Miia Liikanen

IDENTIFYING THE INFLUENCE OF AN OPERATIONAL ENVIRONMENT ON ENVIRONMENTAL IMPACTS OF WASTE MANAGEMENT

ACTA UNIVERSITATIS LAPPEENRANTAENSIS 887

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Abstract

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Ever-increasing waste generation, resource depletion and awareness of adverse environmental impacts all factor into the growing application of life cycle assessment (LCA) as a method for evaluating the potential environmental impacts of waste management. A waste management system is an inherent part of an operational environment because of the close relationship between the waste management and surrounding systems, such as energy production. The environmental impacts of those surrounding systems are thus typically encompassed within the system boundaries when assessing the environmental impacts of waste management. LCA studies of waste management systems in the literature have revealed that the environmental impacts of surrounding systems may well outweigh the impacts generated by waste treatment activities. As operational environments are influenced by socio-economic, political, legislative, technological and geographical aspects of a given case area, these aspects in turn influence the associated waste management system.

The objective of the research herein is to explore the influence of an operational environment on the environmental impacts of waste management, through the lens of LCA as a research method. The comparison of the environmental impacts of waste management in markedly different case studies conducted in distinct corners of the globe, namely in Finland, China and Brazil, enables one to identify the variations in the environmental performance of different waste management alternatives. The three central research questions of this dissertation are as follows: (1) What are the environmental impacts of waste management in the case areas, and how might these be decreased?; (2) How do the environmental impacts of different waste treatment methods differ among the operational environments?; and (3) What are the most important reasons underlying the differences?

This dissertation addresses the objective and research questions through four individual case studies in which LCA has been applied to assess the potential environmental impacts of waste management. Even though the case studies differ in many respects, they do exhibit fundamental similarities, thus enabling their comparison from the standpoint of the thesis. The case studies were carried out acknowledging the context- and case-specific characteristics of the case areas. Thus, in order to facilitate utilization of the results in decision- and policy-making, the assessed scenarios have been outlined case-by-case,

rather than being presented as an arbitrary comparison of different waste treatment alternatives.

In exploring the role of an operational environment in the environmental impacts of waste management, the case studies revealed that socio-economic, technological and geographical aspects have a determining influence on the environmental impacts of waste management. In the case studies, the energy recovery rate of waste incineration was identified as the most important factor influencing the results when the environmental performance of incineration was assessed with respect to other waste treatment methods. The energy recovery rate of waste incineration was influenced by numerous factors, such as waste composition, the technological maturity of waste incineration and, most importantly, the need for the recovered energy. These factors were in turn influenced by the aforementioned aspects of an operational environment. The political aspects of operational environments were not found to directly influence the environmental impacts of waste management, but instead were found have a distinct effect on the goal and scope of the case studies.

The thesis identified the most important reasons underlying the differences among the case studies. The aspects of an operational environment should be acknowledged, particularly when exploring the differences in the environmental performance of waste treatment alternatives in different case areas. This plays a vital role, for instance, when outlining the correlation between the priority order of the waste hierarchy and environmental impacts in different areas and waste management systems.

Keywords: waste management, operational environment, life cycle assessment, environmental sustainability, environmental impact assessment, energy recovery, material recovery, landfill disposal

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While starting my university studies in 2010, little did I know that nine years later I would finish my doctoral dissertation. Nevertheless, the path that began in 2010 has been rather straightforward regardless of few moments of doubt along the way.

This work was carried out in the course of the years 2015 and 2019 at the Unit of Sustainability Science in the School of Energy Systems at Lappeenranta-Lahti University of Technology LUT. During that time, I had the opportunity and privilege to learn about the environmental impacts of waste management systems while working with a number of experts whose contribution to this thesis I wish to acknowledge.

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The research work was carried out in the *Material Value Chains (ARVI)* programme and in the *Life IP on waste – Towards circular economy in Finland* (LIFE-IP CIRCWASTE-FINLAND) project. Tekes, the Finnish Funding Agency for Technology and Innovations (currently Business Finland), as well as industry and research organisations are acknowledged for funding the research conducted in Publications I-III. EU LIFE Integrated programme, as well as from companies and cities are acknowledged for funding the research conducted in Publication IV.

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Miia Liikanen
November 2019
Lappeenranta, Finland

Contents

List of publications

This dissertation is based on the following papers. The papers are listed in chronological order, according to the date of publication. The rights have been granted by publishers to include the papers in the dissertation.

I. Havukainen, J., Zhan, M., Dong, J., Liikanen, M., Deviatkin, I., Li, X., and Horttanainen, M. 2017. Environmental impact assessment of municipal solid waste management incorporating mechanical treatment of waste and incineration in Hangzhou, China. *Journal of Cleaner Production*, 141, pp. 453–461. doi: 10.1016/j.jclepro.2016.09.146.

II. Liikanen, M., Havukainen, J., Hupponen, M., and Horttanainen, M. 2017. Influence of different factors in the life cycle assessment of mixed municipal solid waste management systems – A comparison of case studies in Finland and China. *Journal of Cleaner Production*, 154, pp. 389–400. doi: 10.1016/j.jclepro.2017.04.023.

III. Liikanen, M., Havukainen, J., Viana, E., and Horttanainen, M. 2018. Steps towards more environmentally sustainable municipal solid waste management – A life cycle assessment study of São Paulo, Brazil. *Journal of Cleaner Production*, 196, pp. 150–162. doi: 10.1016/j.jclepro.2018.06.005.

IV. Liikanen, M., Grönman, K., Deviatkin, I., Havukainen, J., Hyvärinen, M., Kärki, T., Varis, J., Soukka, R., and Horttanainen, M. 2019. Construction and demolition waste as a raw material for wood polymer composites – Assessment of environmental impacts. *Journal of Cleaner Production*, 225, pp. 716–727. doi: 10.1016/j.jclepro.2019.03.348.

Author's contribution

Miia Liikanen was the principal investigator and author in Publications II-IV. In Publication I, Dr. Jouni Havukainen was the principal investigator and author, and Miia Liikanen assisted in LCA modelling and contributed to the writing of the article by providing suggestions for improvement and comments.

Supporting publications

Liikanen, M., Sahimaa, O., Hupponen, M., Havukainen, J., and Horttanainen, M. 2016. Updating and testing of a Finnish method for mixed municipal solid waste composition studies. *Waste Management* (52), pp. 25–33. doi: 10.1016/j.wasman.2016.03.022.

Liikanen, M., Havukainen, J., Grönman, K., and Horttanainen, M. 2019. Construction and demolition waste streams from the material recovery point of view: A case study of the South Karelia region, Finland. *WIT Transactions on Ecology and the Environment* (231), pp. 171–181. doi: 10.2495/WM180161.

Nomenclature

Subscripts

0	baseline scenario
eq	equivalent
i	an alternative scenario
max	maximum

Abbreviations

AD	Anaerobic Digestion
AP	Acidification Potential
ARVI	Material Value Chain research programme
AVG	Average
CDW	Construction and Demolition Waste
CHP	Combined Heat and Power
EIA	Environmental Impact Assessment
EC	European Commission
EC-JRC	European Commission Joint Reseach Centre
EP	Eutrophication Potential
EU	European Union
GHG	Greenhouse gas
GNI	Gross National Income
GNP	Gross National Product
GWP	Global Warming Potential
HDPE	High-density polyethylene
HFO	Heavy Fuel Oil
ILCD	International Reference Life Cycle Data System
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LCT	Life Cycle Thinking
LFG	Landfill gas
LHV	Lower heating value
Ltd.	Limited company
MAX	Maximum
MBT	Mechanical-Biological Treatment
MIN	Minimum
MSW	Municipal Solid Waste
PP	Polypropylene
PVC	Polyvinyl chloride
R	Recipe

RWR	Relatively weighted result
S	Scenario
SR	Sensitivity ratio
WPC	Wood Polymer Composite
RDF	Refuse-Derived Fuel

Chemical compounds

CO_2	Carbon dioxide
NO_x	Nitrogen oxygen compounds
PO_4^{3-}	Phosphate
SO_2	Sulphur dioxide

1 Introduction

1.1 Background

Waste is an outcome of our, i.e. mankind's, way of life in the modern world. The concept of waste connotes any given substance or object which the holder or owner discards, intends to discard or is obligated to discard (European Commission, 2008). Thus far, waste has been regarded as an inevitable consequence of the living standards and consumption habits of people in today's world – waste will be generated as long as there are people generating it. However, the concept of waste has started to evolve from mere waste to potential resources over the last decades due to a growing awareness of the adverse impacts of waste on the environment and the depletion of resources all over the globe.

According to the waste framework directive of the European Union (EU) (2008/98/EC), the primary objective of any waste policy should be to minimize the negative impacts of waste and waste management on human health and the environment. Waste policy should also reduce the use of resources and favour the priority order of waste hierarchy. The waste hierarchy is the backbone of waste policy and legislation in the EU. (European Commission, 2008.) Even though the waste hierarchy steers waste policy in the EU, it has also been adopted as a guideline for sustainable waste management elsewhere, for instance in Japan (Dijkgraaf and Vollebergh, 2004). The waste hierarchy defines the priority order for waste prevention and management. Following the priority order, waste generation should foremost be prevented. If waste is, however, generated, it should be primarily prepared for re-use. If that is not possible or applicable, waste should be recycled. If recycling is not possible, waste should be recovered in another manner, for instance, via energy recovery methods. The last option of the priority order is disposal, provided that other treatment methods are not possible or applicable. (European Commission, 2008.)

As a general guideline, the waste hierarchy should lead to the best overall option in light of environmental impacts (European Commission, 2008). However, studies evaluating the potential environmental impacts of waste management systems and applying life cycle assessment (LCA) as a method, have demonstrated that the environmental impacts do not always correlate with the priority order of the waste hierarchy. For instance, Andreasi Bassi et al. (2017) discovered that the environmental impacts of household waste management do not clearly correlate with the rate of recycling. Particularly, the ranking between recycling and energy recovery from the point of view of environmental impacts relies heavily on the study context, referred to henceforth as an 'operational environment' in this dissertation. This has been acknowledged in the waste framework directive; according to the directive, a departure from the priority order may be required for specific waste streams if justified by technical feasibility, economic viability and environment protection, for instance. When such a departure is justified for reasons of environmental protection, life cycle thinking (LCT) has been presented as a method of justification for achieving the best overall environmental outcome in the directive. Laurent et al. (2014a) introduced the idea that policymakers should better acknowledge so-called context-specific waste hierarchies, which are based on LCA studies taking into account, for example, case-specific waste composition, treatment efficiencies and regional energy production. These context-specific waste hierarchies may not always correspond to the priority order. In practical terms, departing from the waste hierarchy is not that straightforward; quite the contrary. In their study, Lazarevic et al. (2012) discussed the issue of justifying a departure from the priority order of waste hierarchy with LCA. They concluded that LCA cannot always provide explicit justifications for a departure due to, for instance, the context specificity to waste management systems and the complexity of LCA leading to an ambiguity of results.

A number of methods for evaluating the environmental impacts of products and systems have been developed and established in academia. Finnveden et al. (2007) evaluated the applicability of environmental impact assessment methods for different purposes of use in the field of waste management. The methods evaluated included environmental impact assessment (EIA), strategic environmental assessment, LCA, risk assessment, material flow accounting and environmental auditing. While the features and characteristics of all these methods are not dealt with herein, the following clarification is required to avoid misapprehension. Even though LCA is a commonly applied method of assessing the environmental impacts of products and systems, it should not be confused with EIA. EIA is a procedural method used to assess the environmental impacts of projects. It is a highly site-specific method. LCA, instead, is a method used to evaluate the potential environmental impacts of products or systems throughout their life cycle; from raw material acquisition to waste treatment, including all phases in-between. (Finnveden et al., 2007.)

A waste management system encompasses the collection and transportation of waste, pre-treatment methods, such as mechanical treatment, material recovery processes, incineration and landfill disposal. Direct emissions are generated in these processes. Since waste management is an inherent part of an operational environment, other closely associated systems, such as local energy production, influence the overall environmental impacts of waste management. In the case of energy production in this context, the impact can occur either directly or indirectly. A direct impact is an outcome of energy consumed in the waste treatment processes, whereas an indirect one is an outcome of energy recovery from waste when the recovered energy substitutes for other energy production in an operational environment. These aspects are also acknowledged when assessing the environmental impacts of a waste management system with LCA (Ekvall et al., 2007a). Therefore, waste LCA studies do not focus solely on a waste management system.

1.2 Objectives

The objective of the present thesis is to explore the influence of an operational environment on the environmental impacts of waste management, by employing LCA as a research method. An assessment of the environmental impacts of waste management in the context of four distinct case studies (executed in geographically disparate Finland, China and Brazil) allows for the variations in the environmental performance of different waste management alternatives to be identified. The primary objective of the thesis may be further broken down into the following three research questions:

(1) What are the environmental impacts of waste management in the case areas in Finland, China and Brazil, and how might these be decreased?
(2) How do the environmental impacts of different waste treatment methods differ among the operational environments?
(3) What are the most important reasons underlying the differences?

The connection between the objectives, research questions and the publications included in this thesis is illustrated in Figure 1.1.

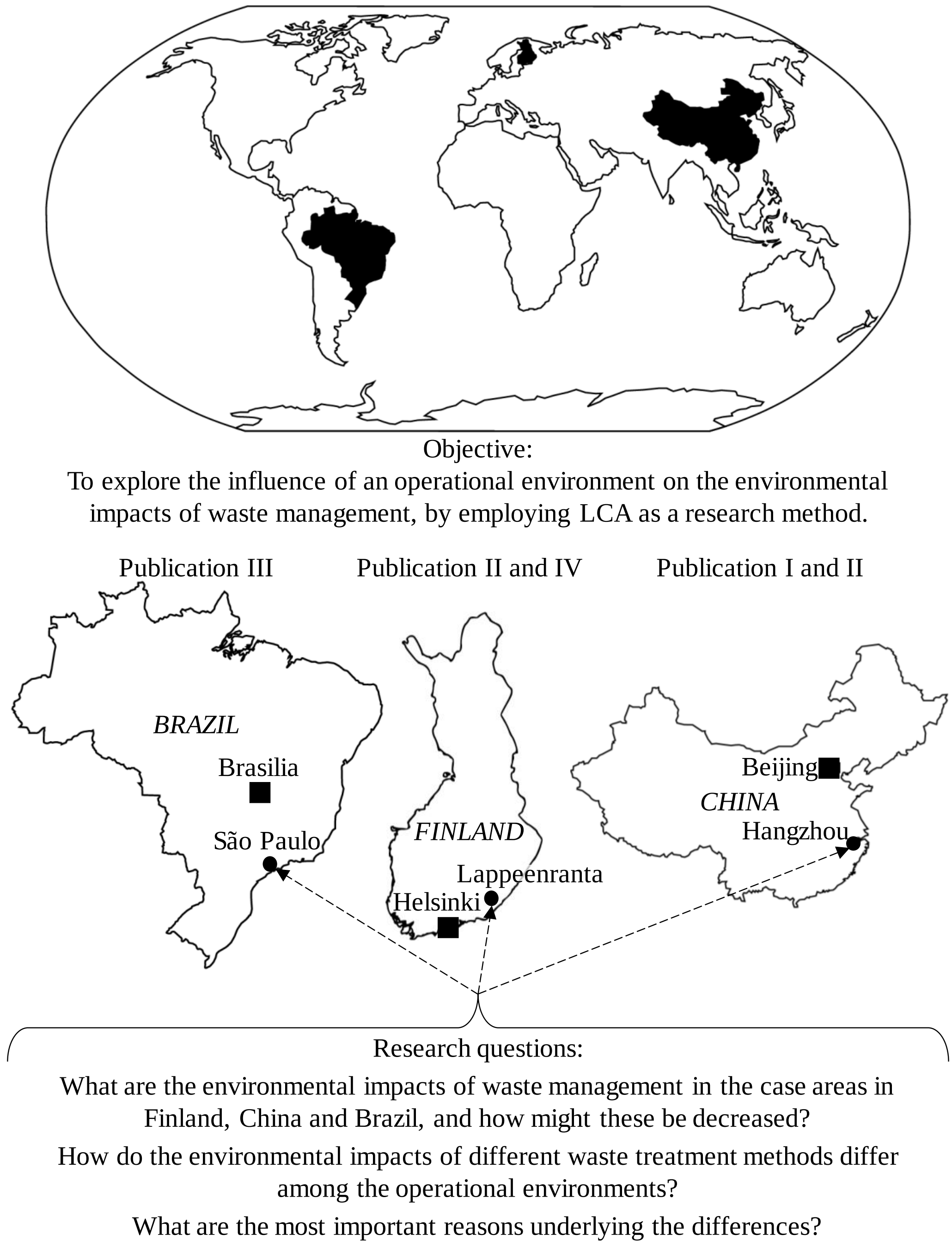

Figure 1.1. Objectives and research questions connected with the publications included in this thesis.

Since this thesis is comprised of individual case studies, the research gaps identified in the case studies are inherently part of the thesis, thereby forming the basis for the research gap of the thesis. The research gaps of the case studies may be summed up as follows. In Publication I, the environmental impacts of introducing mechanical treatment prior to incineration were assessed and compared to the baseline situation, in which mixed waste is incinerated without mechanical treatment. The mechanical treatment prior to incineration enables a decreasing – and potentially even an ending – of the need for auxiliary fuel, namely coal in this case. In Publication II, two distinctly different mixed waste management systems in Finland and China were compared with each other in terms of the influence of various parameters on the total environmental performance of waste management. The comparison of the two case studies made possible the identification of differences between the case studies regarding parameter sensitivity. This shed light on the further analysis of the influence of an operational environment on the environmental performance of waste management contained in this thesis. In Publication III, the environmental impacts of the municipal solid waste (MSW) management system in the city of São Paulo, Brazil were assessed. Previously published LCA studies about waste management in São Paulo have focused on specific treatment methods, rather than taking into account the MSW management system as a whole and as consisting of different treatment options for different MSW streams. Publication IV assessed the environmental impacts of utilizing construction and demolition waste (CDW) fractions as raw materials for wood polymer composites (WPCs), instead of treating the CDW fractions with conventional methods such as landfilling and incineration. Previously published LCA studies concerning the WPCs have focused on the environmental impacts of WPC production rather than on assessing it as part of a CDW management system; i.e. as a material recovery method for CDW.

This thesis reveals the differences in the environmental performance of waste management in distinctly different case studies and operational environments. A similar comparison has been published in the literature, yet only from a geographically more uniform standpoint; for instance, a comparison has been made among selected countries in Europe (e.g. Andreasi Bassi et al., 2017). The study provides insight into how the environmental impacts of waste management are influenced by different aspects of an operational environment; this enables a better understanding of the inherent relationship between waste management and the operational environment. When planning and developing alternative treatment methods and steps for improvement for the waste management system of a specific case study location, having a better overall understanding of the influence of an operational environment on the environmental impacts of the waste management will facilitate the identification of the most effective and (simultaneously) realistic alternatives in a given case area.

1.3 Scope and limitations

The thesis addresses the aforementioned objectives and research questions through case studies. This is a commonly applied research approach in the field of waste management because of the inherent connection between waste management and the surrounding operational environment. Case studies may represent actual waste management systems or merely be hypothetical ones. Whether to apply actual or hypothetical case studies depends on the final purpose of use of the study. Case studies representing actual waste management systems can aid in regional policy- and decision-making (e.g. Hupponen et al., 2015; Turner et al., 2016). Hypothetical case studies are commonly applied to implement different methodological approaches or to develop modelling (e.g. Bisinella et al., 2016; Clavreul et al., 2012). The publications included in this thesis represent actual waste management systems, thus enabling an identification of the influence of an operational environment on the environmental performance of different waste management alternatives. The case studies represent diverse operational environments: Finland, a high-income country in Northern Europe; China, an upper-middle-income country in East Asia; and Brazil, an upper-middle-income country in South America. The geographical scope of the thesis is thus narrowed down to these countries.

Since the thesis is based on case studies, the inherent limitations of these similarly pertain to this thesis. As the term ‘case study’ implies, the studies are typically highly case-specific, which might inhibit the generalization of the results. At the same time, as more case studies are conducted, more information about the research problem is accumulated, thus contributing to the generalization of knowledge and ultimately to the reaching of a consensus. The issue concerning the generalization of results should, thus, be borne in mind when interpreting the results of individual case studies. Due to the above-mentioned reasons, the thesis does not intend to provide a global overview or answers about the subject; it rather focuses on the findings of the case studies and draws conclusions based on those. Therefore, it is important to contrast the findings of case studies with the findings of the previous literature to discover whether the case studies support or challenge the prevailing consensus or knowledge.

Specific waste streams and management systems are investigated in the publications included in the thesis. In Publication I, the environmental impacts of the mixed waste management system in the city of Hangzhou in China are assessed. Mixed waste constitutes the residual proportion of MSW after the source separation of different waste fractions, such as organic waste, metal and glass. A four-bin collection system, having separate collection for organic waste, recyclables, hazardous waste and other waste, has been established in Hangzhou (Dong et al., 2013). However, source separation has been inefficient, and the composition of mixed waste is dominated by food waste, comprising 56% of the mixed waste (Publication I). The composition of mixed waste in China is more similar to the composition of the MSW in Finland than to the composition of mixed waste in Finland, which is one of the key differences between the waste management systems. In Publication II, two different waste management systems, Finnish and Chinese ones, are analysed and compared from the viewpoint of environmental impacts. The case areas assessed are the South Karelia region in Finland and the city of Hangzhou in China. In addition to the differences in the composition of mixed waste, the case studies also exhibit other dissimilarities affecting the environmental impacts of waste management in the case areas, such as in the type of substituted energy production. In Publication III, the mixed waste management system in the city of São Paulo, Brazil is assessed from an environmental impacts point of view. Whilst the Brazilian waste management system shares similarities with the Chinese one; for example, the composition of mixed waste is rather similar in both case areas, they are still quite distinct from each other in terms of the surrounding environment. The waste management system assessed in Publication IV differs from the other waste management systems in this thesis. In Publication IV, the environmental impacts of CDW management are assessed. The geographical location of Publication IV is Finland. Even though the evaluated waste management system is distinct from the other case studies in terms of the assessed waste stream, LCA is once again applied and the same waste treatment methods, such as landfill disposal, incineration and material recovery, are employed in Publication IV.

This thesis focuses on the environmental aspect of sustainability. Therefore, economic and social aspects are not assessed herein, although they should also be taken into account when waste management systems are being developed towards a more sustainable direction. In Publications I, II and III, the environmental impacts assessed are global warming, acidification and eutrophication potentials, whereas in Publication IV, a total of 19 environmental impact categories, of which the primary focus is global warming and abiotic depletion potentials, are assessed.

1.4 Research process and outline of the thesis

Publications I-III were executed in the *Material Value Chains (ARVI)* programme (decision number – 379/143). The programme lasted over the three-year period of 2014 to 2016 and was funded by Tekes, the Finnish Funding Agency for Technology and Innovations (currently called Business Finland), as well as industry and research organisations. The primary objective of the ARVI programme was to promote the sustainable recycling of materials. Furthermore, the programme explored measures for supporting the local analysis of material flows from a systemic point of view, for instance, by applying LCA in the environmental impact assessment of waste management systems. Though the programme partners were Finnish, the programme itself explored material flows and waste management systems abroad, too. China and Brazil were the case countries of the programme. (Clic Innovation Ltd, 2019.)

Publication IV was executed in the *Life IP on waste – Towards circular economy in Finland* (LIFE-IP CIRCWASTE-FINLAND) project (project number LIFE15 IPE FI 004). The project began in 2016 and will last until 2023. Funding for the project was received from the EU LIFE Integrated programme, as well as from companies and cities. In general, the project promotes the efficient utilization of material flows, waste prevention, and new waste and resource management concepts in Finland. The primary objective of the study is to implement the national waste management plan, and thus direct Finland towards a circular economy. The project has been divided into 19 case studies having a more focused emphasis on a resource or waste stream, such as CDW, which was the waste stream assessed in Publication IV. (LIFE15 IPE FI 004, 2019.)

