig. 2). The annualized cost, C_p, in treating incoming waste (€/ton-waste) is the economic objective, as shown by equation (7):

$$Min\ \ C_p = FCI + C_{IM} + C_{labor} + C_{FGA} - C_{sale} \tag{7}$$

in which FCI is the fixed-capital investment, C_{IM} the cost of insurance and maintenance, C_{labor} the labor cost; C_{FGA} refers to cost of flue gas cleaning and ash management, $and C_{sale}$ represents revenue from the sale of electricity. For the environmental aspect, SSI is the objective to minimize, as displayed by equation (8):

$$Min\ \ SSI = \sum\nolimits_{n=1}^{n} DE_n + AM_n + RN_n - ER_n \tag{8}$$

where SSI is the total environmental impact and subscript $n$indicates each of the impact categories, whilst DE_n, AM_n, RN_n, ER_n represent the environmental impacts of direct emission, ash management, reagent, and infrastructure, as well as energy recovery, respectively.

2.5.2. Decision variables

Six decision variables were selected, namely high-pressure turbine (HPT) inlet temperature, HPT inlet pressure, HPT outlet temperature, low-pressure turbine (LPT) outlet pressure, and pump isentropic efficiency. To ensure that the optimization results did not exceed a reasonable range of the typical specification of the equipment and standard steam cycle operation, a range of variables and constraints were introduced.

The actual value of the decision variables that were obtained from the WtE operator, as well as the range of design parameters used in the optimization are shown in Table 1. The numbers of the pipes and equipment in the table refer to Fig. 2. Non-dominated sorting genetic algorithm (NSGA-II)

Table 1
Decision variables and range of variation.

Operation configuration	Description	Actual value	Range of optimization
T_1 (°C)	Steam temperature (pipe 1)	400	380 – 500
P_1 (kPa)	Steam pressure (pipe 1)	4100	3800 – 4500
T_2 (°C)	Steam temperature (pipe 2, 3, 4, 5, 6, 7)	198	185–210
P_7 (kPa)	Steam pressure (pipe 8, 11)	23	20–25.5
η_{pc}	Pump isentropic efficiency (component 4)	0.75	0.75–0.85
η_{pb}	Pump isentropic efficiency (component 6)	0.75	0.75–0.85

NSGA-II is one of metaheuristic genetic algorithms inspired by natural selection that is used to generate solutions in the optimization problem. It employs a generating technique whereby a sequence of searching for many Pareto-optimal solutions and deciding the appropriate trade-off to select one of them is carried out (Sharma et al., 2012). NSGA-II is used because (i) a crowding distance method results in diversity in the solutions, (ii) a non-dominating sorting method can generate solutions that are close to pareto-optimal, (iii) an elitist method preserves the best solution in the next generation (Deb et al., 2002; Subashini and Bhuvaneswari, 2012; Yusoff et al., 2011). The optimization problem was solved using an Excel-based MOO (EMOO) program following the principle of NSGA-II developed by Sharma et al. (2012) and Wong et al. (2016).

Maximum number of generations (MNG) is a common termination criterion used in MOO. The iteration has to be large enough to ensure the solutions are converged, but at the same time it should not be too large so that it will cause an excessive number of computations (Wong et al., 2016). This study used steady-state detection (SSDTC) as the termination criterion. This criterion determines convergence based on steady state detection, where it performs precisely with computational efficiency for single-objective optimization (SOO) (Rhinehart, 2014). Wong et al. (2016) developed SSDTC for MOO, which terminates reliably and produces non-dominated solutions close to MNG with quicker computational time. The crossover probability and mutation probability were set at 0.9 and 0.1, respectively, along with population size of 100.

2.6. Sensitivity analysis

Sensitivity analysis was used to investigate how results differ as an effect of a change in input. We applied perturbation analysis, which was implemented by increasing and decreasing each decision variable by 5% of its value while keeping all other variables at their baseline value. The results from perturbation analysis allows the calculation of ratio change between the initial results and perturbation results.