This thesis is classified as an article thesis (also referred to as a compilation thesis), meaning that it summarizes and outlines the main features and findings of four individual publications. The thesis also positions the publications into a broader context with the introduction and theoretical foundation sections. The connecting threads identified by comparing the results of the case studies with each other enables the author to draw further findings and conclusions in addition to those identified in the publications.

The thesis is organized as follows. Section 1 provides background information and an overview of the topic, introduces the objectives and scope of the thesis, evaluates limitations, and describes the research process and the outline of the thesis. Section 2 focuses on the environmental impacts of waste management. It provides an overview of waste management and associated environmental impacts globally, describes the methodological aspects and details of LCA, and discusses how LCA has been applied in the field of waste management in literature as well as what the results of previously published waste LCA studies indicate about the environmental performance of different waste management methods. Furthermore, Section 2 defines the concept of an operational environment from the standpoint of waste management. Section 3 introduces and describes in detail the case studies included in the thesis. An emphasis has been placed on describing the case areas and their waste management systems. Moreover, the information about the LCA studies, such as functional units and assessed environmental impact categories, is provided in the section. In Section 4, the main results and findings of the publications are first provided and then discussed in a broader context through an analysis of the differences among case studies to identify the influence of an operational environment on the environmental impacts of waste management. Section 5 summarizes the main findings and conclusions of the thesis and outlines recommendations for further research.

2 Theoretical foundation

2.1 Waste management and associated environmental impacts

Ever-growing waste generation is a global issue. To date, global waste generation has increased with an alarming pace as an outcome of population growth as well as urbanization and economic development in lower- and middle-income countries. Even though MSW is the most visible and noted waste stream, it is the fourth largest waste stream after industrial, agricultural, and construction waste streams, respectively, based on global average waste generation data (Kaza et al., 2018). However, since MSW management is under the responsibility of a municipality or other local authorities in most countries, monitored and verified data on waste volumes is more readily accessible. The main focus in this section is therefore on MSW streams.

In 2016, total MSW generation worldwide was estimated to be 2.01 billion tonnes. By 2030, it is estimated to increase to 2.59 billion tonnes. The increasing trend is forecasted to continue at least until 2050, with only a slightly slower pace: by 2050, the global MSW generation is forecasted to reach 3.40 billion tonnes annually. These forecasts assume that MSW generation will primarily grow in tandem with the GDP and population. Therefore, uncertainty is inherent in waste generation forecasts. Nevertheless, the increasing trend in global waste generation is evident according to the best currently available knowledge. The increasing waste generation poses the challenge of simultaneously managing the generated waste volumes in a controlled manner while decreasing the environmental impacts of waste management. The most visible adverse impact of poor waste management is littering. Plastic production has increased drastically over the last few decades. In 2016, 242 million tonnes of plastic waste were generated, which is equal to 12% of all MSW. Plastic waste littering is an outcome of an excessive production and consumption of plastic combined with negligent waste disposal. A low collection rate accelerates littering, and therefore one priority of a sustainable waste management system is extensive waste collection coverage. In high-income countries, waste collection rates approach 100%, whereas in middle- and low-income countries, the collection rates are approximately 50% and 40%, respectively. (Kaza et al., 2018.)

The waste management sector accounts for approximately 5% of annual greenhouse gas (GHG) emissions worldwide. In 2016, the GHG emissions generated in waste management were 1.6 billion tonnes. The World Bank has forecasted that without improvements in the sector, the annual CO_2-eq. emissions of waste management will increase to 2.6 billion tonnes by 2050. The main contributors to the GHG emissions of waste management globally are open dumping and landfill disposal without any landfill gas (LFG) collection systems, which are the predominant waste treatment methods worldwide: 33% and 40% of globally generated waste was openly dumped or disposed of in landfills, respectively. (Kaza et al., 2018.) Methane (CH_4), comprising typically approximately 25-60% of LFG during the first 35 years of landfill disposal (Damgaard et al., 2011), contributes to both global warming and photochemical ozone formation (Xing et al., 2013). The other main component of LFG, carbon dioxide (CO_2), typically comprising approximately 40-70% of LFG (Damgaard et al., 2011), is not included in the GHG inventories for the waste management sector, since the emissions originate from biogenic sources, such as paper and organic waste. In addition to LFG, leachate is also a major emission source in landfills. Since leachate contains harmful and toxic contaminants, such as heavy metals, the direct discharge of it can lead to adverse impacts for both people and the environment (Xing et al., 2013). Leachate also typically contains other contaminants, such as ammonia, chloride and phosphate (Manfredi and Christensen, 2009), which have an adverse impact on human health and ecosystems. In addition to LFG and leachate generation, the use of machinery, such as compactors, e.g. in landfill operations, also negatively affects the environment (Damgaard et al., 2011).

It has been estimated that 11% of MSW generated worldwide is incinerated in modern waste incineration plants (Kaza et al., 2018). The environmental impacts generated in the waste incineration process can be roughly divided into three kinds: those generated in the (1) pre-treatment of waste; (2) combustion process; and (3) treatment of process residues, such as ashes, wastewater and other residues. Environmental impacts are also generated indirectly; for instance, they occur in the manufacturing of chemicals used in the incineration process. The emissions to air contributing to global warming generated in the combustion process depend on the share of fossil carbon in the incinerated waste. Therefore, the proportion of plastics refined from crude oil in waste strongly influences the environmental impacts of waste incineration. In addition to the fossil CO_2 emissions of waste incineration, other adverse emissions are also generated in the incineration process. Sulphur dioxide (SO_2) and nitrogen oxides (NO_x) are examples of emissions having adverse impacts on both the environment and human health. Nowadays, other waste incineration emissions apart from CO_2 emissions are controlled with different flue gas cleaning technologies, such as scrubbing and filtration, in modern incineration plants. This is the reason why the environmental impacts of waste incineration have decreased dramatically over the past decades. (Damgaard et al., 2010.)

Approximately 19% of the MSW generated worldwide is recycled via composting, anaerobic digestion (AD) or other material recovery method (Kaza et al., 2018). Composting of organic waste can occur on a decentralized basis in so-called home composting units or on a centralized basis in composting facilities (Lundie and Peters, 2005). The environmental impacts of composting consist both of energy and diesel consumption in the process, and of emissions generated in the degradation process, such as N_2O, CH_4 and NH_3 emissions. The CO_2 emissions of composting are not included in GHG inventories and are indeed not considered as a GHG emission due to the biogenic origin of the treated waste. The environmental impacts of AD consist of energy consumption and possible CH_4 leakages. The digestate generated in the AD process requires further treatment and is typically composted. Therefore, further emissions are generated in the treatment of the digestate. (Bernstad and Jansen, 2012.) The remaining 30% of globally generated waste not disposed of in a landfill, incinerated or recycled is still openly dumped, i.e. disposed of in uncontrolled manner in terms of monitoring, let alone emission controlling (Kaza et al., 2018).

Awareness of the adverse impacts of waste and waste management on the environment is clearly a driving force for developing waste management practices. Another driver is resource scarcity and depletion. Since waste generation and composition go hand in hand with people's consumption habits, the excessive use of natural resources has turned the conception of waste as waste into one of waste as resources. This change of mindset applies to various waste streams, from mixed waste to more valuable CDW types, and is a cornerstone of waste policies. For instance, the Circular Economy action plan of the EU (European Commission, 2018), encompasses various measures for improving durability, reparability and recyclability of products, thus contributing to the most important action to diminish the environmental impacts of waste management: waste prevention. If in any case waste is generated, the Circular Economy action plan includes revised material recovery targets for different waste streams. For instance, 65% of MSW should be recycled by 2035 (European Commission, 2018). The recycling targets have so far been demonstrated as too ambitious for several member countries, such as Finland, let alone on a global scale. Therefore, the actions of waste policy in the EU can be considered as realistic worldwide only in decades to come, and quite possibly never.

2.2 Life cycle assessment

2.2.1 Principles of the methodology

LCA is an established and widely employed method for assessing the potential environmental impacts of products and systems (e.g. Guinée et al., 2011). The International Organization for Standardization (ISO) has standardized the method: the principles and framework of LCA have been defined in ISO 14040 (2006), and the requirements and guidelines in ISO 14044 (2006). The ISO standardized LCA has been acknowledged and adopted by academia as a tool to identify and assess the environmental performance of different products and systems.

As defined in ISO 14040 (2006), LCA consists of four main phases: goal and scope definition, inventory analysis, impact assessment, as well as interpretation. LCA can be regarded as an iterative technique, as demonstrated with two-directional arrows in Figure 2.1. For instance, the findings in an impact assessment phase might result in a revision in the life cycle inventory phase (EN ISO 14040, 2006).

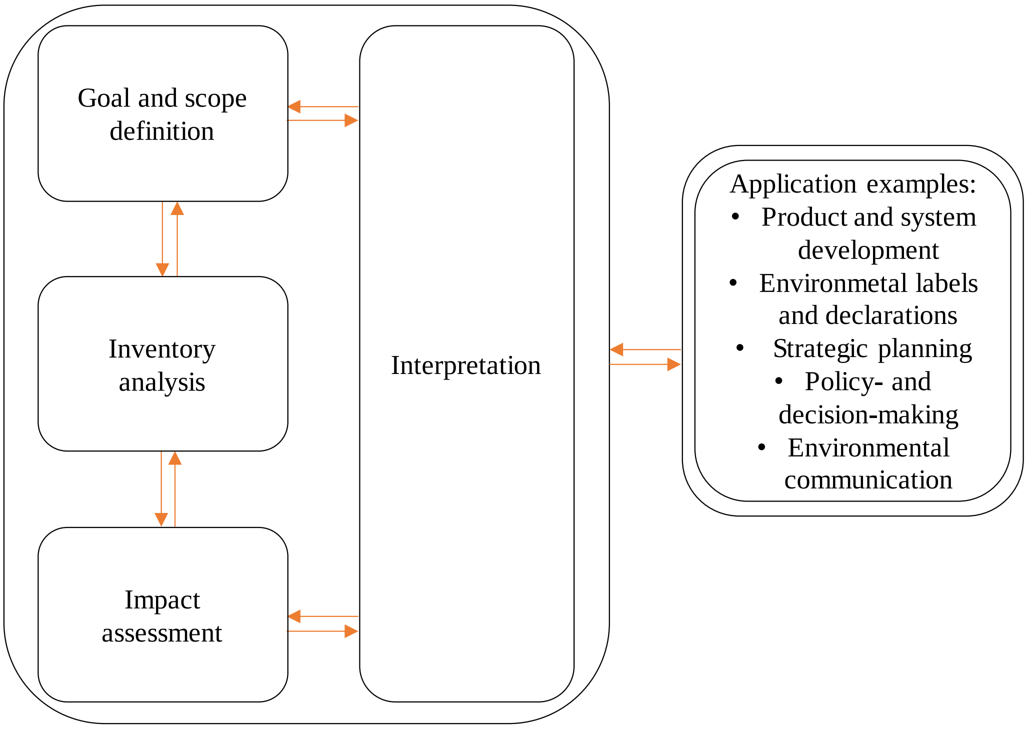

Figure 2.1. Main phases of LCA (EN ISO 14040, 2006).

In the goal and scope definition phase, the intended application of the study, audience of the study and reasons for conducting the study must be defined and must also inform the audience as to whether the results of the study are meant to be used in public comparative assertions. The following technical items must be specified in the goal and scope: the product system, function of the product system, functional unit, system boundary, allocation procedures, impact categories and methodology of impact assessment, data requirements, assumptions, limitations, initial data requirements, type of critical review (if any), and type and format of the report. The functional unit and system boundaries are critical items in the goal and scope phase for the interpretation and comparability of the results, while not understating the importance of the other items of the goal and scope; therefore, these items are further discussed. (EN ISO 14040, 2006.)

The functional unit describes the function(s) of the product or system in a quantified manner. With the functional unit, the reference to which the inputs and outputs of the study are related can be determined and quantified. The functional unit is highly important for the comparability of results. (EN ISO 14040, 2006.) Since the functional unit is a critical factor in LCA studies, particular attention must be paid when defining it, and the specification of it has been found to be problematic in academia. In the worst case scenario, an insufficiently defined functional unit or different functional units can cause various results for the same product system, which undermines the comparability of the results (Reap et al., 2008).

System boundaries define the unit processes included in the system. A main principle of LCA is that the potential environmental aspects and impacts throughout the life cycle of a product or a system are assessed; this is the ‘cradle-to-grave’ approach. Following this principle system, boundaries should include all relevant unit processes, starting from the acquisition of raw materials and ending in the end-of-life phase. (EN ISO 14040, 2006.) Waste LCA studies have a particular characteristic in terms of setting system boundaries. A ‘zero-burden approach’ is commonly applied in waste LCA studies. This approach makes the assumption that the environmental impacts of waste from previous life cycle phases; i.e. those occurring prior to the waste generation, are excluded from the assessment. (Ekvall et al., 2007.) By way of example, a set of hypothetical system boundaries is presented in Figure 2.2, depicting the cradle-to-grave and zero-burden approaches to demonstrate the differences between them. As presented in the figure, elementary flows cross the system boundaries. These flows encompass the material or energy flows entering or leaving the system boundaries. The elementary flows indeed form the basis for the life cycle impact assessment (EN ISO 14040, 2006).

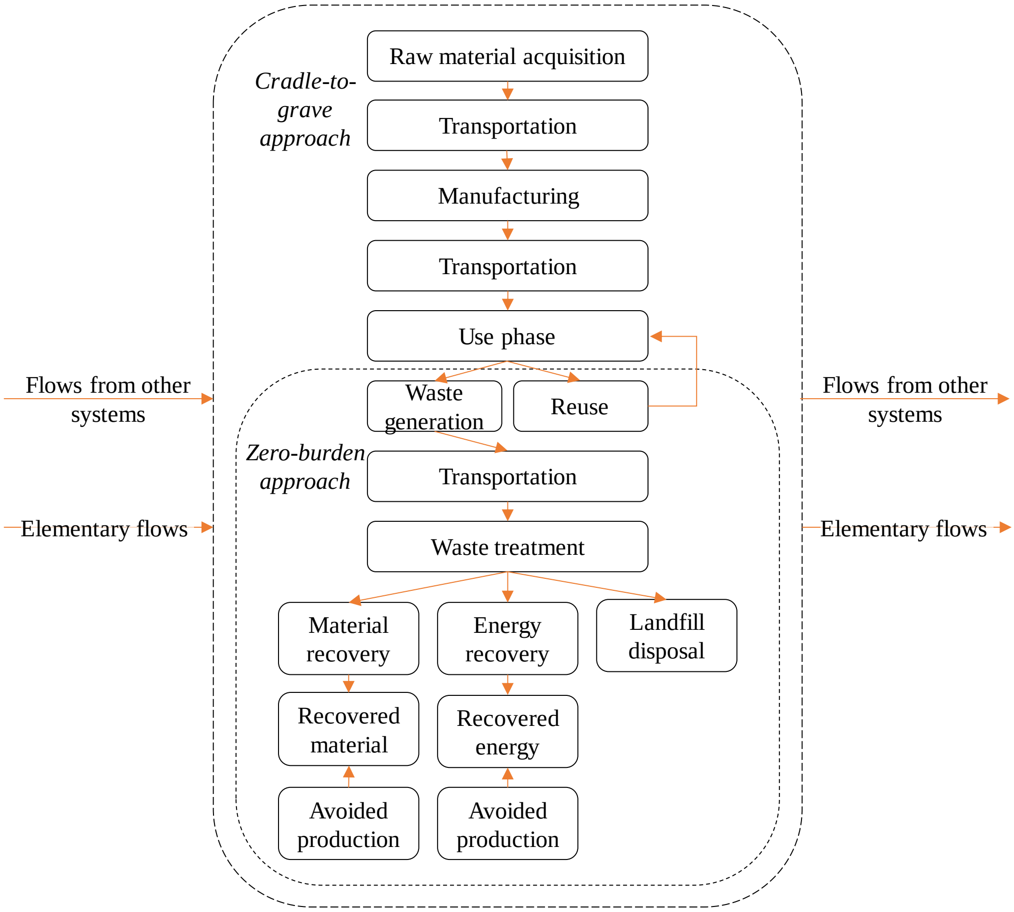

Figure 2.2. An example of system boundaries with the cradle to grave and zero burden approaches (adapted from Ekvall et al., 2007; EN ISO 14040, 2006).

The life cycle inventory (LCI) is the data collection phase of an LCA study. In the LCI phase, the inputs and outputs concerning the product system are collected throughout its life cycle. A challenge may arise when it is not clear which inputs and outputs are relevant to the system. According to EN ISO 14040 (2006), the inputs and outputs that are relevant in order to meet the goals of study should be accounted for in the LCI phase. Therefore, it is not necessary to collect all possible data concerning the system, but rather only the data relevant to the impact categories under investigation. The LCI phase is, by nature, an iterative process. New data requirements or limitations may emerge and be identified when some of the data has already been collected. This can result in the revision of the goal and scope of the study. Inventory data can be classified based on the source of data: primary versus secondary data. Primary data is data obtained by measurement or calculation based on direct measurements, whereas secondary data is from other sources, such as previously published literature and LCA databases. (EN ISO 14040, 2006.) If primary data is obtained within the product system, it can be regarded as site-specific data

(EN ISO 14067, 2018). Since primary data can also be obtained from other product systems, too, not all primary data is site-specific data, but all site-specific data is primary data.

The LCA standards, EN ISO 14040 (2006) and EN ISO 14044 (2006), do not provide recommendations or guidelines concerning data quality in LCA studies, whereas the standard for a carbon footprint calculation, EN ISO 14067 (2018), specifies the following requirements for data quality:

- Site-specific data should be applied to those unit processes that are most important.
- Primary data, which is not however site-specific data, should be applied to those unit processes for which site-specific data collection is not practicable.
- Secondary data should be applied to those unit processes for which primary data collection is not practicable, or to those processes that are least important.

The data quality recommendations found in EN ISO 14067 (2018) also include other recommendations, but these are the main differences for the different data types.

In the life cycle impact assessment (LCIA) phase, the significance of potential environmental impacts is evaluated by using the LCI results. Inventory data is associated with specific environmental impact categories and indicators. This is called ‘classification’. In ‘characterization’, an assigned inventory analysis result is converted with a characterization factor into the common unit of the category indicator. (EN ISO 14040, 2006.) Classification and characterization are mandatory elements of LCIA, whereas ‘normalization’, ‘grouping’ and ‘weighting’ are optional. In normalization, the magnitude of the impact category result is calculated relative to reference information, which can be, for instance, the total inputs and outputs for a given area. With normalization, it can be easier to comprehend the relative magnitude of each indicator result. Grouping involves sorting and possibly ranking the impact categories. Grouping is typically conducted based on value-based choices, which increases the subjectivity and uncertainty of a study. Owing to this, grouping is not commonly applied in scientific articles. Weighting aims at a better understanding of the magnitude and significance of the potential environmental impacts of the study by converting indicator results of different impact categories with numerical factors, which are value-based. Therefore, like grouping, weighting is not scientifically based and is thus rarely applied in scientific articles. (EN ISO 14044, 2006.) The connection between the LCI and LCIA phases is presented in Figure 2.3.

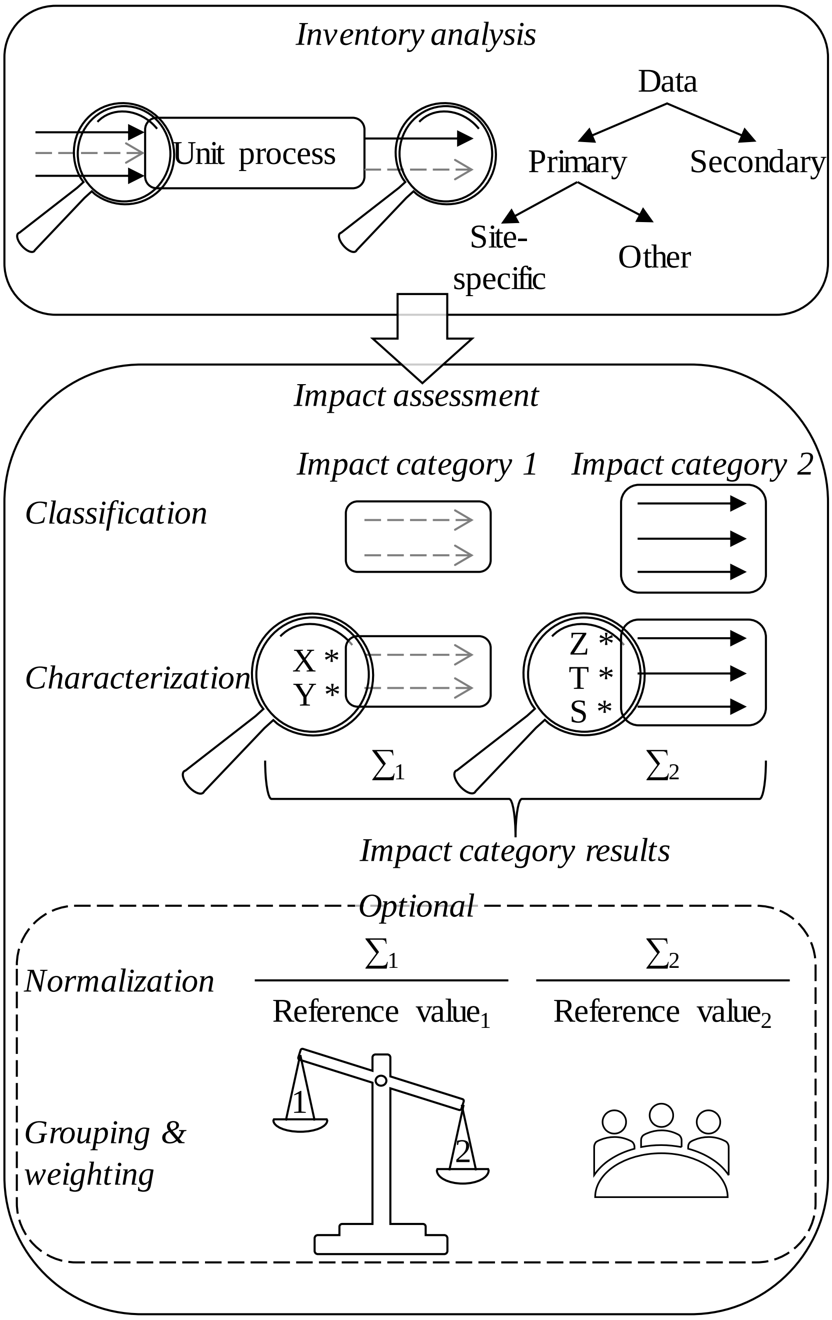

Figure 2.3. Connection between LCI and LCIA (EN ISO 14040, 2006; EN ISO 14044, 2006).

In the interpretation phase of an LCA study, the results of LCI or/and LCIA phases are summarized and discussed, and the basis for conclusions is formed. If conclusions are drawn solely based on the LCI results, the study can be regarded as an LCI study, but this should not be confused with the LCI phase of LCA. The results are also evaluated against the objectives and requirements of the study defined in the goal and scope phase. (EN ISO 14040, 2006). In the interpretation phase, significant issues of the study are identified; completeness, sensitivity and consistency of the study are evaluated; conclusions are drawn; and limitations and recommendation are evaluated and advanced.

Sensitivity analysis is a method used to estimate the uncertainty of an LCA study. Uncertainty may result from the choices made regarding methods, modelling and data. A sensitivity analysis may result in the revision of previous phases of the study if significant issues are identified. For instance, a sensitivity analysis may result in the inclusion of new unit processes and LCI data which have proven to be significant during the sensitivity analysis (EN ISO 14044, 2006.) Sensitivity analyses can be carried out using different techniques, for example by varying an input parameter and determining the influence on the result. This approach is known as 'local sensitivity analysis' in the scientific literature. 'Global sensitivity analysis', then, involves the procedure of assessing how much each input parameter contributes to the output variance. Thus, the variance and uncertainty of the overall results can be estimated with the latter sensitivity analysis approach. (Groen et al., 2017.)

2.2.2 Environmental impact categories and assessment

As mentioned above, the significance of potential environmental impacts is evaluated in the LCIA phase of an LCA study. Impact categories may be subdivided into 'midpoint' and 'endpoint' ones. Midpoint impact categories focus on specific environmental problems, such as climate change and eutrophication. Endpoint categories describe the final influence of environmental problems assessed with the midpoint categories on three areas of protection: human health, natural environment and natural resources. (EC-JRC, 2010.) The relationship between midpoint and endpoint impact categories is depicted in Figure 2.4.

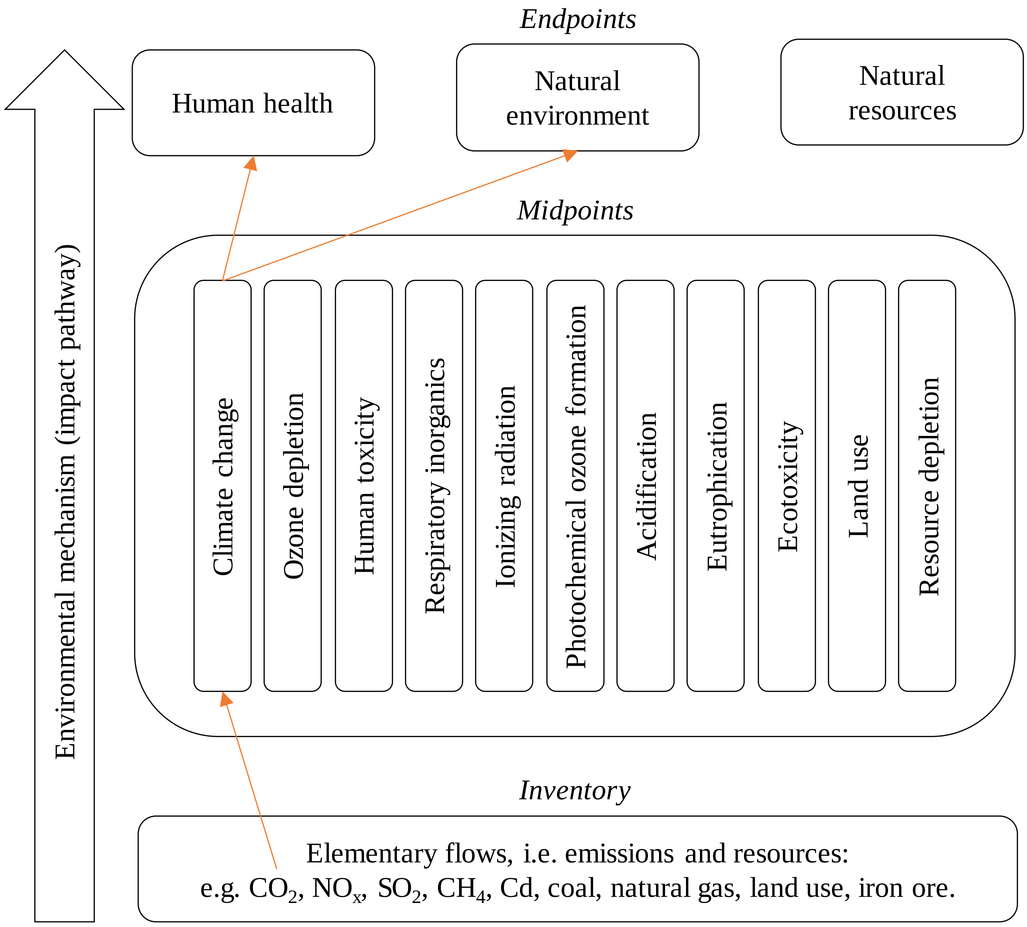

Figure 2.4. The connection of inventory data, and midpoint and endpoint impact categories in the environmental impact assessment (EC-JRC, 2010).

Impact categories should be selected in an LCA study in such a way that all relevant environmental issues related to the assessed product or system are covered. Exclusion of impact categories should always be justified. The International Reference Life Cycle Data System (ILCD) handbook for LCA studies (EC-JRC, 2010) recommends assessing the following midpoint impact categories in LCA studies: climate change/global warming potential (GWP), stratospheric ozone depletion, human toxicity, respiratory inorganics, ionizing radiation, photochemical ozone formation, acidification (land and water), eutrophication (land and water), ecotoxicity, land use, and resource depletion (minerals, fossil and renewable energy resources, water). Even though descriptions of impact categories somewhat vary depending on the selected impact assessment method, the above-mentioned environmental problems should be acknowledged in the selected impact categories based on the ILCD recommendations. The potential environmental impact categories under assessment depend on the goal and scope of the study as well as on the availability and completeness of LCI data. Therefore, the recommendations of the ILCD handbook cannot always be followed.

2.2.3 Multifunctionality

LCA as a research method encompasses several methodological and modelling approaches. Methodological approaches to multifunctional processes play an important role in LCA studies since waste treatment processes are commonly multifunctional. Therefore, the methodological aspects of multifunctionality are covered separately in this section. A multifunctional process refers to a process or system that performs more than one function. Multifunctionality can occur in two ways: (1) a process serves more than one purpose or (2) a process yields more than one output. Problems arise when determining the environmental impact of a single function or product. A waste incineration plant exemplifies multifunctionality from two angles. First, if both electricity and district heat are recovered in a waste incineration plant, there are two outputs, and therefore it can be regarded as a multifunctional process. Second, a waste incineration plant clearly has two functions: waste treatment and energy production, so the waste incineration plant can be considered as multifunctional process in this regard, too. (EC-JRC, 2010.)

Different approaches have been established for assessing and modelling multifunctional processes. The selection of the most appropriate approach depends on (1) the goal and scope of the study, (2) data availability and (3) the characteristics of the multifunctional process of the product. Ideally, the approach for solving the multifunctionality issue should already be determined in the goal and scope phase of an LCA study, because the approach affects the forthcoming LCI phase. (EC-JRC, 2010.) Allocation is one of the approaches. In allocation, the input and output flows of a process or a product system are divided between the product system under assessment and (an) other product system(s) (EN ISO 14040, 2006). Allocation is carried out based on a selected rule or criterion which should be primarily founded on the physical relationships between the products or functions. If such a rule or criterion cannot be established or if it is not representative, allocation can be carried out with a rule or a criterion based on other characteristics or qualities, such as economic value. The application of allocation is not recommended in LCA studies if it can be avoided (in order to diminish the uncertainty it causes) (EN ISO 14044, 2006). Therefore, different approaches which avoid allocation have been established.

The primary approach to avoiding allocation is subdivision of the multifunctional process. In this case, a multifunctional process is subdivided into two or more sub-processes, and LCI data is collected separately for those. (EN ISO 14044, 2006.) In practice, subdivision is not always possible, since dividing up LCI into different functions concerning ‘black box unit processes’, i.e. unit processes including more than one single-operation unit process, has been found to be too difficult and burdensome, or even impossible, in some cases (EC-JRC, 2010).

If the primary approach proves to be inapplicable, the secondary approach to avoiding allocation is system expansion or enlargement. In that case, the product system is expanded so that it includes the additional functions related to the co-products. (EN ISO 14044, 2006.) ‘Substitution’, also known as ‘crediting’ or the ‘avoided burden approach’, is a variant for system expansion. System expansion and substitution are equivalent modelling approaches leading to the same results mathematically. They do, however, exhibit differences in terms of meaning and interpretation. Substitution differs from system expansion in that instead of adding functions related to the co-products, the functions that are not required due to the production of the co-products are subtracted from the analysed system, i.e. credited. (EC-JRC, 2010.)

2.3 LCA of waste management systems

Being that LCA is an established and widely used method for assessing the potential environmental impacts of all kinds of products and systems, it has also been commonly applied in the field of waste management (Ekvall et al., 2007a). LCA has been deemed the most popular system analysis tool in the EU thus far (Pires et al., 2011). According to published LCA studies in the literature, the application of LCA in the field of waste management started in the mid-90’s (e.g. Barton et al., 1996). Since then, the volume of published waste LCA studies has increased significantly, as demonstrated in Figure 2.5. This trend is a distinct reflection of the increasing interest of the environmental impacts of waste management and of the adaptation of the ISO standardized LCA methodology as a method for evaluating environmental impacts (Laurent et al., 2014a).

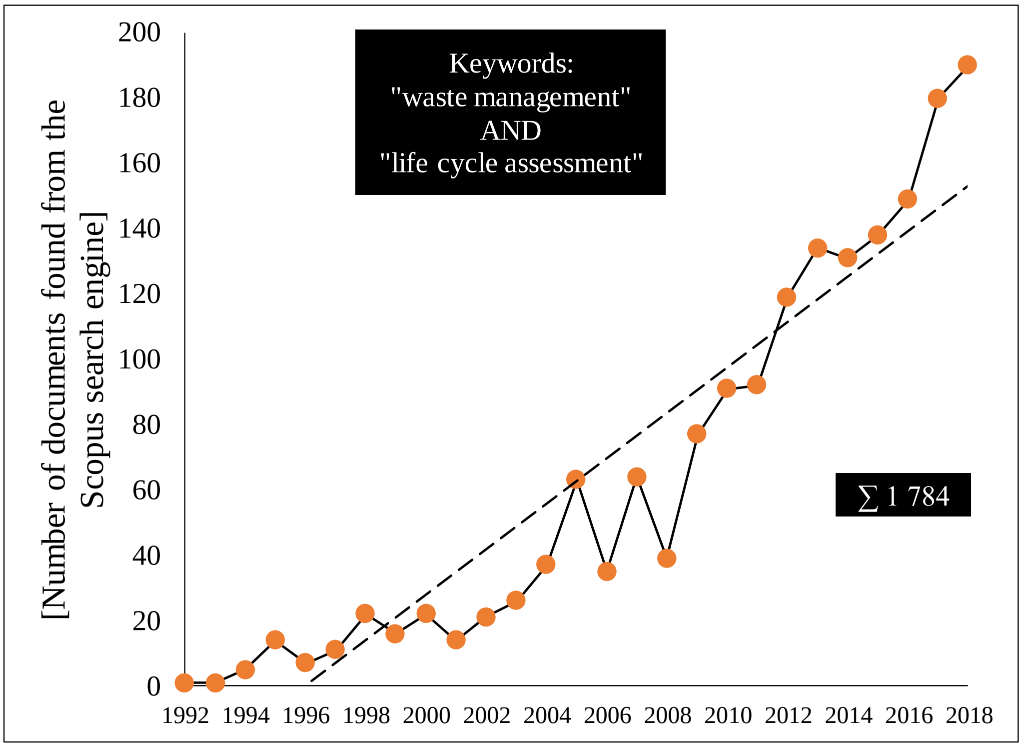

Figure 2.5. Volume of published waste life cycle assessment studies in the time horizon of 1996-2018, according to the Scopus search engine (Scopus, 2019).

As a result of the large volume of published waste LCA studies, several review articles analysing the previous literature on the topic have also been published (e.g. Cleary, 2009; Laurent et al., 2014a, 2014b). Laurent et al. (2014a, 2014b) conducted an extensive review study of the LCA of waste management systems through a critical analysis of 222 LCA studies published between the years 1995 and 2012. The geographical scope of the reviewed studies revealed that the majority of the LCA studies have been conducted in Europe. Waste LCA studies have also been conducted and published elsewhere, e.g. in Asia as well as North and South America, but with a lower intensity considering the quantity of published studies versus the size of the populations. For instance, only a few LCA studies have been conducted in South America. The limitations in the geographical scope of waste LCA studies create the need for a comprehensive understanding of the environmental impacts of waste management in different corners of the globe.

According to EN ISO 14044 (2006), all relevant impact categories for the system studied should be assessed. As mentioned above (see Section 2.2.2), the ILCD handbook for LCA studies (EC-JRC, 2010) recommends assessing the numerous midpoint-level impact categories, such as GWP, human toxicity, photochemical ozone formation, acidification and eutrophication. However, due to the limitations in the coverage of LCI data, all the recommended impact categories cannot always be considered in waste LCA studies. For example, in the review study of Laurent et al. (2014a, 2014b), fewer than 50% of the analysed LCA studies performed a complete LCIA taking all recommended impact categories into account. According to a review study of Cleary (2009), climate change/GWP, acidification potential (AP) and eutrophication potential (EP) are the most commonly assessed impact categories in waste LCA studies.

What makes LCA a useful tool for assessing the environmental impacts of waste management is that it widens the perspective assessment beyond the actual waste management system. This is important since the indirect environmental impacts of surrounding systems, such as energy production, can outweigh the direct impacts of waste management. (Ekvall et al., 2007.) This is relevant in cases in which energy and/or materials is/are recovered from waste. Therefore, waste LCA studies commonly encompass multifunctional processes. When the substitution method is applied for resolving the multifunctionality problem (as discussed in Section 2.2.3), the recovered energy and/or material is/are assumed to substitute for other energy and/or material production. This 'substituted' production can thus be considered avoided, which also leads to avoided environmental impacts. Avoided emissions are considered as negative ones in that case. This is a commonly applied approach for resolving the multifunctionality issue in waste LCA studies (Laurent et al., 2014a), so it is employed in the publications included in this thesis. Since with this approach, the inventory data of the substituted processes is regarded as negative flows, the overall environmental impact can even be negative if the avoided emissions surpass the direct emissions (EC-JRC, 2010).

The environmental impacts and performance of different waste treatment options are dependent on the high-variant parameters and factors of the local context, such as the composition of waste and of the energy supply mix (Laurent et al., 2014b). Optimal waste management strategy and systems vary according to the operational environment, and general conclusions about environmental performance of alternative waste treatment methods cannot always be drawn. Therefore, the environmental priority of material recovery over energy recovery over landfill disposal recommended in the waste hierarchy cannot always be taken for granted (Moberg et al., 2005). As a ground rule, decreasing landfill disposal in favour of material and energy recovery leads to environmental benefits, but the relationship between material and energy recovery is not so unambiguous in this regard for all waste fractions (Andreasi Bassi et al., 2017). In the extensive review study of Laurent et al. (2014a), which can be considered as a state-of-the-art review study about the LCA of waste management systems in previously published literature, the environmental performance of different waste treatment methods for organic waste, paper, plastic and mixed waste were evaluated. These waste types are also the most commonly assessed in waste LCA studies, according to the review study. The results of the comparison among waste treatment methods in terms of their environmental performances are presented in Figure 2.6. The comparative analysis was conducted as pair comparisons between different treatment methods. For instance, the environmental impacts of landfilling versus composting were determined.

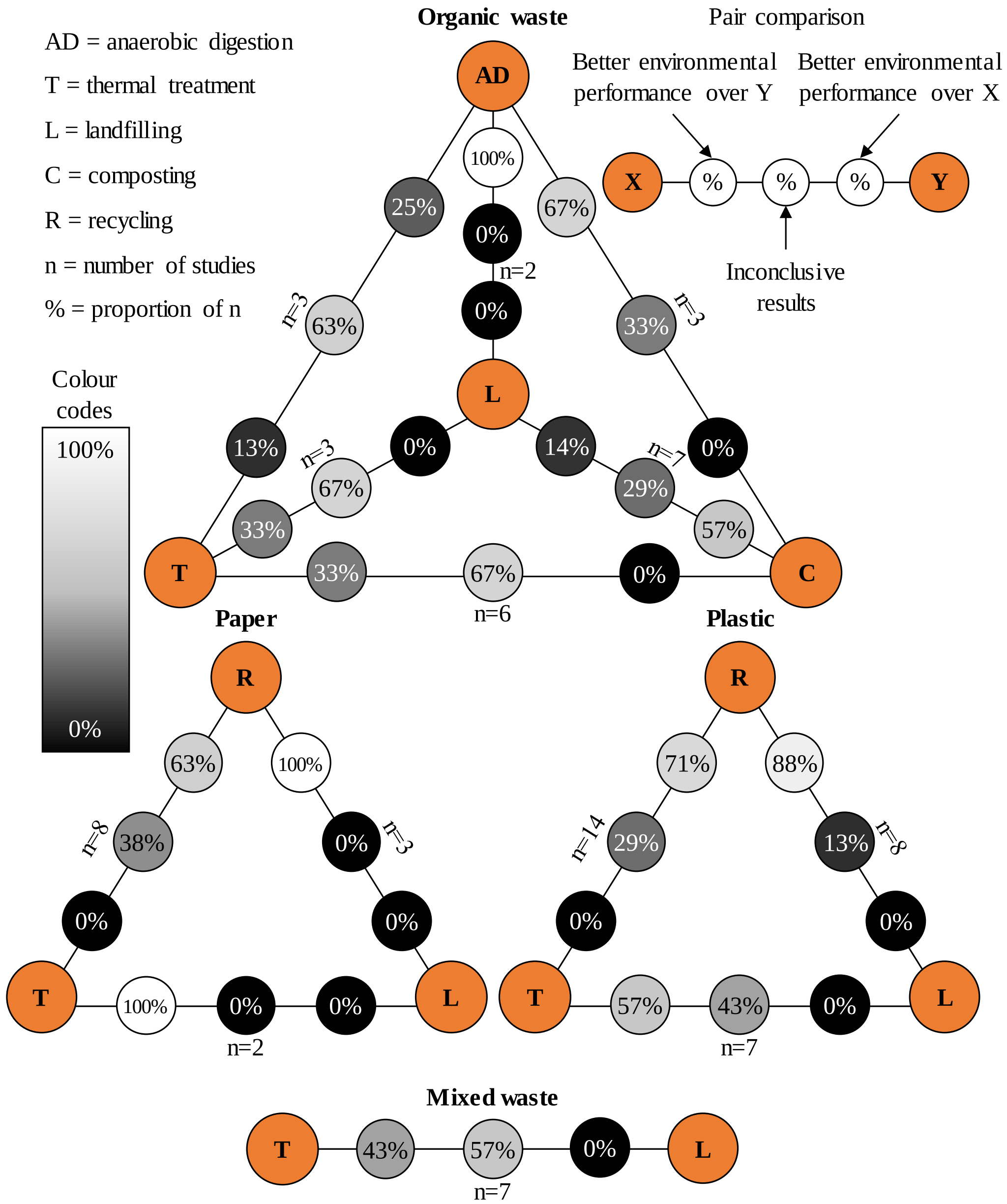

Figure 2.6. Comparison of the environmental performance of different waste treatment methods for organic, paper, plastic and mixed waste, according to the review study by Laurent et al. (2014a).

The environmental performances of the following treatment methods for *organic waste* were assessed in the review study of Laurent et al. (2014a): AD, composting, thermal treatment and landfilling. Based on the comparative analysis, the environmental impacts of landfilling are higher than those of the other treatment methods. The comparison also suggests that the environmental impacts of composting are higher compared to those of thermal treatment and AD, even though a significant portion of the studies included in the comparative analysis produced inconclusive results in this regard. The comparative analysis of the environmental impacts of recycling, thermal treatment and landfilling of *paper* resulted in an expected conclusion: landfilling causes higher environmental impacts than does recycling or thermal treatment. The comparison between recycling and thermal treatment suggests that the environmental impacts of recycling are lower than those of thermal treatment. It should, however, be noted that the comparison was not entirely unambiguous in this regard. Similar findings apply to *plastics*: landfilling of plastics has higher environmental impacts than recycling or thermal treatment, whereas the environmental impacts of recycling are lower than those of thermal treatment. As for *mixed waste*, the comparison between landfilling and thermal treatment indicates that the environmental impacts of landfilling are higher compared to thermal treatment. Though none of the studies included in the comparative analysis suggested that the landfilling of mixed waste has lower environmental impacts compared to thermal treatment, a notable share of the studies had inconclusive results in this regard, weakening the conclusion.

To sum up the findings of the study of Laurent et al. (2014a), no definitive consensus about the environmental performance of different waste treatment methods, apart from landfilling, has been reached in the literature. This is a clear signal of the case-specificity of waste LCA studies, as discussed previously (see Section 1.3). The comparative analyses of treatment methods for plastic and paper suggest that the priority order of waste hierarchy is in line with environmental impacts of the waste treatment methods.

Andreasi Bassi et al. (2017) assessed the environmental impacts of waste management when 1 tonne of household waste is treated in seven European countries, namely in Germany, Denmark, France, the UK, Italy, Poland and Greece. The results of their study revealed that household waste management leads to environmental benefits in most cases when the benefits of material and energy recovery are accounted for. Environmental benefits particularly originate from paper recycling. Additionally, metal and glass recycling both lead to environmental benefits, though to a lesser extent. Energy recovery can lead to either an environmental benefit or burden, depending on the energy source being substituted with the energy recovered from waste, indicating a strong influence of the substituted energy source on the overall environmental impacts of waste management. In addition to a national energy production scheme, the environmental impacts of waste management depend on the national context in terms of waste composition and the level of technology, for instance. The findings of Andreasi Bassi et al. (2017) are in line with previous literature, such as the studies of Laurent et al., (2014a) and Merrild et al. (2012), supporting the hypothesis that the environmental impacts of waste management vary substantially depending on the operational environment.

2.4 Operational environment

The term ‘operational environment’ in itself is a rather generic, even vague, term and is therefore subject to interpretation (De Witte and Marques, 2010). In the context of waste management, an operational environment can be considered to encompass all external variables and factors that exert an influence on a waste management system, and on the environmental impacts of waste management activities (Simões and Marques, 2011). Through a broad lens, an operational environment can refer to a country, whereas through a closer lens, an operational environment might be a region or a city. An operational environment is defined to encompass the following aspects of the surrounding environment in this thesis: socio-economic, political and legislative, technological, and geographical (see Figure 2.7).

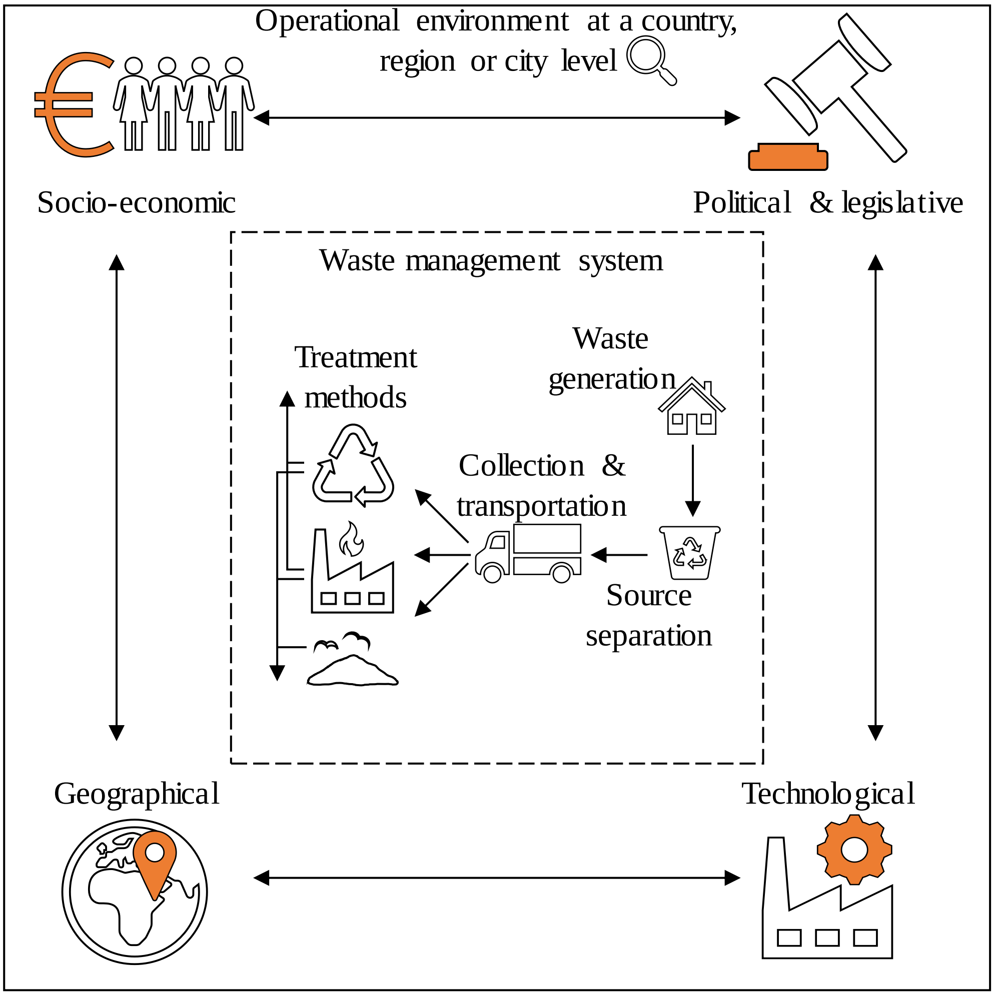

Figure 2.7. A depiction of the aspects of an operational environment in the context of waste management.