2.7. Scenario analysis

Scenario analysis was used to assess the model's robustness based on the change related to waste management and WtE. Three changes were applied to perform scenario analysis: (i) waste composition, (ii) sustainable energy provision (SEP), (iii) fossil energy provision (FEP). In the first scenario, the change was applied only to organic and plastic waste since these two types of waste are typically significant in the waste composition (Martinez-Sanchez et al., 2016). The organic and plastic waste content in the baseline scenario are 45.95% and 16.8%, respectively, while in the scenario analysis they are 30.9% and 31.8%, respectively. For the two scenarios in energy provision, the change was made in the source of marginal energy. Energy source in SEP consisted of wood, wind, and nuclear whereas FEP consisted of nuclear, natural gas, and hard coal. Information about scenario analysis input is given in Supplementary material Tables 5 and 6.

3. Results

3.1. Energy analysis

The total energy input from the waste was 12.71 MJ/kg-waste. The enthalpy of boiler, HPT, and LPT were −13763.11 kW, 1790.86 kW, and 2085.09 kW, respectively (see Supplementary material Table 7). Waste flow rate per hour was 4.6 ton, resulting total electricity of 3245.72 kW, at which 649 kW was for self-consumption. These results corresponded to 16.27% net efficiency of the system. Studies on the efficiency of WtE with electricity recovery ranging about 14–28% (Beylot et al., 2018; Martinez-Sanchez et al., 2016). The low efficiency of WtE with electricity recovery is caused by energy wasted from electricity generation through heat discharge that is not recaptured for further utilization as in a cogeneration plant (Verbruggen, 2008). The energy wasted is particularly pronounced in between source and sink 10–11 when the steam is being cooled.

3.2. Economic analysis

The economic analysis showed the average cost of treating waste per ton. It considered the fixed cost, which consists of fixed-cost investment, insurance and maintenance, labor cost, as well as cost of flue gas cleaning, ash disposal, and revenue from the sale of electricity. The remaining cost is expected to be covered by a gate fee. Table 2 shows the results of cost items in treating the waste per ton in WtE plant. The total average cost was 75 €/ton-waste, where the major contribution was fixed cost and electricity sale. For the total fixed cost, the contribution from fixed cost equipment, insurance and maintenance, and labor cost contributed about 65.8%, 22.96%, and 11.25%, respectively to the total value of 83.63 €/ton. A similar value was reported by Martinez-Sanchez et al. (2016), where the total fixed cost for WtE with electricity recovery was 83 €/ton-waste. However, the total average cost was different due to system efficiency that caused different values in electricity generation. In this study, one ton of waste generated around 705.47 kW of electricity.

The difference between our results compared with other studies can be affected by different calculation methods, cost items, and assumptions used in estimating fixed cost. Investment cost can be calculated based on capacity using a formula devised by Waste to Energy International (Waste to Energy International, 2015) or using information of the known cost and capacity of other plants, and adjusting the value based on the desired capacity. In this case, we calculated the purchased equipment cost (PEC), which consists of steam cycle and air pollution control, then we used ratio of PEC adopted from Lemmens (2016) to calculate in the rest of the cost components in the FCI. Overall, the cost of this study was congruous with WtE plants that have similar capacity, as shown by ENEA (ENEA, 2007).

Table 2
Economic analysis of treating waste in WtE.

Items	Cost (€/ton-waste)
Fixed cost	83.63
Fixed-capital investment (FCI)	55.02
Direct fixed-capital investment (DFCI)	45.26
– Purchased-equipment cost (PEC)	17.96
– Purchased-equipment installation	6.74
– Piping	4.49
– Instrumentation and controls	2.60
– Electrical equipment and material	2.02
– Architectural, civil, and structural work	6.06
– Service facility	5.39
Indirect fixed-capital investment (IFCI)	9.76
– Engineering and supervision	1.64
– Construction and contractor	4.10
– Contingencies	3.30
– Legal cost	0.73
Insurance and maintenance	19.20
Labor cost	9.41
Flue gas cleaning and ash disposal	8.93
Electricity sale	−16.9
Total average cost	**75.63**