2.4.1 Socio-economic aspects

The relationship between waste management systems and socio-economic aspects of operational environments is rather widely discussed in previously published literature (e.g. Afroz et al., 2011; Aleisa et al., 2019; Fu et al., 2015; Grazhdani, 2016; Khan et al., 2016; Kumar and Samadder, 2017; Monavari et al., 2012; Yadav and Samadder, 2017). According to that literature, socio-economic aspects of an operational environment exerting an influence on the environmental impacts of waste management encompass factors such as education, occupation, income and household size. Socio-economic factors play a vital role in the earliest phase in the life cycle of waste: waste generation. Furthermore, socio-economic aspects have an influence on other phases and factors of waste management, such as the capability to invest in waste treatment facilities, unit wages in the waste management sector and the funding of waste management activities.

Kumar and Samadder (2017) assessed the relationship between waste generation and socio-economic parameters in a city in India and discovered an inverse correlation with waste generation and education level: an increase in educational level decreases the waste generation per capita. The study of Grazhdani (2016) supports this finding. Grazhdani (2016) discovered that a 1% increase in education level, in this case involving the percentage of population having an education level of high school and university, decreased annual waste generation per capita by 3 kg in a region in Albania. In turn, the correlation between education level and source separation efficiency is positive: as the education level increases, so does the efficiency of source separation alongside it (Grazhdani, 2016).

Khan et al. (2016) assessed the influence of socio-economic status, including factors such as education, occupation and income level, on the waste generation rate in a district in India. They discovered that particularly the proportion of plastics in household waste varies in accordance with socio-economic status, having a direct correlation: an increase in socio-economic status results in an increase in the share of plastic in household waste. Yadav and Samadder (2017) assessed the influence of income distribution on waste generation and composition worldwide and discovered that both waste generation and composition vary depending on the income distribution. A direct correlation between income level and waste generation was identified in the study, supporting the findings of previously published literature: the higher the income level, the higher the waste generation rate. As for the waste composition, they discovered that higher-income countries generate more paper, plastic and glass (i.e. packaging) waste, whereas lower-income countries generate more biodegradable waste. These finding are also in line with the expectations cited in the previous literature.

Monavari et al. (2012) studied the influence of various socio-economic parameters on waste generation and composition in a city in Iran. An inverse correlation between education level and waste generation was identified in the study, which is in line with the previous literature. Even though the waste generation rate typically increases alongside income level, the correlation between waste generation and income level was inverse in the study. Fu et al. (2015) studied the relationship between consumption and waste generation in China and discovered a significant direct correlation between GDP and waste generation. Vieira and Matheus (2018) investigated the relationship between waste generation and different socio-economic factors in Brazil. The results of their study revealed that waste generation is directly and strongly correlated with income level and distribution. Afroz et al. (2011) conducted a similar investigation in Bangladesh. Their results indicated that household waste generation is directly correlated with household size and income level, whereas an inverse correlation was identified between waste generation and concern about the environment as well as a willingness to source separate.

To sum up the findings of previously published literature in academia, the following conclusions can be drawn. Several socio-economic aspects and factors exercise an influence on waste generation and composition. A direct correlation between waste generation and income level can be considered as a prevailing consensus, even though contrary findings have also been published (however, to a lesser extent). The correlation between waste generation and education level is, on the contrary, inverse. In addition to education level, environmental awareness and attitudes are also factors which have been demonstrated as having an inverse correlation with waste generation. Since waste generation and composition are closely related, socio-economic factors also affect waste composition. A higher income level typically results in increased consumption of goods and commodities, which in turn leads to an increased share of packaging materials in mixed waste, indicating a direct correlation between the proportion of packaging materials in mixed waste and income level. Nonetheless, the correlation between income level and the proportion of organic waste in household waste is most commonly an inverse one. Therefore, a high proportion of organic waste in household waste is a typical characteristic of waste in lower-income countries.

2.4.2 Political and legislative aspects

As a waste management system is an inherent part of the surrounding operational environment, political and legislative aspects play an important role in the consideration of potential scenarios for waste management. These aspects may not always have a direct influence on waste management; they instead guide and direct waste management activities, and in this way indirectly affect environmental impacts.

Political and legislative aspects of an operational environment thus do not necessarily correlate with the environmental impacts of waste management. The waste hierarchy of the EU (European Commission, 2008) is an example of this. Even though the waste hierarchy is a backbone for waste policy and legislation in the EU, the priority order suggested in it is not always in line with the environmental impacts of waste management for all waste fractions, as discussed previously in this thesis (see Section 2.3). From a broader perspective, following the waste hierarchy leads to environmental benefits, and if deviation from the hierarchy would be an environmentally more favourable option, the deviation must be justified by LCA or by a corresponding comprehensive analysis of environmental impacts.

The waste policy of the EU steers the waste legislation of the member countries. In this way, the political and legislative aspects directly affect the environmental impacts of waste management. The landfill ban on organic waste in several member countries, such as Finland, is a representative example of a legislative aspect influencing waste management and the environmental impacts thereof. The manner in which the political and legislative aspects of an operational environment directly affect the environmental impacts of waste management is through regulations concerning emission control of waste treatment methods, such as landfilling (e.g. LFG and leachate collection and treatment) and incineration (e.g. combustion conditions and emission limits for pollutants). The operational environments differ distinctly from each other in this regard. For example, the emission control of waste incineration in Finland is regulated by both the EU's waste incineration directive and the country's waste incineration decree. The historical development of flue gas cleaning regulations of the EU exhibits a tightening trend in the development of emission control (Damgaard et al., 2010). In contrast, the flue gas control regulations in China include higher limits for pollutants than the limits of the EU's waste incineration directive (Wen et al., 2018).

The influence of political aspects of a regional- or city-level operational environment is also evident. Source separation regulations are an example of this. As advanced previously in this thesis (see 2.4.1), education level and environmental awareness have an influence on mixed waste composition. Source separation regulations and guidelines are closely related with these factors. Source separation regulations are typically dictated on a regional level. The comparison of Finland and China also exemplify this. In the South Karelia region of Finland, collections are arranged for mixed waste, biowaste, paper, cardboard, glass and metal. In the city of Hangzhou in China, separate collections are arranged for hazardous waste, food waste, recyclables and other waste. (Liikanen et al., 2017.) As a result of this difference and of other operational environment aspects, such as income level, waste composition differs distinctly between these two case areas.

2.4.3 Technological aspects

Technological aspects of an operational environment are closely related to the economic aspects (e.g. gross national product (GNP)). It is therefore reasonable to assume that the better the economic situation in an operational environment, the higher the level of technological development. Ciroth et al. (2002) have defined technological aspects to encompass all aspects other than those covered by temporal and geographical considerations. Regarding waste management, the technological aspects of an operational environment can be considered to include at least the aspects and factors related to infrastructure (e.g. roads), energy production and distribution, and industrial structure. These all influence the level of technological development or maturity in an operational environment. For instance, if industrial operators utilizing materials or energy recovered from waste are located within a case area, this has an impact on both the environmental and economic preferability of energy and material recovery methods over landfill disposal and transportation distances.

The technical level and maturity of waste treatment methods greatly influence the environmental impacts of waste management. LFG collection technology and efficiency are examples of this. The methane emissions of landfill disposal are in direct correlation with the efficiency of LFG collection. The collected LFG can be treated with various technologies. LFG may, for example, be used in combined heat and power production (CHP) with a gas engine, or upgraded to biomethane, which can be used as a transportation fuel, for instance. (Niskanen et al., 2013.) The technical development level of both LFG collection and treatment is important to the environmental performance of landfilling.

The technological development level in the emissions control of waste incineration is another example of the importance of the technological aspects of an operational environment. The process-specific flue gas emissions of waste incineration depend on the flue gas treatment technologies used and their efficiencies. Damgaard et al. (2010) studied the historical development in the flue gas treatment of waste incineration and discovered that the technological development in flue gas treatment has significantly reduced air emissions (e.g. particles and NO_x) and consequently diminished the environmental impacts of waste incineration. In addition to flue gas treatment, the technical development and/or maturity of waste incineration also concern(s) the efficiency of waste incineration, which plays a vital role regarding the environmental performance of waste incineration. A substantial improvement in the efficiency of energy recovery of waste incineration was also found in the study of Damgaard et al. (2010).

2.4.4 Geographical aspects

Different definitions for geographical aspects or conditions have been employed in LCA studies. Following the definition proposed by Aleisa et al. (2019), the geographical conditions of an operational environment may be divided into meteorological and geological conditions. In their study, the influence of these on the environmental impacts of waste management was evaluated, using LCA as a method. The conditions evaluated were precipitation, evaporation and geological formations, such as the physical geologic characteristics of aquifers. In the study of Ciroth et al. (2002), geographical differences in the LCI data of waste LCA studies were discussed. They define geographical differences as the differences between the conditions in the case area and in the geographical area covered by the LCI data. Geographical conditions may instead encompass climate, areas of protection, technological and natural infrastructure (such as electricity production and supply mix), rivers and waterfalls. For the purposes of this thesis, geographical aspects refer to climate and soil conditions, natural resources, ecosystems and landforms inherent in the surrounding environment, i.e. the operational environment.

The results of Aleisa et al. (2019) revealed that when case-specific data on meteorological and geological conditions are incorporated in waste LCA studies, more regionalized and accurate results are achieved, indicating a close relationship between the operational environment and waste management in this regard. Leachate and LFG generation are examples of phenomena influenced by climate conditions (e.g. Al-Yaqout and Hamoda, 2003; Bruce et al., 2018). The study of Yang et al. (2015) discovered that leachate generation in Chinese landfills with similar technology levels in different geographical locations is influenced by variations in climate conditions.

Energy production is highly dependent upon geographical conditions and aspects due to the uneven distribution of energy sources across the globe. Climate conditions also affect energy production. Renewable energy sources, such as wind, solar and hydro power, are a good example of the spatial variability of energy sources (e.g. Scaramuzzino et al., 2019). The connection between energy production and waste management is inherent because of treatment processes in which (1) energy is consumed (i.e. through pre-treatment and recycling processes) and (2) energy is substituted (i.e. with treatment processes in which energy is recovered from waste and this recovered energy substitutes for the other energy production). As the environmental impacts of the surrounding systems, e.g. energy production, can outweigh the environmental impacts of the waste management system, energy production typically has a strong influence in waste LCA studies (Ekvall et al., 2007b). As a rule of thumb, it can be stated that the "dirtier" the energy to be substituted is, the more environmental benefits are gained from the energy substitution. When assessing the environmental impacts of waste management, in addition to the direct emissions generated in waste treatment processes, the avoided emissions originating from the energy substitution should also be considered. Since geographical conditions have a direct influence on the type or mix of energy production in an operational environment, and since energy production has an influence on the environmental impacts of waste management, the connection between geographical aspects and waste management in this regard does not call for further justification.

In addition to energy production, geographical conditions, particularly climate conditions, also affect energy demand and consumption in an operational environment. Operational environments with cold climate conditions have an advantage in energy recovery from waste due to the need for district heat: the higher the energy recovery rate from waste, the more the environmental benefits. Since the electricity production efficiency of waste incineration is typically rather modest, approximately in the range of 20-30% owing to low steam temperatures (Münster and Lund, 2010), it is beneficial if district heat is also recovered. The heat production efficiency of waste incineration is considerably higher than the electricity production efficiency, approximately in the range of 60-80% (Münster and Lund, 2010), increasing the overall energy recovery efficiency of waste incineration. Thermal energy recovered from waste can also be utilized in industrial processes if industrial plants requiring process steam are located nearby (Lombardi et al., 2015).

3 Materials and methods

Four individual waste LCA case studies are included in this dissertation. As mentioned earlier, research in the field of waste management is commonly conducted using case studies, due to the inherent connection between a waste management system and the surrounding environment, i.e. the operational environment. Subsequently, waste management systems are highly case-specific. If the environmental impacts of waste management were assessed at a general level without considering the case-specific characteristics and features of the waste management system, it would considerably limit the utilization of LCA results in policy- and decision-making, as well as in other development measures. As LCA was the research method used in all four case studies, the structure of the case studies followed the main phases of LCA: goal and scope definition, inventory analysis, impact assessment and interpretation (see Figure 3.1).

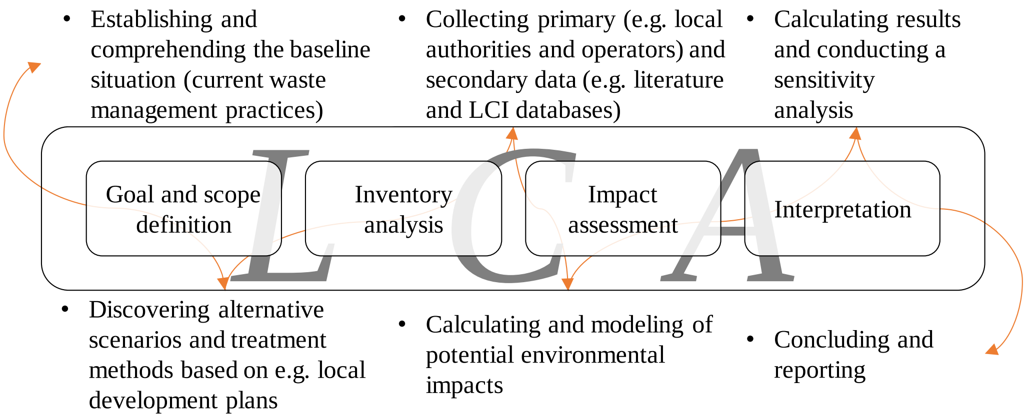

Figure 3.1. A depiction of the main phases of the case studies in line with the phases of life cycle assessment.

The baseline of each case study is the actual waste management situation in each case area for a reference year. The reference year was selected mainly based on data availability and completeness at the time when the LCI data collection was conducted. The preceding year was typically selected as the reference year in the case studies. The local and regional development actions and plans were considered when establishing scenarios for a case study. For instance, the waste management plan of the city of São Paulo was the starting point for determining alternative scenarios in the goal and scope definition of Publication III. Due to the iterative nature of LCA, the scenarios of the case studies were finally defined after the LCI data collection phase and during the preliminary LCIA. For instance, based on the preliminary LCIA results in Publication III, a rather low energy recovery rate from mixed waste incineration combined with the type of substituted energy production resulted in a surprisingly low environmental performance of waste incineration. Therefore, the possibility of recovering the energy content of waste in cement production, and subsequently substituting that for coal, which is the primary energy source used in cement kilns, was also considered in the scenarios, even though it was not initially included in them in the goal and scope definition phase. In the case studies, primary data was collected from local waste management authorities and operators, e.g. via interviews. If primary data was not available, secondary data from the literature and LCI databases was applied. The environmental impacts of waste management systems were calculated and modelled with an LCA dedicated software, GaBi (Thinkstep, 2019a), in all case studies. Sensitivity analyses were conducted to evaluate the influence of parameters, factors and modelling assumptions on the overall environmental impacts. The results of the case studies have been reported in the scientific literature. In the following sections, the case studies are described in detail.

3.1 Mixed waste management in Hangzhou, China

3.1.1 Description of the case area and waste management system

In 2017, the population of China was 1.386 billion inhabitants. China is classified as an upper-middle-income country: the gross national income (GNI) was 8 690 USD/capita in 2017. (World Bank, 2019.) Hangzhou is the capital city of the Zhejiang province in Southeast China (see Figure 3.2, where key information about Hangzhou is presented).

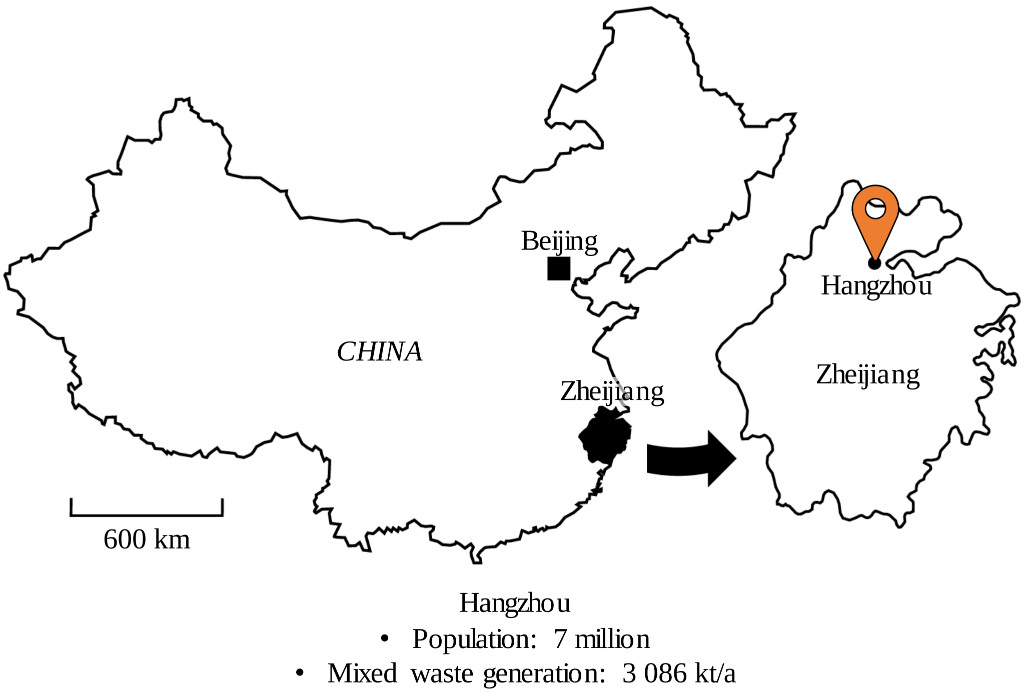

Figure 3.2. Background information about the city of Hangzhou (2013 as a reference year) (Publication I).

MSW generation in Hangzhou has substantially increased over the last couple of decades. Between the years 2003 and 2013, an average annual growth rate of 10% was detected in MSW generation in Hangzhou. Separate collection systems for hazardous waste, food waste, recyclables and other waste have been established and are part of the official MSW management system. In addition to this system, valuable recyclables are unofficially collected by residents and scavengers. The composition of mixed waste in Hangzhou differs from that in high-income countries such as Finland, due to, for example, the low level of source separation of organic waste. The most noteworthy differences are the higher proportion of organic waste and lower proportion of recyclables in mixed waste in Hangzhou. (Publication I.) The composition of mixed waste in Hangzhou is presented in Figure 3.3.

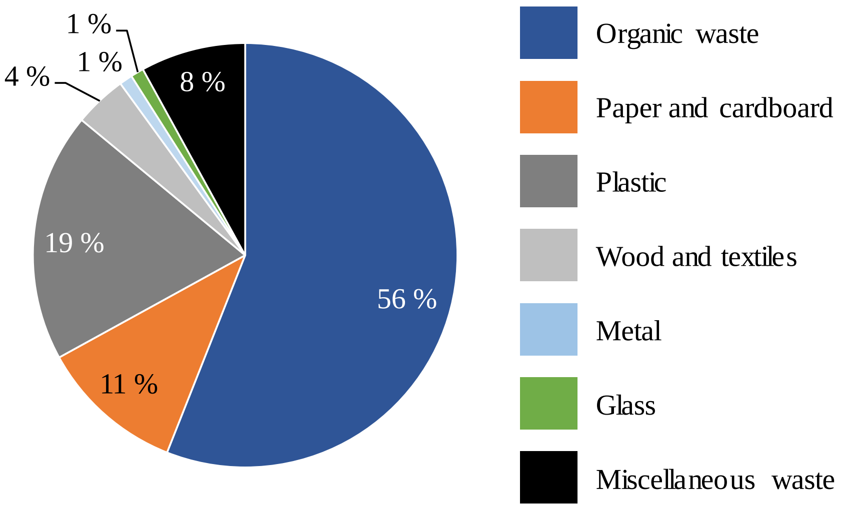

Figure 3.3. The composition of mixed waste in Hangzhou (Publication I).

Incineration and landfill disposal were the main treatment methods for mixed waste in Hangzhou in the reference year of the study, 2013. At that time, 58% of mixed waste was disposed of in landfills, and the remaining 42% was incinerated. In 2016, there were two municipal landfills (Tianziling and Liugongduan), and four waste incineration plants (Lvneng, Qiaosi, Yuhang and Xiaoshan) in Hangzhou. Three of these plants, namely Qiaosi, Yuhang and Xiaoshan, utilize fluidized bed technology. Coal is used as an auxiliary fuel in the fluidized bed boilers. However, the Lvneng plant utilizes grate technology instead, and coal is not used as an auxiliary fuel, according to the information received from the incineration plant. (Publication I.)

The mixed waste management system in Hangzhou was assessed in Publications I and II. Publication I focused solely on the Hangzhou case study. In Publication II, two very different mixed waste management systems were analysed, one which being the Hangzhou case study. Publication II aimed to identify key factors, i.e. processes and input parameters, in the LCA study. Publication II also brought attention to factors having only a minor influence on the total results of the LCA study. Such factors can play an important role in the consideration of options for simplifying the LCA of waste management systems. In Publication II, the possibility of using secondary data instead of direct data in order to diminish the workload of data acquisition of the LCI phase was advanced and discussed.

3.1.2 Functional unit and assessed impact categories

The LCA study was carried out in accordance with ISO standards 14040 and 14044 (EN ISO 14040, 2006; EN ISO 14044, 2006). The modelling for the study was done with GaBi LCA modelling software (version 6) (Thinkstep, 2019a) and CML 2010 (version April 2013); an impact assessment method for midpoint categories (Thinkstep, 2019b), was applied for impact assessment. The functional unit of the study was the treatment of mixed waste generated in Hangzhou in a year. At the time when the LCI data for Publication I was collected, the latest statistic about MSW generation in Hangzhou dated back to the year 2013. Therefore, 2013 was selected as the reference year for Publication I. Back then, 3 086 kt of mixed waste was generated in Hangzhou. The assessed environmental impact categories were global warming, acidification and eutrophication potentials.

3.1.3 System boundaries and scenarios

The objective of the Hangzhou case study was to determine the environmental impacts of the present MSW management system and compare those to alternative scenarios. Thus far, mixed waste has been co-incinerated with coal. In the alternative scenarios, mixed waste is mechanically treated, thereby producing refuse-derived fuel (RDF). When RDF is incinerated instead of mixed waste, the need for the auxiliary fuel, namely coal, can be decreased, since the lower heating value (LHV) of RDF is higher than that of mixed waste, and therefore the high combustion temperatures required for the destruction of toxic organic compounds can be more easily achieved. Incinerating RDF instead of mixed waste would also decrease mechanical problems, such as the corrosion and wearing associated with poor waste quality, in furnaces and auxiliary facilities. The objective was also to identify the most environmentally favourable treatment method for the organic reject generated in the mechanical treatment of mixed waste. The research questions of the Hangzhou case study were the following:

- What are the environmental impacts of mixed waste management at present (the baseline situation of the study)?
- What influence do RDF production and incineration have on the environmental impacts of mixed waste management compared to the baseline situation?
- What is the most environmentally sound treatment method for the organic reject generated in the mechanical treatment of mixed waste?

The study contains three main scenarios. Scenario 0, the baseline scenario, represents the actual mixed waste management system in 2013. As mentioned above, 58% of mixed waste was disposed of in landfills, and 42% was incinerated at that time. Scenario 1 represents a situation in which mixed waste incinerated in the fluidized bed boilers is mechanically treated, and the RDF produced, which has a higher LHV compared to untreated mixed waste, is incinerated at Qiaosi, Yuhang and Xiaoshan waste incineration plants. Thus, the need for the auxiliary fuel, coal, is decreased or avoided. In Scenario 0, mixed waste is incinerated in these plants using fluidized bed technology without mechanical treatment. Scenario 2 is similar to Scenario 1, apart from the waste incineration plants. Instead of incinerating RDF at Qiaosi, Yuhang and Xiaoshan plants, RDF would be incinerated at new, hypothetical, yet commercially available incineration plants with higher electricity production efficiencies. Scenarios 1 and 2 each have four sub-scenarios (referred to henceforth as Scenarios 1.1-1.4; 2.1-2.4) which represent different treatment options for the organic reject generated in the mechanical treatment of mixed waste. These different treatment options are landfill disposal (Scenarios 1.1 and 2.1), biodrying prior to incineration (Scenarios 1.2 and 2.2), AD prior to composting (Scenarios 1.3 and 2.3) and ethanol production prior to AD and composting (Scenarios 1.4 and 2.4). The system boundaries of the study are presented in Figure 3.4, and the mass flow of MSW is directed to different treatment options in Table 3.1.

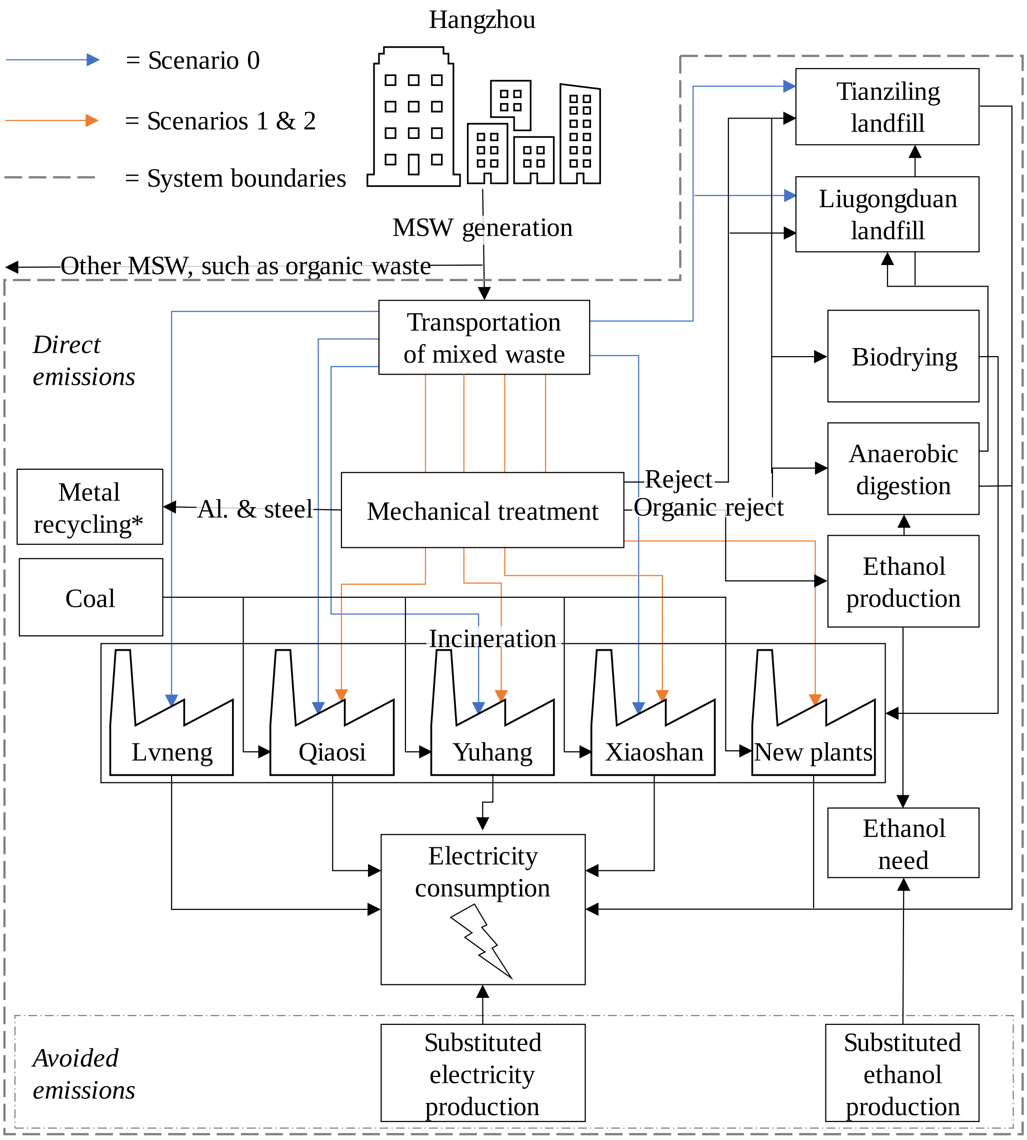

Figure 3.4. System boundaries of the Hangzhou case study (processes, in which both direct and avoided emissions are generated, are denoted with an asterisk (*)).

Table 3.1. Mixed waste mass flows directed to landfill disposal, incineration and RDF production in the Hangzhou case study.

Scenario	Mass flows of mixed waste [kt]							∑
	Landfill		Waste incineration or mechanical treatment					
	Liugongduan	Tianziling	Lvneng	Qiaosi	Yuhang	Xiaoshan	New plants	
0	367	1 432	204	411	256	416	-	3 086
1.1	13	1 432	204	490	343	604	-	3 086
1.2	293	1 432	204	394	276	487	-	3 086
1.3	13	1 432	204	490	343	604	-	3 086
1.4	13	1 432	204	490	343	604	-	3 086
2.1	-	1 388	204	-	-	-	1 494	3 086
2.2	249	1 432	204	-	-	-	1 201	3 086
2.3	-	1 388	204	-	-	-	1 494	3 086
2.4	-	1 388	204	-	-	0-	1 494	3 086

3.2 Mixed waste management in the South Karelia region, Finland

3.2.1 Description of the case area and waste management system

The population of Finland was approximately 5.5 million in 2017. Finland is classified as a high-income country: the GNI of Finland was 44 580 USD/capita in 2017. (World Bank, 2019b.) South Karelia is a region in South-East Finland consisting of nine municipalities. The population of the region corresponds to 2% of the total population in Finland (Regional Council of South Karelia, 2019). Background information about the South Karelia region is presented in Figure 3.5.

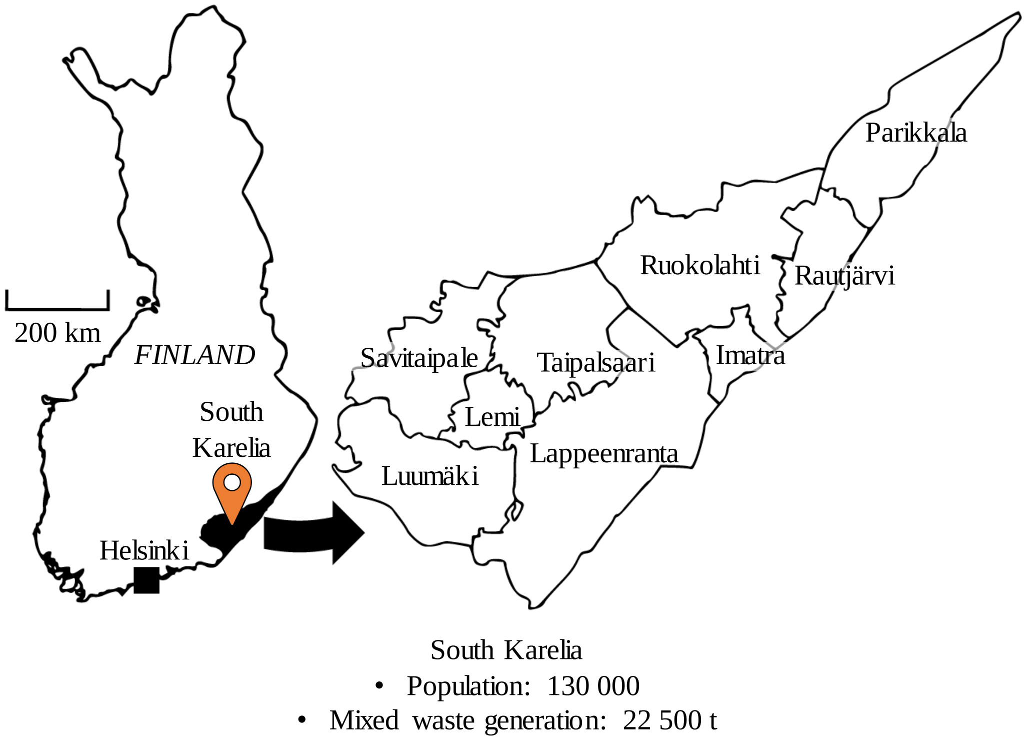

Figure 3.5. Background information about the South Karelia region (2012 as a reference year) (Publication II).

In 2012, the reference year of Publication II, approximately 22 500 t of mixed waste was generated in South Karelia. Separate collections systems for mixed waste, organic waste, paper, cardboard, glass and metal have been established in the region. Source separation of different MSW fractions is more efficient in South Karelia compared to Hangzhou, which is reflected in the composition of mixed waste (Figure 3.6). The most noteworthy difference is the lower proportion of organic waste in mixed waste. (Publication II.)

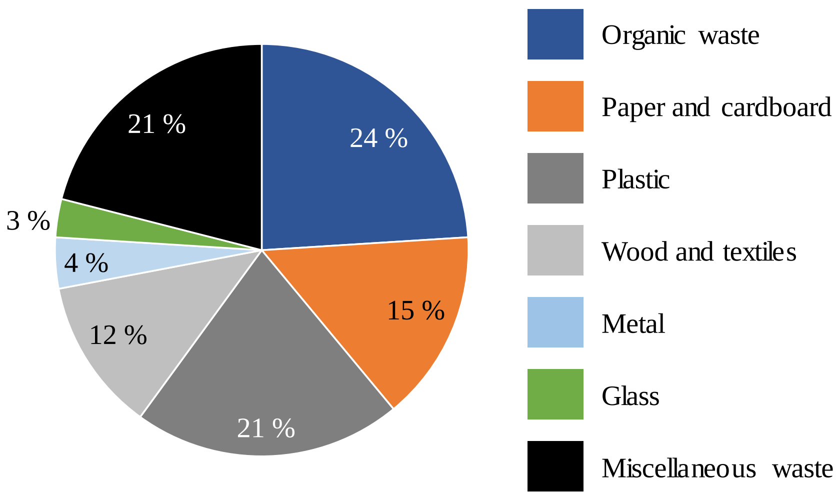

Figure 3.6. The composition of mixed waste in South Karelia (Publication II).

As in the Hangzhou case study, landfill disposal and incineration of mixed waste were assessed in Publication II. In 2012, the reference year of the study, all mixed waste generated in the region was disposed of in a landfill located in South Karelia, more precisely in the city of Lappeenranta. The incineration of mixed waste generated in the region started in 2013 and increased rapidly in stages due to the landfill ban on organic waste, which came into force in 2016; currently mixed waste is no longer disposed of in the landfill. Since no waste incineration plant is located within the region, mixed waste is transported to an incineration plant in a city located *c.* 220 km away from the region. (Publication II.)

3.2.2 Functional unit and assessed impact categories

The LCA of the mixed waste management system in South Karelia was conducted following the principles and requirements of ISO standards 14040 and 14044 (EN ISO 14040, 2006; EN ISO 14044, 2006). GaBi 6.0 LCA modelling software (Thinkstep, 2019a) was used in the study, and CML 2010 (version November 2010) (Thinkstep, 2019b) was applied for impact assessment. The functional unit of the study was the treatment of mixed waste generated in the region over one year. The reference year of the study is 2012 in order to identify the difference between landfill disposal and incineration from the standpoint of environmental impacts. In 2012, mixed waste was still disposed of in the landfill. During that year, 22 500 tonnes of mixed waste were generated in the region. The assessed environmental impact categories were once again GWP, AP and EP.

3.2.3 System boundaries and scenarios

The objective of the South Karelia case study assessed in Publication II was to determine the environmental impacts of two different management options for mixed waste: landfill and incineration. Additionally, the study aimed to find out, from an environmental impacts perspective, in which of the assessed waste incineration plants should mixed waste be treated. In the baseline scenario, mixed waste is disposed of in a landfill in Lappeenranta. In three sub-scenarios, mixed waste could be transported to three different waste incineration plants, which are located in separate cities outside the South Karelia region.

The South Karelia case study was the other mixed waste management system assessed in Publication II (in addition to the Hangzhou case study; see Section 3.1). Another objective was thus to identify key factors influencing the environmental impacts of mixed waste management in the LCA study. In addition to those, factors having only a minor contribution to the total environmental impacts were assessed in Publication II. The main objective of the South Karelia case study and Publication II can be broken down into the following research questions:

- What are the environmental impacts of mixed waste management at present (the baseline situation of the study in which all mixed waste generated in the region is disposed of in a landfill)?
- What influence does incineration have on the environmental impacts of mixed waste management compared to the baseline situation, and which waste incineration scenario is preferable from the environmental impacts point of view?
- What are the key factors affecting the environmental impacts of the mixed waste management system?
- Which factors have, by contrast, a minor influence on the total results?

The South Karelia case study contains two main scenarios. Scenario 0, the baseline scenario, represents the mixed waste management system in 2012 in the South Karelia region. At that time, all mixed waste generated in the region was disposed of in a landfill in Lappeenranta. In Scenario 1, mixed waste is incinerated in a waste incineration plant, instead of landfilled. Scenario 1 comprises three sub-scenarios (Scenarios 1.1-1.3). What differentiates these from each other is the waste incineration plant where mixed waste is incinerated. In Scenario 1.1, mixed waste is incinerated in an incineration plant in Riihimäki, a city located *c.* 220 km away from the region. In Scenario 1.2, mixed waste is incinerated in a waste incineration plant in Kotka, located *c.* 120 km from the region. In Scenario 1.3, the mixed waste is incinerated in a waste incineration plant in Leppävirta, *c.* 210 km from the region. In addition to the varying transportation distances, the incineration technology also varies among the waste incineration plants. The waste incineration plants in Riihimäki and Kotka employ grate furnace technology, and because of this, mixed waste requires only minor pre-treatment, such as removal of unsuitable large waste objects based on visual observation prior to the incineration. The waste incineration plant in Leppävirta instead employs fluidized bed technology, which requires pre-treatment of mixed waste. Therefore, prior to incineration, the mixed waste is treated mechanically, and RDF is produced from it. Both electricity and district heat are recovered in the incineration processes in all incineration plants assessed in the study. Additionally, in Scenario 1.2, the produced process steam is recovered and utilized in another industrial process near the waste incineration plant. In all incineration plants, the produced and recovered electricity is assumed to substitute for average grid mix electricity in Finland. However, the substituted district heat production varies among the sub-scenarios. In Scenarios 1.1 and 1.2, the produced district heat substitutes for heat produced from natural gas; whereas in Scenario 1.3, the produced district heat substitutes for heat produced from multiple energy sources: 72% of the substituted heat production is produced from biomass, 19% from plastic waste, 7% from heavy fuel oil (HFO) and 2% from coal. In Scenario 1.2, the recovered process steam substitutes for the process steam produced from natural gas. The types of substituted heat production in the scenarios were determined as realistically as possible, based on the information received from different stakeholders in the case areas. For instance, if a waste incineration plant had not been located in the case area in Scenario 1.3, the district heat would have been produced in a pulp and paper mill located in the area.

The system boundaries of the South Karelia case study are presented in Figure 3.7. Since the scenarios compare two different treatment methods for mixed waste generated in the region, and not a combination of them, and since all mixed waste is either incinerated or landfilled, the mass flows of mixed waste are the same in each scenario, i.e. 22 500 tonnes are treated in each scenario.

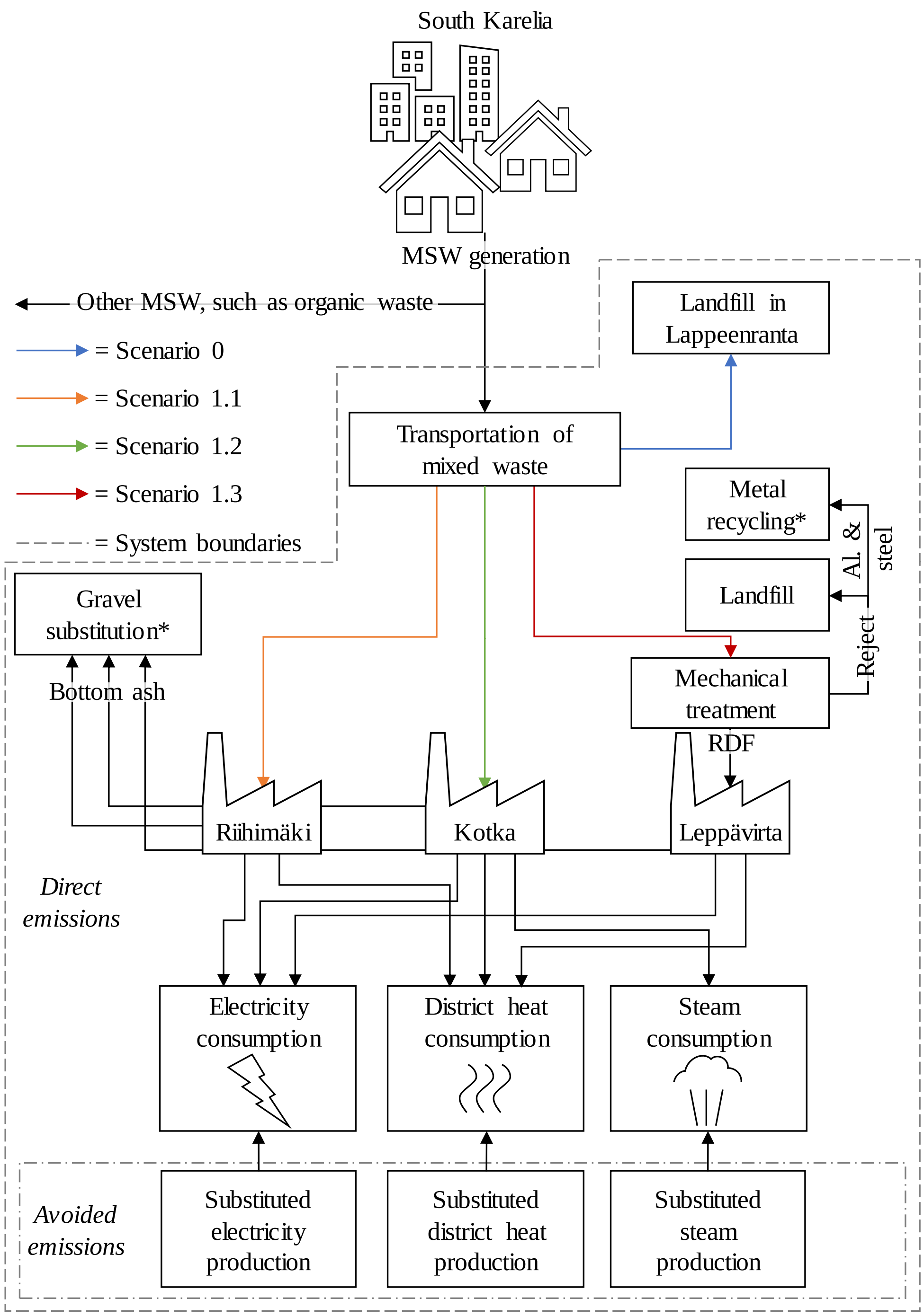

Figure 3.7. System boundaries of the South Karelia case study (processes, in which both direct and avoided emissions are generated, are denoted with an asterisk (*)).

3.3 Mixed waste management in the city of São Paulo, Brazil

3.3.1 Description of the case area and waste management system

The population of Brazil was approximately 209 million in 2017. The World Bank has classified Brazil as an upper-middle-income country: the GNI was 8600 USD/capita in 2017. (World Bank, 2019c.) São Paulo is the capital city of the State of São Paulo in Southeast Brazil (see Figure 3.8, where background information about São Paulo is presented).

São Paulo

- Population: 11-12 million, depending on the source
- Mixed waste generation: 3 800 kt

Figure 3.8. Background information about the city of São Paulo (2015 as a reference year).

Brazil is the world's fourth largest generator of MSW after China, the United States of America and India, respectively. Landfill disposal has been the main treatment method for mixed waste in the city of São Paulo thus far. São Paulo will be facing tough challenges in the future when simultaneously developing and modernizing its MSW management system and managing ever increasing MSW volumes. Source separation of different waste fractions is rather inefficient in São Paulo. This is reflected in the composition of mixed waste, which is dominated by the share of organic waste (see Figure 3.9). Recyclables are collected both formally and informally. Though separate collection has been established for recyclable waste fractions, such as plastics, cardboard and metal, the formally collected recyclables comprise only approximately 1% of the total MSW generation. As is common in middle- and low-income countries, an unofficial waste management sector plays a noteworthy role in São Paulo. Unofficial individual collectors and collector organizations collect valuable recyclables, such as metals and plastics. (Publication III.)

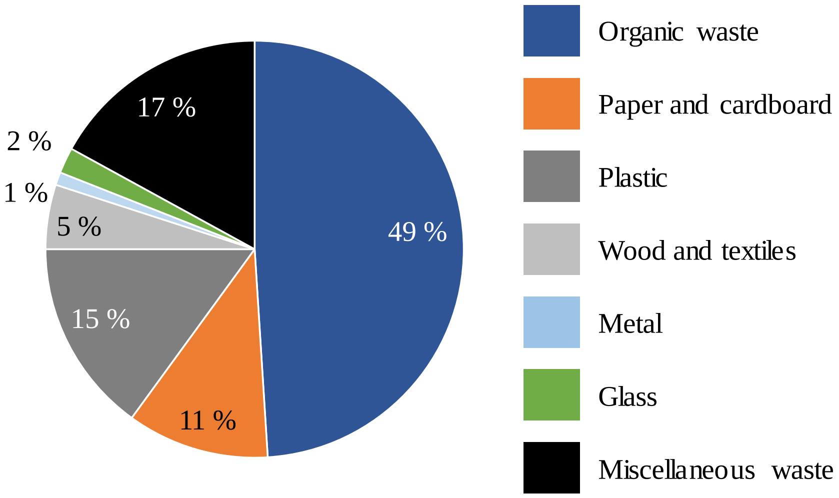

Figure 3.9. The composition of mixed waste in São Paulo (Publication III).

MSW management services in the city of São Paulo are contracted out to private companies, which have their own collection and transportation fleets and waste management facilities. The mixed waste generated and officially collected in São Paulo is disposed of in two landfills in the city: CTL and CTVA Caieiras landfills. At the time the LCI data collection for Publication III was conducted in 2016, there were no other waste treatment facilities (e.g. composting plants) in operation in the city. Therefore, all mixed waste officially collected in São Paulo was disposed of in the landfills. Although all mixed waste is currently landfilled, São Paulo has plans to develop MSW management in the city in the future. One of the main priorities of the development plans is to reduce the volume of mixed waste disposed of in landfills. Since approximately half of the mixed waste is composed of organic waste, the separate treatment of it could play an important role in achieving this objective. Therefore, São Paulo has plans to establish source separation and separate treatment of organic waste. Composting (including home composting) and AD have been proposed as potential treatment methods for organic waste in the city. In 2016, when the LCI data for Publication III was collected, no composting or AD was in operation. At that time, there were some small-scale initiatives to promote home composting, but these were not widespread. In addition to these treatment methods, mechanical-biological treatment (MBT) was proposed as a potential treatment method for mixed waste in the city's development plans. (Publication III.)

3.3.2 Functional unit and assessed impact categories

The LCA of the mixed waste management system in the city of São Paulo was carried out following the principles and requirements of ISO standards 14040 and 14044 (EN ISO 14040, 2006; EN ISO 14044, 2006). GaBi LCA modelling software (version 7.0) (Thinkstep, 2019a) was used for modelling in the study, and CML 2001 (version April 2015) was used for impact assessment (Thinkstep, 2019b). The functional unit of the study was the treatment of mixed waste generated and officially collected in São Paulo in a year. The reference year of the study was 2015. That year, 3 800 kt of mixed waste were generated and officially collected in São Paulo. The assessed impact categories were again GWP, AP and EP, because the required and sufficiently extensive LCI data was available for assessing those particular impact categories.