3.3. Environmental analysis

3.3.1. Total impact

On the midpoint level, the global warming potential from direct emission and total emission per ton waste input were 510 kg CO_2-eq and 175 kg CO_2-eq, respectively. Lower total value compared with direct emission were the results of the benefit from energy recovery. The midpoint results were converted into normalized endpoint and weighted score so that SSI can be calculated. For the endpoint, the highest impact was from global warming regarding human health with the value of 1.13×10^{-3} Pt/ton-waste, whereas the highest benefit was fossil resource scarcity at -1.20×10^{8} Pt/ton-waste. The SSI showed net benefit of -1.21×10^{8} Pt/ton-waste. The total impact of treating waste in WtE plant shows a negative environmental impact, or in other words, it provides an environmental benefit from avoided process. Hence, the benefit depends on the amount and the source of electricity being substituted. Information regarding life cycle inventory, midpoint impact, and endpoint impact is given in the Supplementary material Tables 8–10.

3.3.2. Contribution analysis

Contribution of different activities to the environmental impact is shown in Fig. 3. Across all impact categories, energy recovery provided benefits (shown by negative impact), ranging from 27% up to 99% of the total benefits and impacts of the WtE in absolute value. This value means a proportion of energy recovery in its absolute value relative to the sum of impacts from direct emission, ash management, energy recovery (absolute value), as well as infrastructure and reagent. In 10 out of 22 impact categories, energy recovery made the highest contribution to the total impact and benefit. These impact categories were fine particulate matter formation, mineral resource scarcity, freshwater eutrophication, ionizing radiation, fossil resource scarcity, terrestrial acidification, human carcinogenic toxicity, terrestrial ecotoxicity, land use, and freshwater ecotoxicity.

Direct emission contributed around 0–72% of the total impact and benefit across the impact categories. It represented the highest contributor for 9 out of 22 impact categories, namely stratospheric ozone depletion, marine ecotoxicity, human non-carcinogenic toxicity, global warming on terrestrial ecosystem, global warming on freshwater ecosystem, ozone formation on human health, marine eutrophication, ozone formation on terrestrial ecosystem, and global warming on human health. The contribution of reagent and infrastructure ranged around 0–65% across all the impact categories. The highest contribution was found in the impact of water consumption on human health, aquatic ecosystem, and terrestrial ecosystem. Lastly, the management of bottom ash and fly ash only contributed about 0–11% across all impact categories.

3.4. Multi-objective optimization

The MOO was solved ten times using EMOO followed by computation of the true Pareto-optimal front as the outcomes is displayed by Fig. 4. On average, steady state detection (SSDTC) terminated the calculation in generation 141, with 29 as the standard deviation. Maximum improvement for single objective were 13.4%, 10.3%, and 14.8% for thermal, economic, and environmental, respectively. However, these improvements cannot be achieved altogether due to conflicting objectives. Higher efficiency results an increase in cost exponentially, whilst linear correlation is found between environmental impact and efficiency. Therefore weighted percentage deviation factor (WPD) was applied to determine the optimal solution as shown by equation (9) (Inghels et al., 2019).

$$WPD = \sum_{j=1}^{j} W_j \cdot \left[\frac{|f_{j,s} - f_{j,o}|}{f_{j,o}} \right] \qquad (9)$$