3.3.3 System boundaries and scenarios

The objective of the São Paulo case study was to evaluate the potential environmental impacts of different waste management alternatives to find a pathway towards a more environmentally sound manner for mixed waste treatment in the city. The study laid an emphasis on different treatment alternatives for organic waste, since organic waste comprised such a high proportion of the mixed waste. The research questions of the case study were as follows:

- What are the environmental impacts of mixed waste management in the city of São Paulo at present (using 2015 as a reference year)?
- Towards which direction should the mixed waste management system be developed, from the viewpoint of environmental impacts?

The scenarios assessed in the study represent potential treatment methods for mixed waste in São Paulo and take into account the development plans of the city. The study contains five main scenarios. In Scenario 0, the baseline scenario, 100% of generated and officially collected mixed waste is disposed of in landfills. Scenario 1 combines landfill disposal with home composting: 5% of organic waste, equal to 2.5% of the total mixed waste, is home composted, and the rest is landfilled. The 5% home composting rate was deemed plausible, although the development plans of the city include notably higher targets for home composting. In Scenario 2, the home composting of organic waste is complemented by the separate collection and treatment of organic waste: 20% of organic waste, equal to 9.8% of mixed waste, is source-separated and collected for further treatment. The 20% separate collection rate was employed based on the development plans of the city regarding the establishment of new organic waste treatment facilities. Scenario 2 contains two sub-scenarios, according to the treatment plant type. In Scenario 2.1, the separately collected organic waste is treated in composting plants; whereas in Scenario 2.2, the organic waste is treated in AD plants. The residual mixed waste is disposed of in landfills in Scenarios 2.1 and 2.2.

In Scenarios 3 and 4, in addition to the development steps taken in Scenarios 1 and 2, the MBT of the residual mixed waste is employed. Of the residual mixed waste, which is 17.6% of the total mixed waste, 20% is treated in MBT facilities, and the rest is landfilled. Since São Paulo's development plants do not have specific targets for the MBT of mixed waste, a realistic mid-term MBT capacity was assessed in Scenarios 3 and 4. What differentiates these scenarios from each other is the treatment method for the RDF produced. This is treated in waste incineration plants in Scenario 3, and subsequently electricity is recovered from waste. Due to the geographical location of São Paulo, there is no need for district heat, so only electricity is recovered from waste. The produced electricity substitutes for the average electricity grid mix in Brazil. In Scenario 4, the RDF produced is utilized in cement production as energy. In that case, the RDF substitutes for coal, which is typically the primary fuel in cement production. This enables avoided mining, processing and combustion of coal. As in Scenario 2, Scenarios 3 and 4 also each have two sub-scenarios (Scenarios 3.1 and 3.2; 4.1 and 4.2) representing different treatment options for the source-separated organic waste as well as for the organic reject generated in the mechanical treatment of mixed waste. In Scenarios 3.1 and 4.1, the treatment option is composting, whereas in Scenarios 3.2 and 4.2, the treatment option is AD. The system boundaries of the study are demonstrated in Figure 3.10, and the mass flows of MSW directed to the different treatment options may be found in Table 3.3.

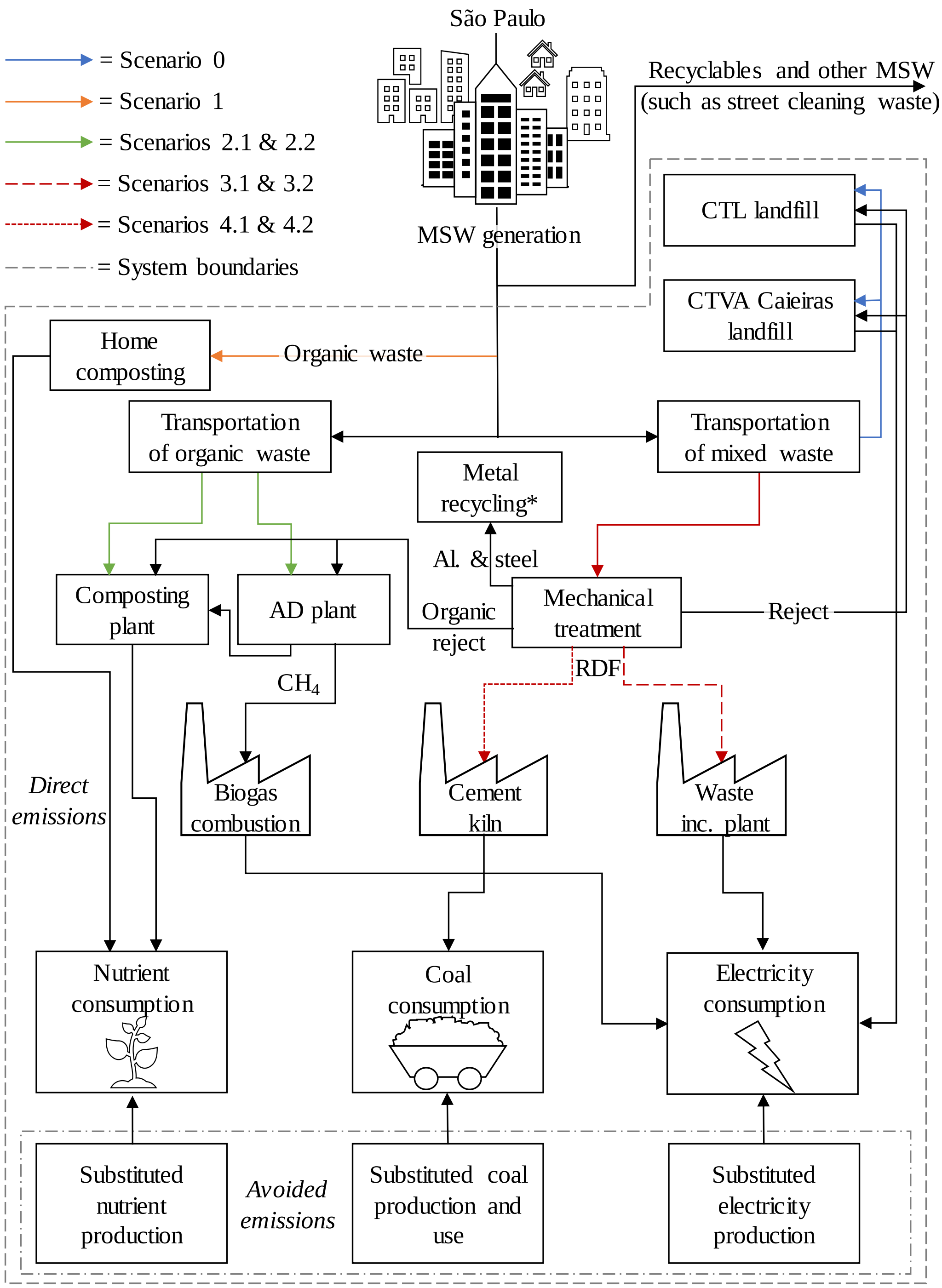

Figure 3.10. System boundaries of the São Paulo case study (processes, in which both direct and avoided emissions are generated, are denoted with an asterisk (*)).

Table 3.3. Mixed waste mass flows directed to different treatment options in the São Paulo case study.

Scenario	Mass flows of MSW [kt]							Σ
	Landfill		Treatment of organic waste			Incineration plant	Cement kiln	
	CTL	CTVA Caieiras	HC	CP	AD			
0	2 274	1 527	-	-	-	-	-	3 800
1	2 218	1 489	93	-	-	-	-	3 800
2.1	1 995	1 340	93	372	-	-	-	3 800
2.2	1 995	1 340	93	-	372	-	-	3 800
3.1	1 596	1 072	93	372	-	667	-	3 800
3.2	1 596	1 072	93	-	372	667	-	3 800
4.1	1 596	1 072	93	372	-	-	667	3 800
4.2	1 596	1 072	93	-	372	-	667	3 800

3.4 Construction and demolition waste management in Finland

3.4.1 Description of the case area and waste management system

The geographical location of Publication IV is Finland. The geographical scope has not been defined more accurately at a regional or city level because the study was not based on an actual case study. This distinguishes Publication IV from the other publications included in this thesis. Since the geographical location of the study is the same as that in Publication II, i.e. the country of Finland, the main characteristics of Finland, e.g. population, are described above in 3.2.1. The environmental impacts of utilizing CDW fractions as raw materials for WPCs were assessed and compared to the baseline situation, in which CDW fractions are treated conventionally. The CDW fractions assessed were wood, plastic, mineral wool and plasterboard. The recipes of WPCs assessed in the study are presented in Figure 3.11.

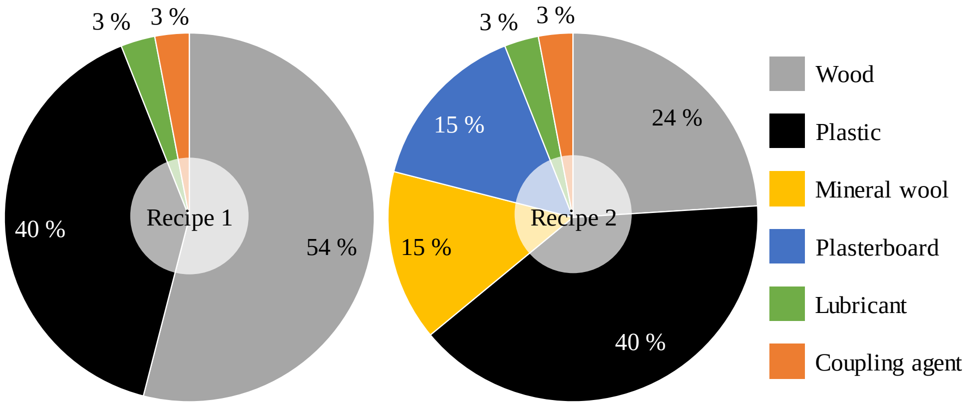

Figure 3.11. The recipes of wood polymer composites assessed in the study.

In Finland, approximately two million tonnes of non-hazardous CDW is generated annually (Dahlbo et al., 2015). The material recovery rate of non-hazardous CDW is currently 58% (Salmenperä et al., 2016), which is notably lower than the material recovery target of the EU: 70% by 2020 (European Commission, 2016). This calls for further action to develop the material recovery of CDW in Finland, and WPC production has been identified as a material recovery method for CDW. Extrusion and injection moulding are the most commonly used production technologies for WPCs. In the study, WPCs are produced with extrusion production technology. In addition to the extrusion or injection moulding, the production process includes other process phases, such as pre-treatment of raw materials. A simplified depiction of the WPC production process is presented in Figure 3.12.

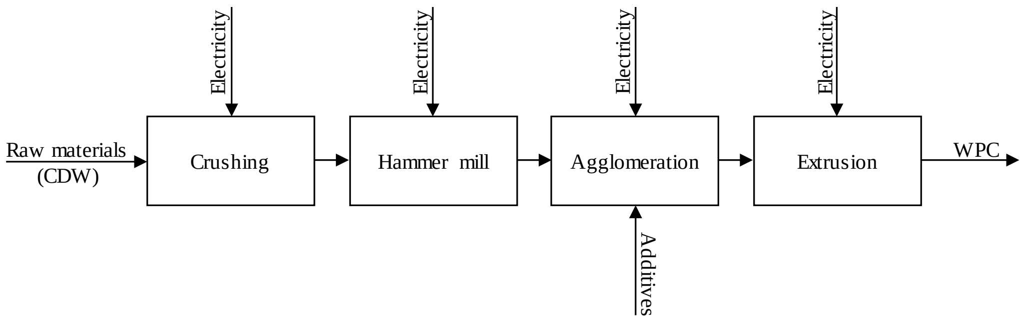

Figure 3.12. The production process of wood polymer composites.

3.4.2 Functional unit and assessed impact categories

The LCA of CDW management in Publication IV was conducted following the principles and requirements of ISO standards 14040 and 14044 (EN ISO 14040, 2006; EN ISO 14044, 2006). The GaBi LCA modelling software version 8.7.0.18 (Thinkstep, 2019a) was used in the study. Recipe 2016 v.1.1 (midpoint and hierarchist time) was applied in impact assessment (RIVM, 2018; Thinkstep, 2018). The functional unit of the study was the treatment of 940 kg of CDW, which corresponds to 1000 kg of WPC produced according to the recipes assessed in the study. Therefore, 1 tonne of produced WPC is the reference flow of the study. The reference year of the study is 2017. In total, 19 impact categories were assessed in the study. The study focused particularly on the climate change and fossil resources depletion impact categories because they were deemed the most relevant in the goal and scope of the study.

3.4.3 System boundaries and scenarios

The aim of the study (Publication IV) was to find out how the environmental impacts of CDW management might vary when the CDW fractions are utilized as raw materials in WPC production instead of being treated conventionally. Since the emphasis of the study was laid on WPC production, which is considered as an intermediate step between landfilling or energy recovery and mono-material recovery, the study did not consider the possibility of recovering all CDW fractions separately. Therefore, the assumption was that CDW fractions which would otherwise be disposed of in a landfill or incinerated should primarily be used in WPC production. The research questions of the study were the following:

- What are the environmental impacts of CDW management at present compared to a situation in which the CDW fractions are used as raw materials for WPCs?
- What CDW fractions should be favoured as raw materials for WPCs?
- What influence does material substitution (i.e. the WPCs produced substituting for conventional materials) have on the total environmental impacts?

The study has five main scenarios. The baseline scenario, Scenario 0, represents the actual situation of CDW management in Finland in 2017. Wood and plastic fractions of CDW are incinerated in a waste incineration plant assumed to be located 120 km away, and mineral wool and plasterboard fractions are disposed of in a landfill assumed to be located in the same region as the CDW fractions are generated. The energy, i.e. district heat and electricity, recovered from wood and plastic substitutes for the average district heat and electricity production in Finland. The reason why wood and plastic are not landfilled in the baseline scenario is due to the landfill ban on organic waste, which has been in force since 2016. However, landfill disposal of mineral wool and plasterboard is allowed, due to the low content of organic carbon.

In Scenario 1, mineral wool and wood are treated in the same way as in the baseline scenario, but more advanced waste treatment methods are applied in the treatment of plastics and plasterboard. Thirty percent of plastics are recovered as material in a conventional manner (so-called mono-material recovery), and the remaining 70% of plastics are incinerated. The division between material and energy recovery is based on the assumption that plastics are first sorted at the site with a limited efficiency based on a visual inspection. After source separation, plastics are treated mechanically with a limited separation rate. As an outcome of both separation phases, the share of plastics recovered as material, 30%, is assumed to be rather low, yet realistic. The recycled plastics substitute for virgin high-density polyethylene (HDPE) plastic in a mass-based substitution rate of 0.73:1. Plasterboard consists of gypsum (96%) and paper (4%). In the material recover process of plasterboard, gypsum can be recovered, and the recovered gypsum can substitute for conventional gypsum, flue gas desulphurisation gypsum, in a market- and mass-base rate of 0.19:1(Fisher, 2008). The paper separated from plasterboard contains impurities and is therefore incinerated. Energy recovered from the incineration of plastics (70%) and wood substitutes for average district heat and electricity production in Finland, as in the baseline scenario. (Publication IV.)

In Scenarios 2-4, CDW fractions are used as raw materials for WPCs. The scenarios differ in terms of the material which is assumed to be substituted with the produced WPC. The produced WPC substitutes for different types of plastics in the three sub-scenarios of

Scenario 2: for polypropylene (PP) in Scenario 2.1, for polyvinyl chloride (PVC) in Scenario 2.2. and for HDPE in Scenario 2.3. In Scenario 3, the produced WPC substitutes for wood materials. Scenario 3 has four sub-scenarios in which different wood materials are substituted with the produced WPCs: plywood in Scenario 3.1, solid timber in Scenario 3.2, laminated wood in Scenario 3.3 and particle board in Scenario 3.4. Being that wood and plastic are the main raw materials in WPCs, they are also the most plausible materials for substitution with the produced WPCs, due to having rather similar mechanical and physical properties. The study also assessed the possibility that the produced WPCs would substitute for aluminium, which can be possible in specific applications. Therefore, the produced WPCs substitute for aluminium profiles in Scenario 4. An aluminium profile is assumed to be made of 75% recycled and 25% virgin aluminium, which represents standard aluminium production in Finland (Kuusakoski, 2018). The scenarios are presented in Table 3.4.

Table 3.4. Mass flows of the CDW management case study (MW stands for mineral wool and PB for plasterboard).

<table>
<tr><th colspan="2" rowspan="2">Scenario</th><th colspan="2">Landfill [kg]</th><th colspan="2">Incineration [kg]</th><th colspan="2">Material recovery [kg]</th><th>WPC production [kg]</th><th rowspan="2">Substituted material [kg]</th></tr>
<tr><th>MW</th><th>PB</th><th>Plastic</th><th>Wood</th><th>Plastic</th><th>PB</th><th>All fractions</th></tr>
<tr><td rowspan="2">0</td><td>R1</td><td>-</td><td>-</td><td>400</td><td>540</td><td>-</td><td>-</td><td rowspan="2">-</td><td rowspan="2">-</td></tr>
<tr><td>R2</td><td>150</td><td>150</td><td>400</td><td>240</td><td>-</td><td>-</td></tr>
<tr><td rowspan="2">1</td><td>R1</td><td>-</td><td>-</td><td>280</td><td>540</td><td>120</td><td>-</td><td rowspan="2">-</td><td rowspan="2">-</td></tr>
<tr><td>R2</td><td>150</td><td>-</td><td>280</td><td>240</td><td>120</td><td>150</td></tr>
<tr><td rowspan="2">2.1</td><td>R1</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td rowspan="18">940 CDW + 60 additives = 1000 WPC</td><td rowspan="2">1 000 PP</td></tr>
<tr><td>R2</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td rowspan="2">2.2</td><td>R1</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td rowspan="2">1 000 PVC</td></tr>
<tr><td>R2</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td rowspan="2">2.3</td><td>R1</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td rowspan="2">1 000 HDPE</td></tr>
<tr><td>R2</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td rowspan="2">3.1</td><td>R1</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td rowspan="2">1 000 plywood</td></tr>
<tr><td>R2</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td rowspan="2">3.2</td><td>R1</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td rowspan="2">1 000 solid timber</td></tr>
<tr><td>R2</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td rowspan="2">3.3</td><td>R1</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td rowspan="2">1000 laminated wood</td></tr>
<tr><td>R2</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td rowspan="2">3.4</td><td>R1</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td rowspan="2">1000 particle board</td></tr>
<tr><td>R2</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td rowspan="2">4</td><td>R1</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td rowspan="2">1000 aluminium profile</td></tr>
<tr><td>R2</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td></tr>
</table>

The zero-burden approach (Ekvall et al., 2007a) has been utilized in the CDW study, as in the other case studies included in this dissertation. The system boundaries of the study start from the moment when CDW fractions are transported to a waste treatment centre. The collection and transportation of the CDW fractions to the centre are excluded from the assessment because all the scenarios assume the same transportation difference; therefore, this phase does not impact on the differences among the scenarios. The system boundaries include the transportation of the CDW to the waste incineration plant and recycling facilities, landfilling of mineral wool and plasterboard, incineration of plastic and wood, and production of WPCs. In these phases or unit processes, direct emissions are generated. In addition to direct emissions, the system boundaries encompass the avoided productions and emissions of energy (i.e. district heat and electricity) and different materials (i.e. gypsum, plastic, wood and aluminium). In the scenarios where the CDW fractions are used as raw materials for WPCs, the system boundaries end at the WPC production phase. Therefore, the use and end-of-life phases of the produced WPC and the substituted materials are excluded from the system boundaries. Another reason for this exclusion is the functional unit of the study, the treatment of 940 kg of CDW, which lays the emphasis primarily on the waste management perspective. The system boundaries of the study are presented in Figure 3.13.

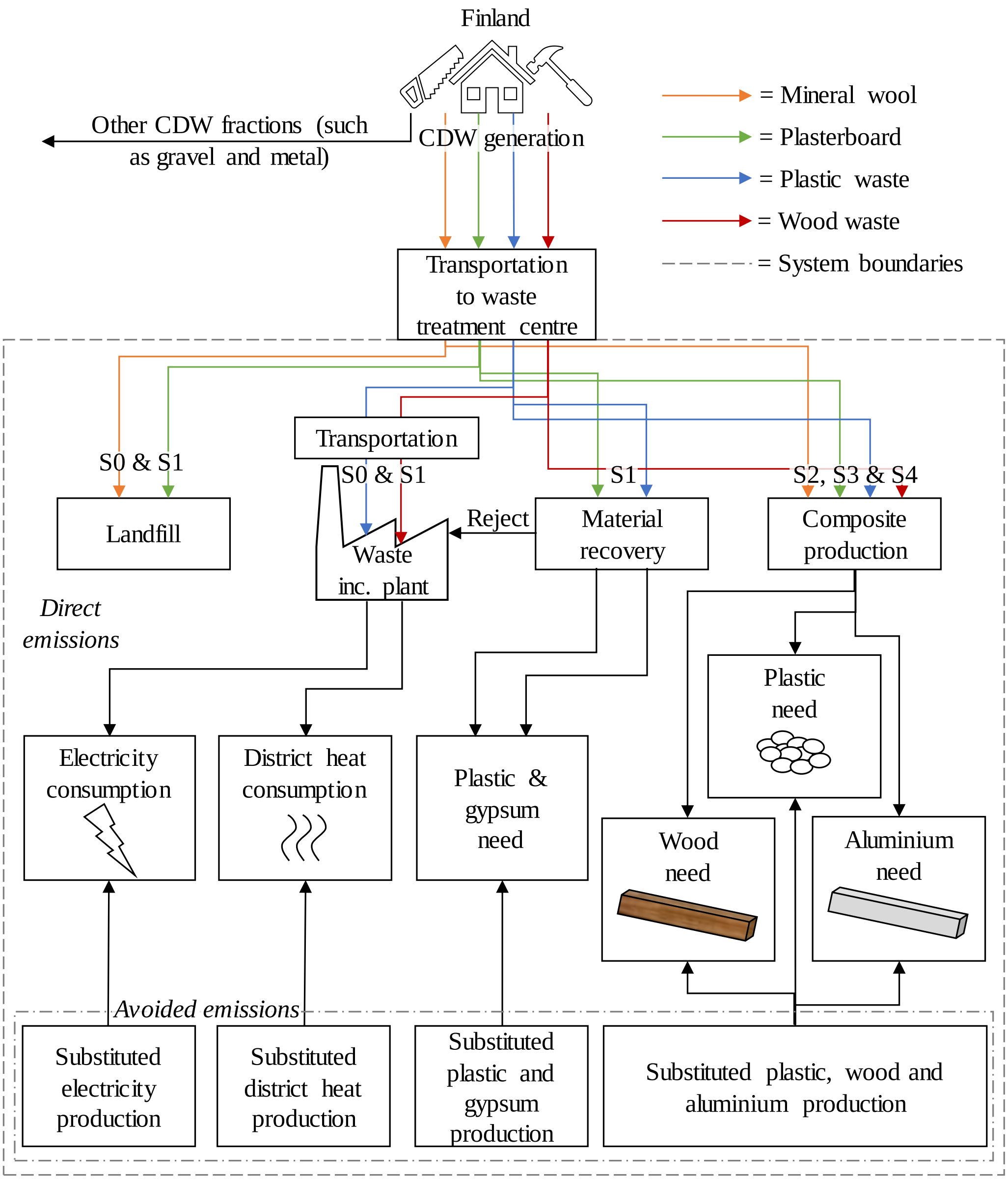

Figure 3.13. System boundaries of the construction and demolition waste management case study.