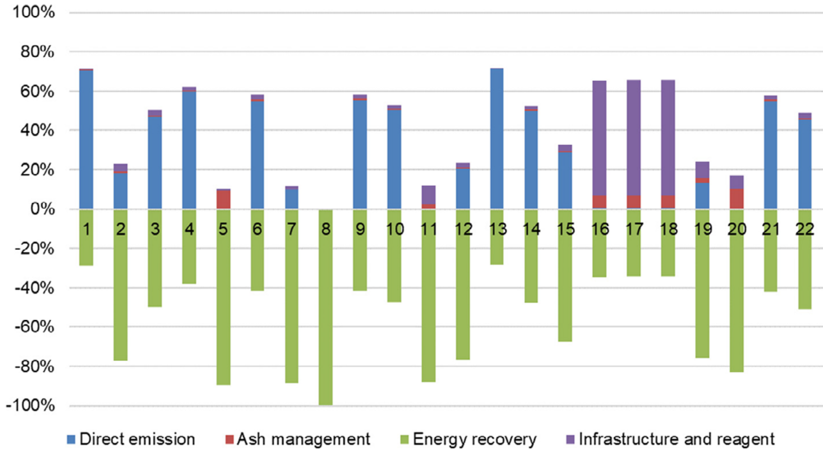

Fig. 3. Normalized endpoint impacts of WtE. The endpoint impacts consist of: 1 Stratospheric ozone depletion; 2 Fine particulate matter formation; 3 Marine ecotoxicity; 4 Human non-carcinogenic toxicity; 5 Mineral resource scarcity; 6 Global warming, terrestrial ecosystems; 7 Freshwater eutrophication; 8 Ionizing radiation; 9 Global warming, freshwater ecosystems; 10 Ozone formation, human health; 11 Fossil resource scarcity; 12 Terrestrial acidification; 13 Marine eutrophication; 14 Ozone formation, terrestrial ecosystems; 15 Human carcinogenic toxicity; 16 Water consumption, human health; 17 Water consumption, aquatic ecosystems; 18 Water consumption, terrestrial ecosystem; 19 Terrestrial ecotoxicity; 20 Land use; 21 Global warming, human health; 22 Freshwater ecotoxicity.

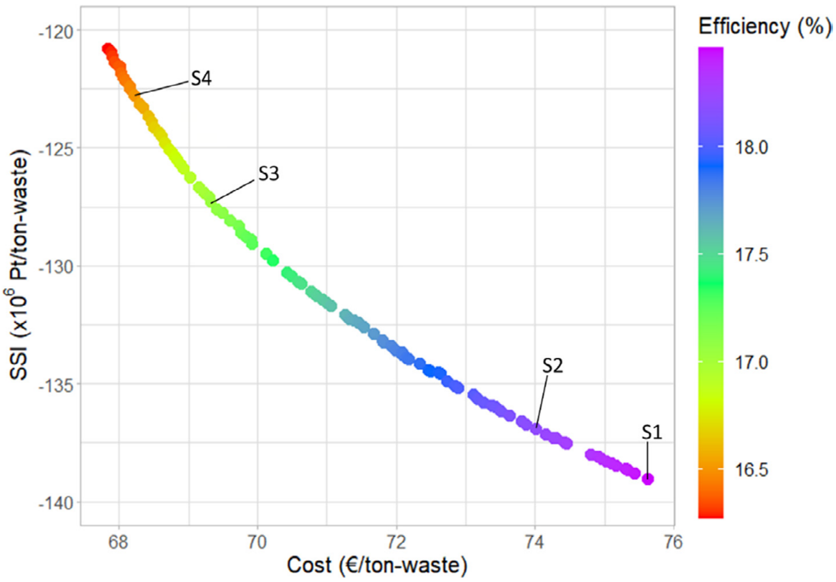

Fig. 4. True Pareto-optimal front of MOO with environmental, economic, and thermal objectives.

where j and W_j indicate objective function and the weight assigned, respectively. The value of jth objective function obtained from true Pareto optimal front and best value of each objective are represented by $f_{j,s}$ and $f_{j,o}$, respectively. The lowest WPD_s is the selected solution due to its closeness to the best value for all objectives.

The outcome of single optimal solution depends on the weight assigned to each objective function by the decision makers. Different set of weight was applied to the environmental objective (W_{en}), economic objective (W_{ec}), and thermal objective (W_{th}) to show the effect of weight factor to the optimal solution. The set of weight including situation (i) S1 that assigns equal weight to all objectives, (ii) S2 with $W_{th} = W_{en} = 0.3,$ *and* W_{ec}= 0.4, (iii) S3 which assumes $W_{th} = W_{en} = 0.2,$ *and* W_{ec} = 0.6, and (iv) S4 with $W_{th} = W_{en} = 0.15,$ *and* W_{ec} = 0.7. Table 3 summarizes the operation configuration for different weight.

3.5. Sensitivity analysis

Sensitivity analysis is used to investigate the varied results due to change in the input variables. This analysis can identify the decision variables that have a significance influence on each objective. Perturbation analysis, where change applied to one variable while holding the rest to the initial value, is conducted by changing six decision variables by +5% and −5%, followed by a calculation of the ratio of change. Relationship of the ratio of change and decision variables is shown by Fig. 5.

Table 3
Operation configuration for different weighting factors.

Operation configuration	Actual value	S1	S2	S3	S4
T_1 (°C)	400	446.73	440.22	414.07	402.85
P_1 (kPa)	4100	4356.80	4214.98	3803.71	3804.12
T_2 (°C)	198	189.29	187.47	185.05	185.05
P_7 (kPa)	23	20.71	20.71	20.71	20.71
η_{pc}	0.75	0.75	0.77	0.79	0.77
η_{pb}	0.75	0.75	0.75	0.75	0.75

Similar results were found for the thermal and environmental objectives, where they are most sensitive with T1. The rest of the decision variables affected the thermal and environmental objectives by less than 1%. These similarities are expected since the MOO shows positive linear correlation between the environmental and thermal objective. Environmental benefit depends on the amount of energy recovery which is the direct definition of efficiency. However, there is a slight difference in the actual value: for example, with a reduction of 5% in T1, efficiency and environmental benefit show a change of about −6% and −6.79%, respectively. For the economic objective, the cost results are most sensitive to T2. When the variable T2 was increased and decreased by 5%, the change in cost results were about 63% and 20%, respectively. Unlike the thermal and environmental objectives, where one decision variable has a much more significant effect on the results, in the economic objective, all variables affect the cost by changing the results by at least 13.5%.

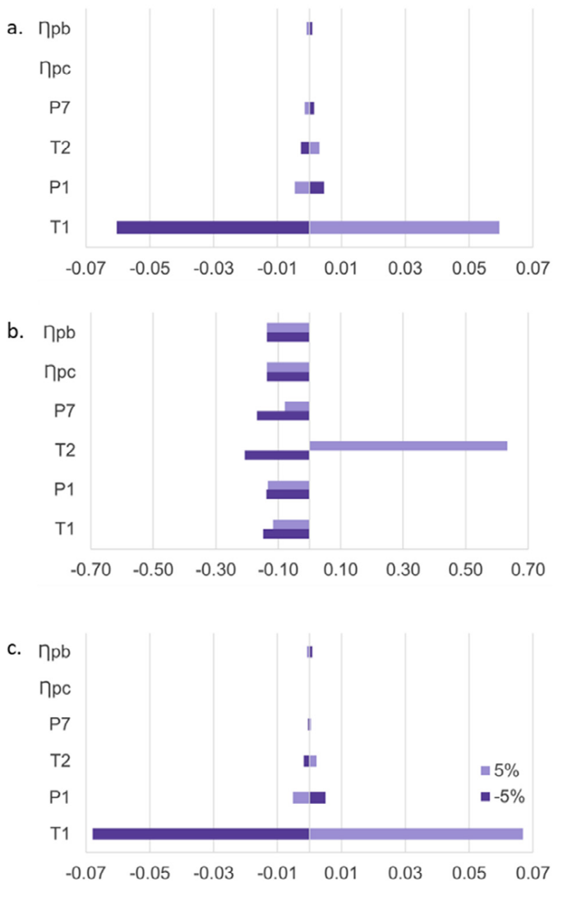

Fig. 5. Sensitivity results of (a) efficiency, (b) cost, and (c) environmental impact due to variations of the decision variables.

3.6. Scenario analysis

3.6.1. Modification of waste composition

A change in waste composition resulted in different outcomes compared with the baseline. The change occurred in thermal, economic, and environmental assessment. The energy balance provided higher results due to the change of waste input. Waste input in a WM scenario has higher LHV at 16.94 MJ/kg, and the system is assumed to have the same efficiency, hence the power output increased as well. The enthalpy of boiler, HPT, and LPT were −18621.85 kW, 2949.84 kW, and 2652.47 kW, respectively. The highest difference compared with baseline scenario occurred in gross energy output of the HPT, at 65%.