3.5 Comparison of the case studies and operational environments

The socio-economic aspects of the operational environments assessed in the case studies differ considerably, as the GNIs of the countries demonstrate (see 3.1.1, 3.2.1 and 3.3.1 above). As discussed previously, socio-economic aspects influence numerous factors concerning waste and waste management, waste composition being one of those. Waste composition is influenced by income and education levels, as examples of socio-economic factors of an operational environment. The waste composition data applied in the case studies reflect this phenomenon clearly (see Table 3.5). The proportion of packaging materials, such as paperboard and plastic, typically increases alongside income level (the higher the income level, the higher the proportion of packaging materials), as a result of an increased consumption of goods, commodities and pre-cooked food. The composition data of the case studies support this hypothesis: the highest proportion of plastic and paperboard attributable to a high-income level was indeed identified in the Finnish case study. In contrast, an opposite influence is typically detected between income level and the proportion of organic waste: the higher the income level, the lower the proportion of organic waste. The waste composition data of the case studies also supports this hypothesis: the proportion of organic waste is clearly lowest in the Finnish case study, whereas in the Chinese and Brazilian case studies, organic waste makes up a substantial proportion of mixed waste.

Table 3.5. Composition of mixed waste in the case studies.

Waste fraction	Proportion in mixed waste [%]		
	China	Finland	Brazil
Organic waste	56	24	49
Paper and cardboard	11	15	11
Plastic	19	21	15
Wood and textiles	4	12	5
Metal	1	4	1
Glass	1	3	2
Miscellaneous waste	8	21	17
∑	100	100	100

As is common in waste LCA studies, the surrounding systems influence the total environmental impacts of waste management, and they may even override the environmental impacts of the waste management system itself (Ekvall et al., 2007b). Regardless of the differences among the case studies and the operational environments, a common factor of all the case studies is the energy recovery from waste. In the Hangzhou and São Paulo case studies, electricity is recovered and produced from mixed waste. Due to their geographical locations experiencing warm climate conditions, there is no need for district heat, and only electricity is recovered and produced. In the South Karelia case study, energy is recovered with high efficiency from mixed waste, but unlike in Hangzhou or São Paulo, in addition to electricity, district heat is produced from the energy recovered from mixed waste in the assessed scenarios. Furthermore, process steam is recovered and utilized in an industrial process nearby in one of the assessed scenarios. In the case study focusing on CDW management in Finland, both electricity and district heat are recovered from waste. The multifunctionality of simultaneously treating waste and recovering energy, both district heat and electricity, is managed and modelled by applying ‘substitution’, also known as ‘crediting’ and the ‘avoided-burden’, approach. When one applies this approach, the energy recovered and produced from waste is assumed to substitute for other energy production, typically energy produced from conventional fuels. The environmental impacts of the substituted energy production are encompassed within the system boundaries as negative, avoided environmental impacts. In the case studies, the waste-derived electricity was assumed to substitute for average grid mix electricity in each case country (see Table 3.6). The substituted district heat and process steam production varied case-by-case, depending on the assumptions made and on local conditions.

Table 3.6. Electricity grid mixes in the case areas in 2015 (Thinkstep, 2019c).

Energy source	Proportion in grid mix [%]		
	China	Finland	Brazil
Nuclear	2.9	34.0	2.5
Lignite	-	-	1.3
Peat	-	4.5	-
Hard coal	68.8	7.5	2.0
Coal gases	1.3	0.8	1.4
Natural gas	2.5	7.6	13.7
Heavy fuel oil	0.2	0.3	5.1
Biomass	0.9	15.5	8.3
Biogas	-	0.5	0.1
Waste	0.2	1.3	-
Hydro	19.3	24.5	61.9
Wind	3.2	3.4	3.7
Photovoltaic	0.8	0.01	0.01
∑	100.0	100.0	100.0
Renewable + waste	24.3	45.2	74.0
Non-renewable	75.7	54.8	26.0

The proportions of renewable and fossil energy sources in the average electricity grid mixes differ substantially among the countries. The highest proportion of non-renewable energy sources in the average electricity production grid mix is in China: approximately 76%. The lowest proportion is in the Brazilian grid mix: 26%. Finland lies in between Brazil and China in this regard, with a 55% proportion of non-renewable energy sources in the average electricity grid mix. It should, however, be noted that the proportion of nuclear energy is rather high, 34%, in the Finnish grid mix. In terms of global warming, acidification and eutrophication potentials, for instance, the environmental impacts of nuclear energy are low. Therefore, it is more informative to concentrate on the environmental impacts of average electricity production in these countries (see Figure

3.14), in which the emission factors contributing to the GWP, AP and EP impact categories of average grid mixes are presented as mass of equivalent emission per produced kWh). The emission factors of average grid mixes indicate that the Chinese grid mix has the highest environmental impacts contributing to GWP, AP and EP. The environmental impacts of the average electricity grid mix in Finland are, instead, the lowest in this regard. Though hydropower comprises a large share of the average electricity grid mix in Brazil, the environmental impacts contributing to these particular impact categories are higher than those of the average Finnish grid mix, due to the rather high proportions of natural gas (14%) and heavy fuel oil (HFO) (5%) in the Brazilian grid mix, as compared to Finland's.

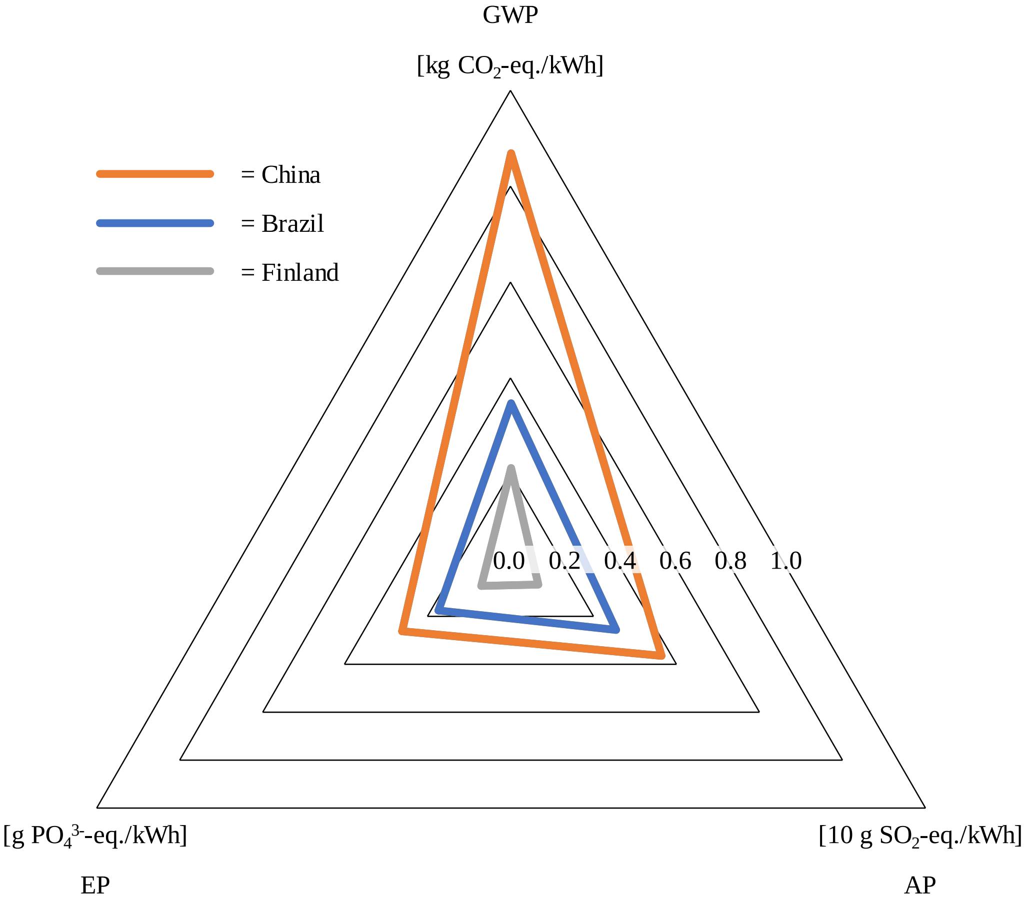

Figure 3.14. Global warming potential, acidification potential and eutrophication potential of average electricity production in the case countries (Thinkstep, 2019a).

Various materials are recovered from waste in the case studies. In all case studies, metals, namely steel and aluminium, are recovered through a mechanical separation process, or from the bottom ash in incineration plants. The recovered metals are assumed to substitute for the conventional production of metals. The conventional production does not necessarily equate to production using virgin metals, since metals as valuable materials already circulate to some extent. The avoided environmental impacts of substituted production of metals are accounted for in the case studies. In addition to metals, other material production is also assumed to be substituted with a material recovery from waste. For instance, treated and stabilized bottom ash is assumed to substitute for the production of gravel in the South Karelia case study. The assumptions regarding substituted energy and material production in the case studies are summed up in Table 3.7.

Table 3.7. Substituted energy and material production in the case studies.

Substitution context	Publication I & II	Publication II	Publication III	Publication IV
	Mixed waste management (Hangzhou, China)	Mixed waste management (South Karelia, Finland)	Mixed waste management (São Paulo, Brazil)	CDW management (Finland)
Substituted energy production type, achieved through energy recovery from waste				
Electricity	x	x	x	x
District heat		x		x
Steam		x		
Substituted material production type, achieved through material recovery from waste				
Metal	x	x	x	x
Gravel		x		
Nutrient			x	
Ethanol	x			
Coal			x	
Gypsum, plastic and wood				x

4 Results and discussion

Contribution analysis is a method widely used to present the results of LCA studies (Heijungs and Kleijn, 2001). In contribution analysis, the overall result of an LCA study is broken down into separate contributions. Direct and avoided emissions are typically presented separately to avoid misunderstanding when interpreting results. As discussed previously (see Section 2.2.1), sensitivity analysis is a method for analysing the uncertainty of LCA studies (EN ISO 14044, 2006). A common approach to carrying out a sensitivity analysis is to vary the value of one input parameter at a time and to then determine the influence of the variation on the total result. This approach is also known as the ‘local sensitivity analysis approach’ in the literature (Groen et al., 2017). Sensitivity analyses were carried out using this approach in the case studies. The results of the contribution and sensitivity analyses are presented in the following sections.

4.1 Mixed waste management in Hangzhou, China

4.1.1 Contribution analysis

The contributions of the scenarios to the impact categories of GWP, AP and EP in the Hangzhou case study are presented in Figure 4.1 as direct and avoided emissions. At first glance, it can be noted that the GWPs of the scenarios are positive, whereas the APs and EPs of the scenarios are negative, except for the EPs of Scenario 1.2. In other words, the avoided emissions from substituted electricity production contributing to AP and EP outweigh the direct emissions generated in waste treatment activities and processes, and vice versa in terms of emissions contributing to GWP. Taking a closer look, Scenarios 2.3 and 2.4 have the lowest environmental impacts across all three impact categories. These results also lead to the following three findings. Firstly, the results indicate that AD and ethanol production are the most environmentally favourable treatment methods for organic waste and reject of the assessed treatment methods. Secondly, the results indicate that the new hypothetical incineration plants are more environmentally favourable than the current plants, as was expected initially. Finally, the results indicate that mechanical treatment prior to incineration is more environmentally favourable in regard to the GWP than to the AP or EP impact categories, compared to the baseline situation in which no pre-treatment occurs.

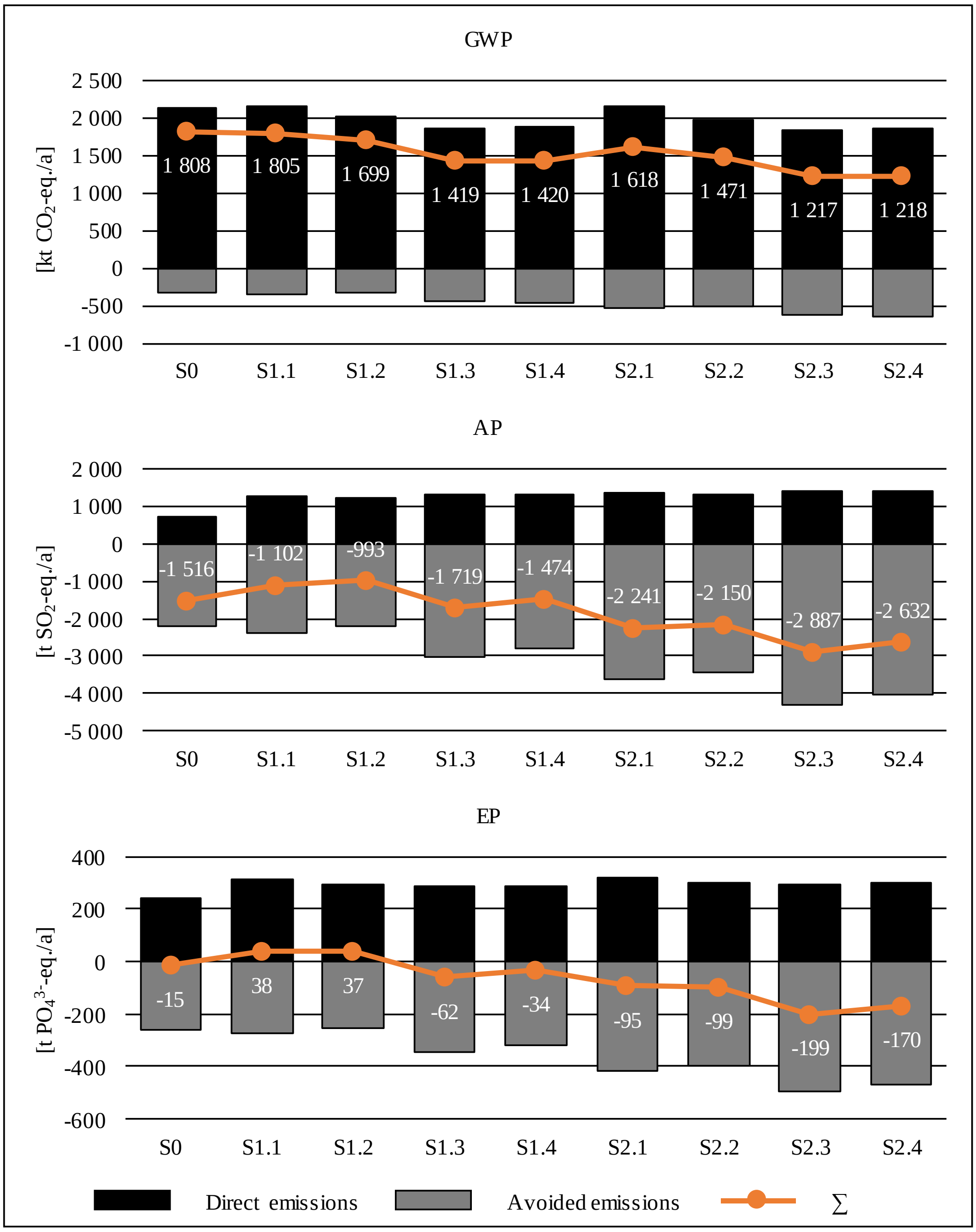

Figure 4.1. Contributions of the scenarios in the Hangzhou case study.

In interpreting the results, it was found that landfill disposal was the main contributor to the GWP in all scenarios, accounting for 60-70% of the direct emissions contributing to GWP. Incineration caused 30-40% of the direct emissions contributing to GWP. As Figure 4.1 demonstrates above, the direct emissions contributing to AP and EP are lower than the avoided emissions, resulting mainly from the substituted electricity production, assumed to be attributable to the fossil energy-dominated average electricity production in China. Therefore, electricity substitution particularly plays a vital role in these impact categories. Material substitution, i.e. metal recycling, did not yield a notable amount of avoided emissions; this is due to the relatively low yield and degraded quality of recovered metals.

4.1.2 Sensitivity analysis

A sensitivity analysis was conducted by applying the local sensitivity analysis approach to the study, i.e. by varying one input parameter at a time and then determining the influence on the total result. The influence on the total result was determined using the following equation (1):

$$SR = \frac{\frac{\Delta \text{result}}{\text{initial result}}}{\frac{\Delta \text{parameter}}{\text{initial parameter}}} \tag{1}$$

, where SR = Sensitivity ratio;
Δresult = Difference between the initial result and the result of the sensitivity analysis; and
Δparameter = Difference between the initial value of a parameter and the value of a parameter applied in sensitivity analysis.

As presented, the SR is the proportion of the relative change of the total result to the relative change of an input parameter (Clavreul et al., 2012). A variation in an input parameter thus results in an SR-fold variation in the total result. For example, when the SR of a parameter is 5, a 20% increase in the parameter's value corresponds to a 100% (5*20%) increase in the total result. The positive or negative sign of an SR indicates the influence of a parameter on the total result: parallel (positive sign) or reverse (negative sign). Calculating SRs for parameters enables one to identify the most important input parameters in regard to their sensitivity. It should be borne in mind that sensitivity does not correspond to uncertainty, although they are closely related. The purpose of a sensitivity analysis is to identify sensitivity of inputs, i.e. the effect of changes in the input on the total result, while an uncertainty analysis focuses on identifying the total uncertainty of results, considering both parameter sensitivity and variability (Clavreul et al., 2012).

In this study, the focus was on the sensitivity aspect because reliable information about the inherent uncertainty of parameters was not available. The SRs of 61 parameters were determined in the study in order to identify the most sensitive ones. The most sensitive parameters and their appertaining SRs concerning landfilling and incineration are presented in Figure 4.2. The figure indicates that the most sensitive parameters for landfilling concern LFG generation and collection, and the proportion of CH_4 in LFG. In addition to the high sensitivity, uncertainty is associated with LFG generation because it is a difficult parameter to measure, and therefore its modelling typically encompasses high uncertainty. Leachate generation and the concentration of pollutants in leachate are also sensitive parameters in the EP of landfill disposal. When analysing parameters concerning incineration, electricity recovery efficiency and the LHV of mixed waste are found to be among the most sensitive parameters in all impact categories. Concerning the GWP, parameters directly related to the fossil CO_2 emissions of incineration are also highly sensitive. It should also be noted that electricity consumption in an incineration plant is shown to be a sensitive parameter with respect to the AP and EP impact categories.

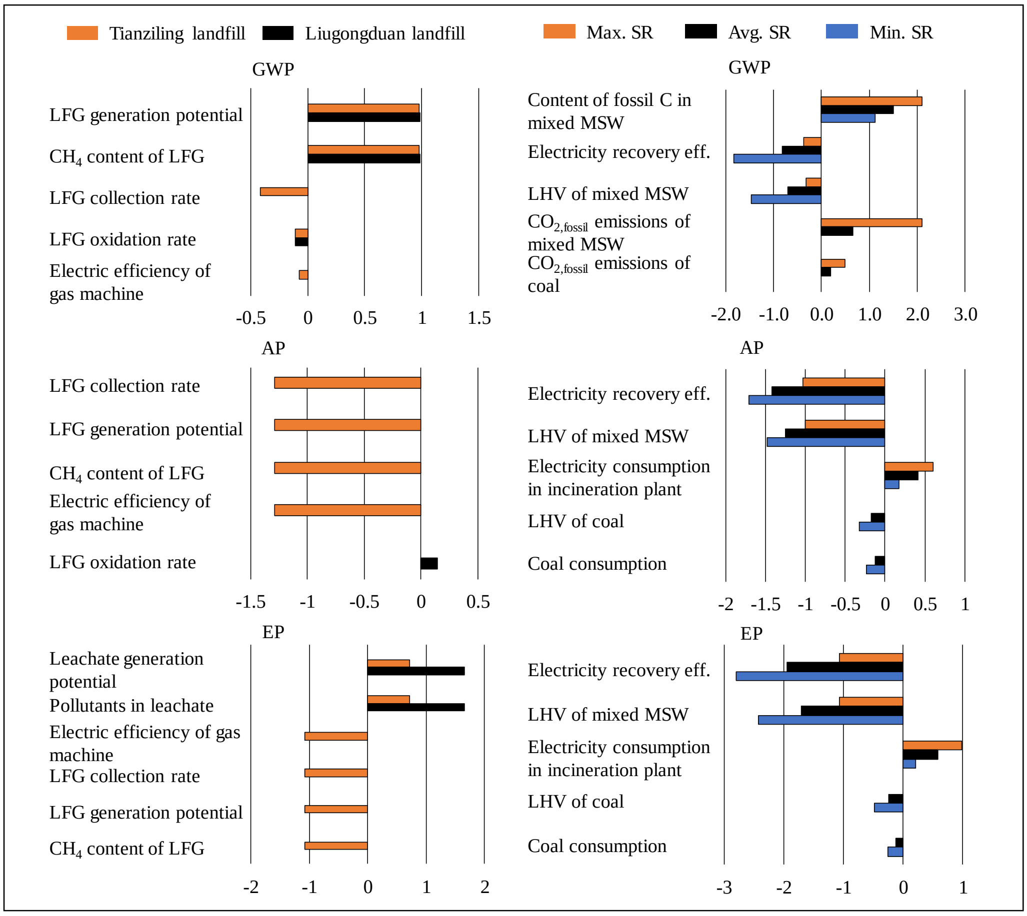

Figure 4.2. The most sensitive parameters concerning landfilling (on the left) and incineration (on the right) and their resulting sensitivity ratios in the Hangzhou case study.

4.2 Mixed waste management in the South Karelia region, Finland

4.2.1 Contribution analysis

The contributions of the scenarios to the GWP, AP and EP impact categories in the South Karelia case study are presented in Figure 4.3 as direct and avoided emissions. The results reveal unambiguously that incineration (Scenarios 1.1, 1.2 and 1.3) is a more environmentally favourable treatment option for mixed waste than is landfill disposal. The results also reveal the high influence of substituted energy production on the total results. The direct emissions generated in Scenarios 1.1-1.3 are of the same order of magnitude, whereas the avoided emissions differ notably among the scenarios. In Scenarios 1.1 and 1.2, the recovered heat is assumed to substitute for heat produced by natural gas; whereas in Scenario 1.3, the recovered heat is assumed to substitute for heat mainly produced by biomass. Regarding the GWP impact category, biomass is considered to be a neutral energy source, due to its biogenic origin. As a result, heat substitution yields a notably lower amount of avoided emissions in Scenario 1.3 compared to Scenarios 1.1 and 1.2. In contrast, compared to biomass, combustion of natural gas generates fewer emissions contributing to AP and EP; therefore, a lower amount of avoided emissions is achieved from heat substitution in Scenarios 1.1 and 1.2. Scenario 1.3 thus has the lowest environmental impact in the AP and EP impact categories.