The overall cost in treating one ton of waste was 85.61 €, showing an increase of about 13% compared to the baseline. Higher fixed cost and higher revenue were obtained when waste input has higher LHV, with a slight decrease in the cost of flue gas cleaning and ash disposal. The SSI of waste modification scenario was $-1.63\text{x}10^{8}$ Pt/ton-waste, showing that modified waste provided higher benefit to the environment for about 35%. This is caused by the higher power output so that more electricity production can be avoided and substituted by WtE production. See Supplementary material for complete results in WM scenario (Tables 11–13).

The WM model was then solved ten times using EMOO for a comparison with the baseline scenario. On average, the calculation terminated at generation 134 with a standard deviation of 32. A similar improvement can be found in baseline and WM scenarios as a result of the MOO. The maximum improvements in energy efficiency in the baseline and WM scenario were about 13% and 15%, respectively. The economic objective could be improved by around 11.5% and 12.6% at the highest in the baseline and WM, respectively. Meanwhile, the environmental objective had the highest improvement of about 13% and 14% for baseline and WM scenario, respectively.

Performance metrics (PM) were calculated to compare the performance of MOO in finding the non-dominated solutions (Sharma et al., 2017). PM are useful in measuring the performance of MOO algorithm so that they were utilized to evaluate the model when modification was made (Wong et al., 2016). Normalized spread (nSP) and generational distance (nGD) are used as performance metrics in this study. The objectives are normalized using extreme value to avoid bias (Sharma et al., 2017). The first metric, nSP, is used to identify the scope of computed Pareto-optimal fronts so that the larger value is the better one (Audet et al., 2020), whereas nGD measures the convergence performance at which the lower value indicates the closest solutions to true Pareto-optimal front (Sharma and Rangaiah, 2013).

The value of nGD for baseline and WM scenario were similar at about 0.000234 and 0.000227, respectively. Both models provide non-dominated solutions that are equally close to the value of true Pareto-optimal. For spread, the nSP results were 0.5297 and 0.4916 for baseline and WM, respectively. This shows that the baseline scenario has a wider extent of spread in a Pareto-optimal front.

3.6.2. Modification of electricity mix

The second type of scenario applied change in the source of the marginal energy mix. SEP comprised of greener energy sources compared to baseline, whereas FEP consisted of an energy mix that was less green compared with the baseline. The calculation assumed that the electricity price remained the same regardless of the source of the energy. Hence, the change in outcome was only found in the environmental benefit derived from avoided electricity production. The environmental benefit in the SEP and FEP scenario were -2.49×10^7 Pt/ton-waste and -3.43×10^8 Pt/ton-waste, respectively. SEP and FEP scenario differed by about −93% and 183% from the baseline scenario, respectively (see Supplementary material Table 14).

SEP and FEP scenarios were optimized to evaluate how the model would behave with a modification. The average termination for SEP and FEP were generations 98 and 129, respectively, whereas the standard deviations were 27 and 29, respectively. The results of performance metrics nGD for baseline, SEP, and FEP were 0.000234, 0.000575, and 0.000266, respectively. FEP showed similar nGD with the baseline, which implied that the non-dominated solutions were close to the true Pareto-optimal front. Meanwhile, the value of nGD for SEP was two times higher than the baseline and FEP, indicating that the non-dominated solutions were less converged. For spread, nSP results for baseline, SEP, and FEP were 0.5297, 0.5517, and 0.7037. For these metrics, similarity was found in the baseline and SEP, where the spread of non-dominated solutions was less extensive than FEP. In both PMs, FEP scenario showed better performance.

4. Discussions

4.1. Importance of waste composition

Waste compositions affect the results of thermal, economic, and environmental assessments. It determines the LHV and chemical contents that will affect the combustion process, emission type and quantity, and the operating cost. Therefore, difference can be found in different studies regarding LCA of WtE although comparable pattern exists across different studies. Midpoint climate change (CC) impact of this study as a result of a direct emission in every ton of waste is 510 kg CO_2-eq. Similar findings were found in Beylot et al. (2018) where the value was around 400 kg CO_2-eq. Comparable results were found in studies by Astrup et al. (2009) and Damgaard et al. (2010) where direct CC impact were 347–371 kg CO_2-eq and 300 kg CO_2-eq, respectively. Within Norway context, Lausselet et al. (2016) reported the CC impact in different scenarios ranging from 265 to 637 kg CO_2-eq.

Waste composition also affects the cost in treating per ton waste in WtE plant. The baseline of this study shows that the cost in treating incoming waste is 75.63 €/ton-waste. The result increases to 85.61 €/ton-waste in scenario analysis as the waste composition is modified. Martinez-Sanchez et al. (2016) confirmed the pattern when waste input has higher LHV. The cost increased with higher LHV due to lower mass flow rate treated in the plant.

4.2. Importance of assumptions and assessment method

The assumptions, system boundary, functional unit, and methods affect the results of LCA, thermal analysis, cost calculation, and optimization problem. The average condition, common method, and FU are used to accommodate the differences among all possible value and enable comparison across studies. For the LCA, there are various impact assessment methods that include different substances, classify impact categorization differently or present the results as midpoint or endpoint result. Midpoint results are commonly used in LCA study, hence it is used as well in this study for comparison purpose. However, for the MOO, the single score method was apply. ReCiPe midpoint impacts mm and endpoint impacts consist of 18 and 22 categories, respectively. Using each impact as separate objective function in either midpoint or endpoint method will become impractical. Single score can simplify the calculation while containing all different impact categories at one. This simplification comes with caveat that some information may be condensed resulting higher uncertainty (Meijer, 2014).

The choice of system boundary and economic assumption must be representative for the system being assessed and commonly used for comparison with other studies. This study covers the direct emission and indirect emission including system expansion method. This choice is made to avoid overlooking environmental benefit from energy recovery. System boundary can be defined iteratively along with inventory analysis to reassure the relevant boundaries are covered (Baumann and Tillman, 2004). Broad range of economic assumption such as discount period, discount rate, electricity price, and fixed-capital investment cost that is calculated using percentage of PEC influence the cost function. Gate fee is not included in this study as it should be decided after the cost of treating waste is known. So that the economic assessment focus on the cost in treating waste instead of the revenue from selling electricity.

The finding also highlights the role of decision makers in determining optimal solution through assigning weight to each objective function. The total of weight across different objective function must be 1, and the objective function that is considered relatively more important has to be assigned higher weight. Various factors such as stringency of environmental policy in certain region, labor wage and the price of consumables, thermodynamics characteristics of the equipment, and the sources of marginal energy may affect the way the decision makers prioritize the objective function.

4.3. MOO parameters

SSDTC terminates the computation for various scenario in generation 98–141. Other termination criteria is maximum number of generations (MNG) that is commonly used in MOO. MNG must be large enough to make sure the results are converged but not too large that it can cause unnecessary computation. It was reported by Roosen et al. (2003) that an increase in MNG from 150 to 730 resulted marginal improvement, and computation for more than 1000 generations provided negligible improvements. MNG for NSGA-II for power generation study can range from 400 to 700 (Behzadi et al., 2018; Ghasemian and Ehyaei, 2018; Hajabdollahi et al., 2012). The use of alternative termination criteria other than MNG can save computational time.

Crossover and mutation probability in NSGA can range around 0.7–0.9 and 0.01–0.2, respectively (Ghasemian and Ehyaei, 2018; Hajabdollahi et al., 2012; Mousavi-Avval et al., 2017). There is no general value to use for crossover and mutation probability, and it can be problem specific (Hassanat et al., 2019).

4.4. Sensitivity and scenario analysis

Perturbation analysis shows how sensitive the thermal and environmental model to T1, and the cost model to T2. The analysis is useful to assess the sensitivity of the model to the decisions variable so that the MOO can focus on fewer decision variables that are most sensitive with expectations of saving computational requirement for the optimization. The high sensitivity of these variables also shows that only small change is required to optimize the system without violating the range of equipment specifications shown by Table 1.

Scenario analysis demonstrates the importance of waste composition as discussed in Section 4.1. The change in waste composition will shift the energy balance including the power output of the system, environmental impact, and cost function. Although it should be noted that differences on the outcomes are also affected by ash management, APC technology, impact assessment methods, energy recovery as well as underlying assumptions used in the study such as electricity source being substituted (Beylot et al., 2018; Fruergaard Astrup et al., 2015; Lausselet et al., 2016; Turconi et al., 2011). Attention is required as well to the background system as the modification of the energy mix shows significant change in LCA results. It implies that the more sustainable the sources of the marginal energy, the less environmental benefit is obtained. Whereas WtE provides more environmental benefits when marginal energy sources are less sustainable. It is possible that WtE provides no benefit to the environment if the marginal energy has exceptionally sustainable source.

Scenario analysis can also be used to evaluate the EMOO by measuring nGD and nSP. The change in the foreground system, represented by waste modification, does not change the convergence of the solutions resulted by the EMOO as shown by comparable nGD, however an extent of spread for baseline is better than WM scenario. The change in the mixed of marginal energy source represents a shift in background system. SEP scenario performs worst in the convergence of non-dominated solutions while FEP performs best for the spread. The variety resulted by scenario analysis indicates that this study is contextual so that careful consideration is needed when generalizing the results of this study.

4.5. Implications and limitations

The results demonstrate that an improvement in WtE plant is possible by applying small changes in the operation configuration without requiring new investment. The relationship between the three objective functions indicated the conflict between cost and efficiency, while positive linear correlation presents the environmental impact and efficiency because the benefit from WtE is derived from the amount energy being recovered. Nevertheless, a separate environmental objective is necessary to ensure that WtE still provides environmental benefit, otherwise waste diversion for different treatment may be required. The method of the study can be implemented not only for WtE plant that is in ongoing operation, but also in the design phase. In designing new WtE plant, the decision variables can be expanded by considering different types of APC technologies and ash management.

The study covers a broad range of aspects that require large data input and various methodologies. Unavailable data were estimated, and this could lead to uncertainty. The choice of methodologies and formula affected the results of the study. Data and methodological issues are especially pronounced in economic and environmental assessment. To address this, the most common methodologies were chosen as well as the implementation of sensitivity analysis and scenario analysis to study how the model behaves and what parameters affect the model the most.

MOO calculation provides different choices for termination criteria, mutation probability, and crossover. However, we applied only one type of these aforementioned categories based on a previous study of the use of EMOO program (Wong et al., 2016). The use of different values of crossover and mutation probability can provide different results since there is no global value to use for these parameters. Our focus on using value and termination criteria that haves been tested limits the study on the effect of these parameters.

5. Conclusion

This paper has presented an MOO that integrates LCA to assess environmental objective. The integration of LCA and the use of single score endpoint allowing comprehensive assessment of the environmental objective that are commonly presented as damage cost or CO_2 emission. The use of MOO can improve the performance of WtE plant although a conflict occurs between the economic and thermal objectives, while positive linear correlation is found between the thermal and environmental objective. Each objective shows maximum improvement for about 13.4%, 10.3%, and 14.8% for thermal, economic, and environmental, respectively. These findings present an important role of decision makers to weigh the priority of each objective and generate optimal solution. The study suggests incorporating MOO not only during operational phase of WtE, but also during the planning phase of building a WtE by including more decision variables such as different type of equipment or technology to improve its design. This will provide general information about how the WtE will perform during its operational time.

The paper also demonstrates that each decision variable affects the outcomes differently. By obtaining the information about the most influential variables with regards to the optimization results, modification to the optimization problem can be applied by reducing the number of decision variables to save computational time. Furthermore, applying MOO will help the plant to continuously evaluate the environmental benefit derived from WtE. As the marginal energy sources changes, the environmental benefit will change up to the point that WtE operation is not environmentally beneficial. Knowledge about this matter can help decision makers to formulate waste management policy regarding appropriate treatment or a decision in diverting waste stream.

Overall, WtE plant can be optimized by modifying operation configuration without making new investment. Careful consideration is required when generalizing this study because (i) the WtE operation is specific for plant with a certain steam cycle structure, waste composition, energy recovery, APC technologies, and ash management, (ii) the assessment was carried out using the Finnish or European context, (iii) the impact assessment method for the environmental objective used ReCiPe (H), and (iv) the cost function depends on equipment with specific thermodynamic properties.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2021.04.042.

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