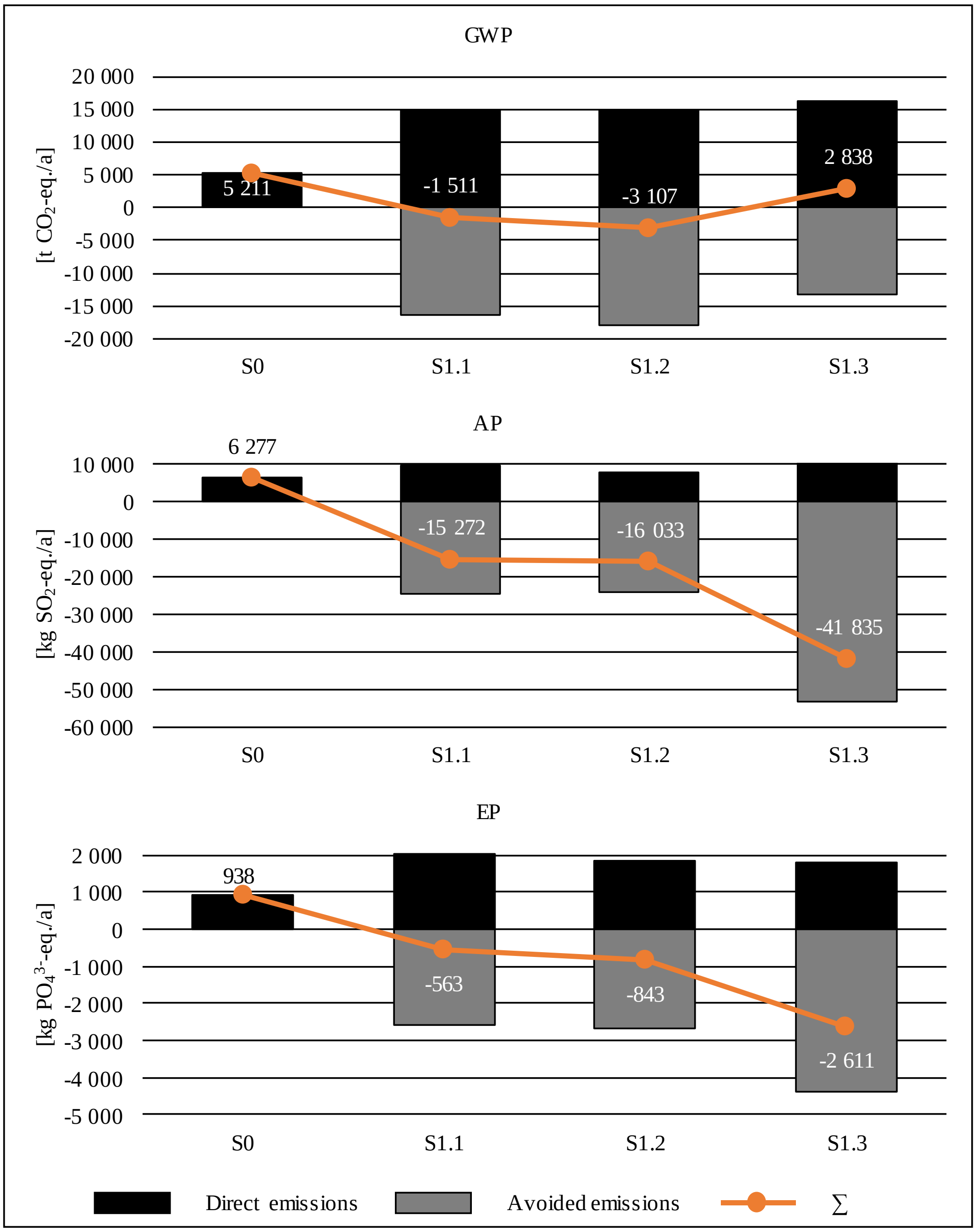

Figure 4.3. Contributions of the scenarios in the South Karelia case study.

4.2.2 Sensitivity analysis

As was the case in the Hangzhou case study, a sensitivity analysis was conducted using the local sensitivity analysis approach to determine the sensitivity ratios for input parameters (equation (1), see Section 4.1.2) in the South Karelia case study. SRs were determined for 55 input parameters. The most sensitive parameters regarding landfilling and incineration are presented in Figure 4.4. As can be seen, the most sensitive parameters concerning landfilling are related to the direct air emissions of landfilling, such as the LFG collection rate, CH_4 emissions from uncollected LFG and treatment of collected LFG. The landfilling of mixed waste contributes more to the GWP impact category compared to the other two assessed ones, AP and EP, due to the CH_4 emissions generated in the degradation process. Therefore, the SRs of parameters concerning landfilling are notably less sensitive in the AP and EP impact categories. The LHV of mixed waste and energy recovery efficiencies are found to be the most sensitive parameters concerning incineration in all three impact categories. Concerning GWP, the fossil CO_2 emissions of incineration also exert a high influence on the total results.

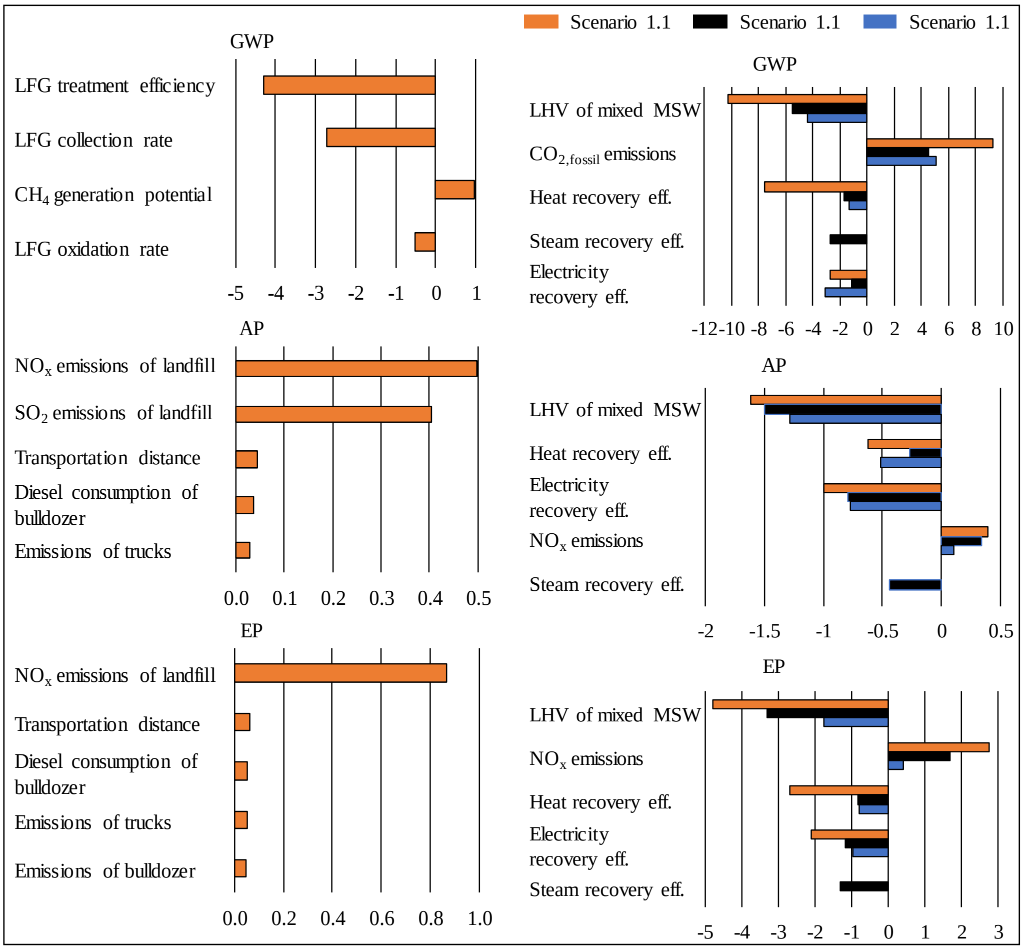

Figure 4.4. The most sensitive parameters concerning landfilling (on the left) and incineration (on the right) and their resulting sensitivity ratios in the South Karelia case study.

4.3 Mixed waste management in the city of São Paulo, Brazil

4.3.1 Contribution analysis

The contributions of the scenarios to the GWP, AP and EP impact categories in the São Paulo case study are presented in Figure 4.5 as direct and avoided emissions. Scenarios 4.2 and 4.1, respectively, have the lowest environmental impacts in all three impact categories, indicating that compared to incineration, the cement kiln is a more environmentally favourable utilization option for the produced RDF. It should be particularly noted that the incineration of RDF (Scenario 3) does not diminish the GWP impact, since the direct emission of electricity production from incineration outweighs the avoided emissions from electricity substitution. The results also indicate that AD

(Scenarios 2.2, 3.2 and 4.2) is a more favourable treatment option for organic waste than composting (Scenarios 2.1, 3.1 and 4.1). The results are unambiguous in this regard across all assessed impact categories. Home composting (Scenario 1) slightly decreases the GWP impact, but has the reverse effect on the AP and EP impact categories, indicating a trade-off situation in this regard.

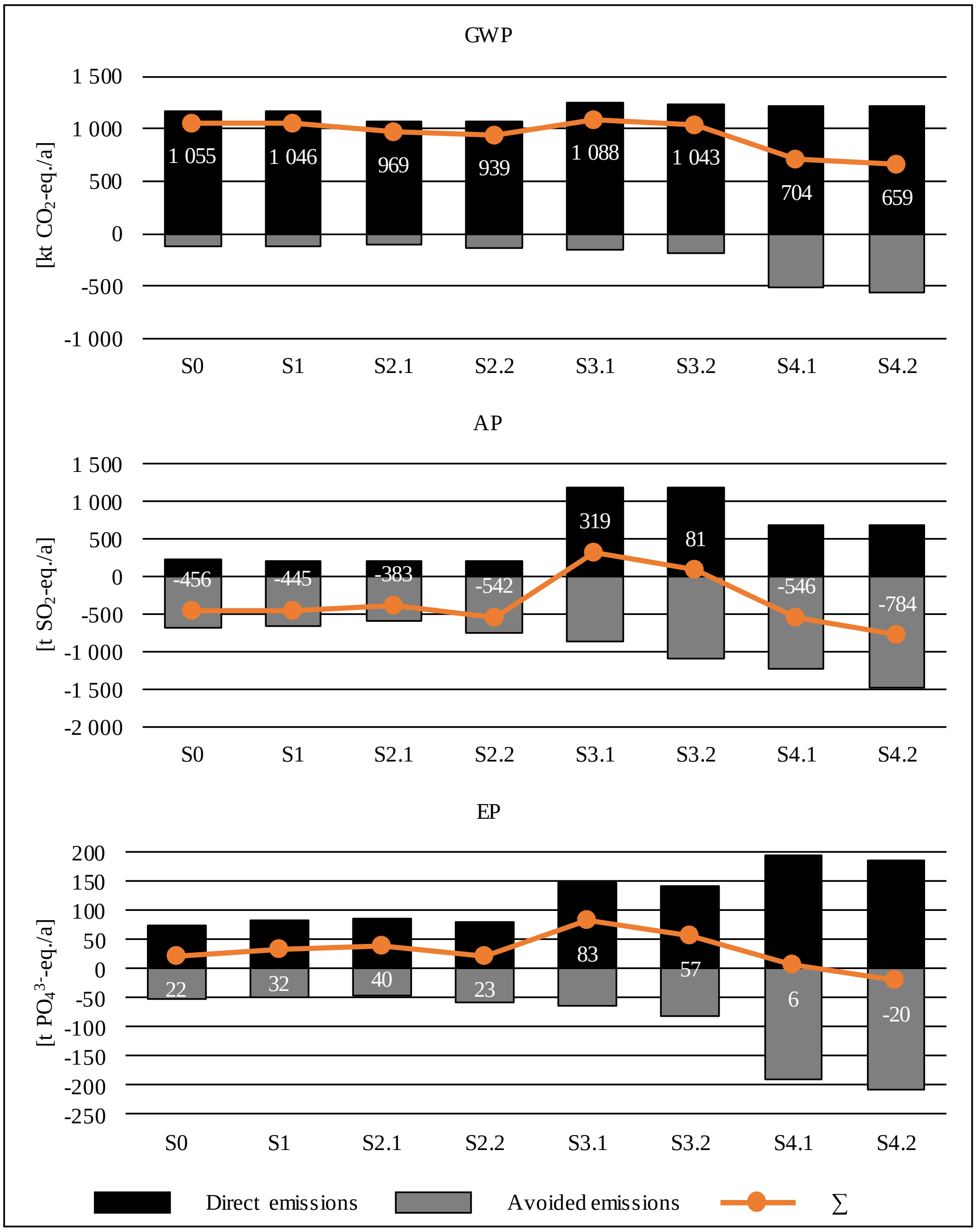

Figure 4.5. Contributions of the scenarios in the São Paulo case study.

4.3.2 Sensitivity analysis

The study assumed that the electricity recovered and produced from RDF substitutes for average electricity production in Brazil. As the results revealed that the direct emissions of incineration are greater than the avoided emissions achieved from electricity substitution, the assumption regarding the type of substituted electricity production was further evaluated in the sensitivity analysis. Instead of the average electricity grid mix, the produced electricity could also substitute for other energy production, such as energy produced from natural gas and HFO. Natural gas and HFO are deemed to be potential energy sources for marginal energy production in Brazil, so they were selected as alternative energy sources in the sensitivity analysis. The influence of different electricity substitution alternatives on the total result was analysed by determining relative weighted results (RWRs). The purpose of the sensitivity analysis was to discover how the type of substituted energy production affects the environmental performance of incineration compared with landfilling. Determining the RWRs enables one to identify the ranking between the scenarios with different electricity substitution assumptions. The RWRs were determined using equation (2).

$$\mathrm{RWR} = \frac{\mathrm{Result(Scenario_0)\text{-}Result(Scenario_i)}}{\mathrm{Max}\Delta} \qquad (2)$$

, where Result($\mathrm{Scenario_0}$) = The net result of the baseline scenario (Scenario 0) in a given impact category;

Result($\mathrm{Scenario_i}$) = The net result of a given scenario in a given impact category; and

MaxΔ = The maximum difference between the result of the baseline scenario (Scenario 0) and the results of the scenario with the lowest environmental impacts in a given impact category.

In determining the RWRs, the results are weighted relative to the result of the baseline scenario, Scenario 0. The RWR of Scenario 0 is thus always 0. If the RWR of a given scenario (i.e. any scenario other than Scenario 0) is > 0, it implies that the scenario is environmentally more favourable than Scenario 0. Conversely, if the RWR of a given scenario is < 0, it implies that the scenario is environmentally less favourable than Scenario 0. Due to the calculation principle of RWR presented in equation (2), the maximum value of RWRs is 1. Therefore, if the RWR of a given scenario is 1, the scenario is environmentally the most favourable of all assessed scenarios. The RWRs were determined separately for each assessed impact category. Figure 4.6 presents the RWRs of the scenarios.

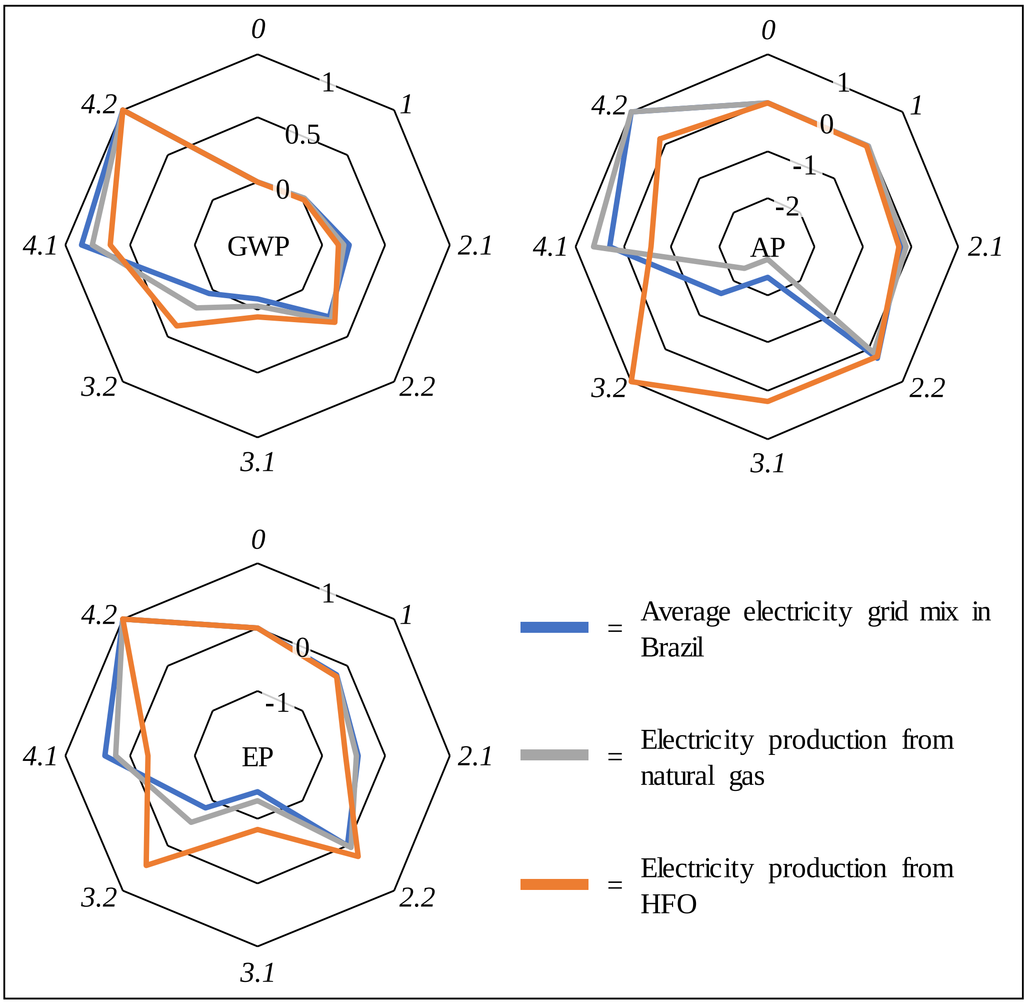

Figure 4.6. Relative weighted results of the São Paulo case study, with different alternatives for the substituted electricity production.

When analysing the RWRs in detail, one can see that the GWPs of the scenarios do not change notably when different alternatives for substituted electricity production are applied. When the substituted electricity production is derived from natural gas, the ranking of the scenarios does not vary compared to the baseline scenario. The GWPs of Scenarios 3.1 and 3.2 decrease slightly when the substituted electricity production is derived from HFO, making incineration a more environmentally reasonable utilization option for the produced RDF compared to the initial result. Even though the results indicate an improvement in the environmental performance of incineration, the utilization of RDF in a cement kiln (Scenarios 4.2 and 4.1) is still environmentally more favourable. Variations in the substituted electricity production have a greater influence on the APs of the scenarios. When the substituted electricity production is derived from natural gas, the APs do not vary significantly, and the ranking of the scenarios remains the same.

Conversely, the ranking of the scenarios changes substantially when the substituted energy source is HFO. As an outcome of the decrease in the APs of Scenarios 3.2 and 3.1, environmentally speaking, incineration turns out to be a more favourable utilization option for RDF than the cement production one. Variations in the substituted electricity production do not have a significant influence on the EPs of the scenarios: Scenario 4.2 is the most environmentally favourable, regardless of variations. When the substituted electricity production is derived from HFO, the environmental performance of incineration (Scenarios 3.1 and 3.2) improves, and the EP of Scenario 3.2 is the second lowest. When considering the influence of variations in substituted electricity production on the organic waste treatment options, composting and AD, the ranking between them remains the same: AD is the environmentally more favourable option, regardless of the variations.

In addition to the effect of varying the type of substituted electricity production, the influence of varying the value of a single parameter was also analysed in the sensitivity analysis of the case study. Based on the findings of Publication II included in this dissertation and in previous literature, the collection rate of LFG is a key parameter contributing to the environmental impacts, particularly the GWP, of landfill disposal. Therefore, different LFG collection rates were employed in the sensitivity analysis to find out whether the results would vary if collection rates were lower. Based on data received during site visits to CTL and CTVA Caieiras landfills in São Paulo, the LFG collection rates are the following: 80% for the CTL landfill and 64% for the CTVA Caieiras landfill. Compared to previously published waste LCA studies, the LFG collection rates are higher than what is typical for Brazil. For instance, in the studies of Mendes et al. (2004) and Bernstad Saraiva et al. (2017), 50% LFG collection rates were employed. Consequently, 50% LFG collection rates were applied to both landfills in the sensitivity analysis. The GWPs of scenarios with 50% collection rates together with the initial results having higher collection rates are presented in Figure 4.7.

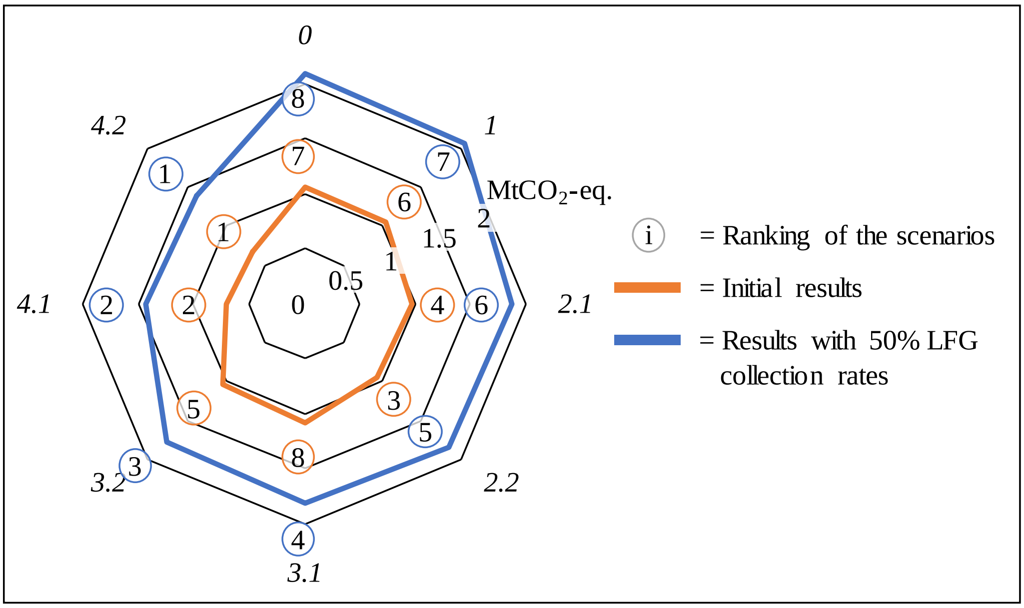

Figure 4.7. Contributions of the scenarios to the GWP with 50% LFG collection rates in the São Paulo case study.

Only the GWPs of the scenarios are presented in the figure, because lower collection rates did not significantly influence the results and the ranking among the scenarios remained the same in the AP and EP impact categories. As can be seen, when lower LFG collection rates are employed in the modelling, the environmental performance of incineration improve notably compared to the initial results. In that case, the GWPs of Scenarios 3.2 and 3.1 are third and fourth lowest, respectively. Initially, the GWP of Scenario 3.1 was the highest, and that of Scenario 3.2 was the fourth highest.

Even though the LFG collection rate proved to have a notable influence on the GWPs of the scenarios, the main findings of the study remain the same: AD is environmentally speaking a more favourable treatment option for organic waste compared to composting, and RDF utilization in cement kilns is environmentally more favourable compared to RDF incineration. As a further remark, it should be noted that when lower LFG collection rates are employed, the GWP of every alternative scenario is lower than the GWP of the baseline scenario, which is in line with the initial expectations when the goal and scope of the São Paolo case study were defined.

4.4 Construction and demolition waste management in Finland

4.4.1 Contribution analysis

The contributions of the scenarios to the GWP and fossil fuel depletion (ADP_{fossil}) impact categories in the CDW waste management case study are presented in Figure 4.8 as direct and avoided emissions. The results are presented as two side-by-side bars for each scenario, representing the two recipes for WPCs assessed in the study. The difference between Recipe 1 and 2 is more notable in Scenarios 0 and 1, in which CDW fractions are treated conventionally. The environmental impacts of Recipe 2 are higher than those of Recipe 1 for Scenarios 0 and 1 in the aforementioned impact categories. For instance, the environmental impacts contributing to the GWP of Recipe 2 are 30% higher than those of Recipe 1 in Scenario 0. In the WPC production scenarios, Scenarios 2-4, no noteworthy differences between the recipes could be identified. Therefore, Recipe 2 results in higher environmental impacts in Scenarios 0 an 1 and is an environmentally preferable option to Recipe 1, because larger benefits can be achieved when CDW fractions are utilized in WPC production instead of being treated with conventional waste treatment methods. When inspecting the environmental impacts among the scenarios, Scenario 0 has the highest contribution to the GWP impact category, whereas in the ADP_{fossil} impact category, the environmental impacts of Scenarios 3.1-3.4 are the highest. In Scenarios 3.1-3.4, in which the produced WPC substitutes for wood materials, the environmental impacts contributing to the ADP_{fossil} category are positive, i.e. the direct emissions generated in the WPC production process outweigh the avoided emissions gained in the substituted production of wood materials, due to the biological origin of wood and its neutral effect in this impact category. The contributions of Scenarios 0 and 1 are negative in the ADP_{fossil} impact category due to the avoided environmental impacts from substituted energy production and the low environmental impact of waste treatment. The lowest environmental impacts are achieved in both impact categories in Scenarios 2.1-2.4 and 4 (in which the produced WPC substitutes for plastic and aluminium, respectively), owing to the high amount of avoided emissions through material substitution.

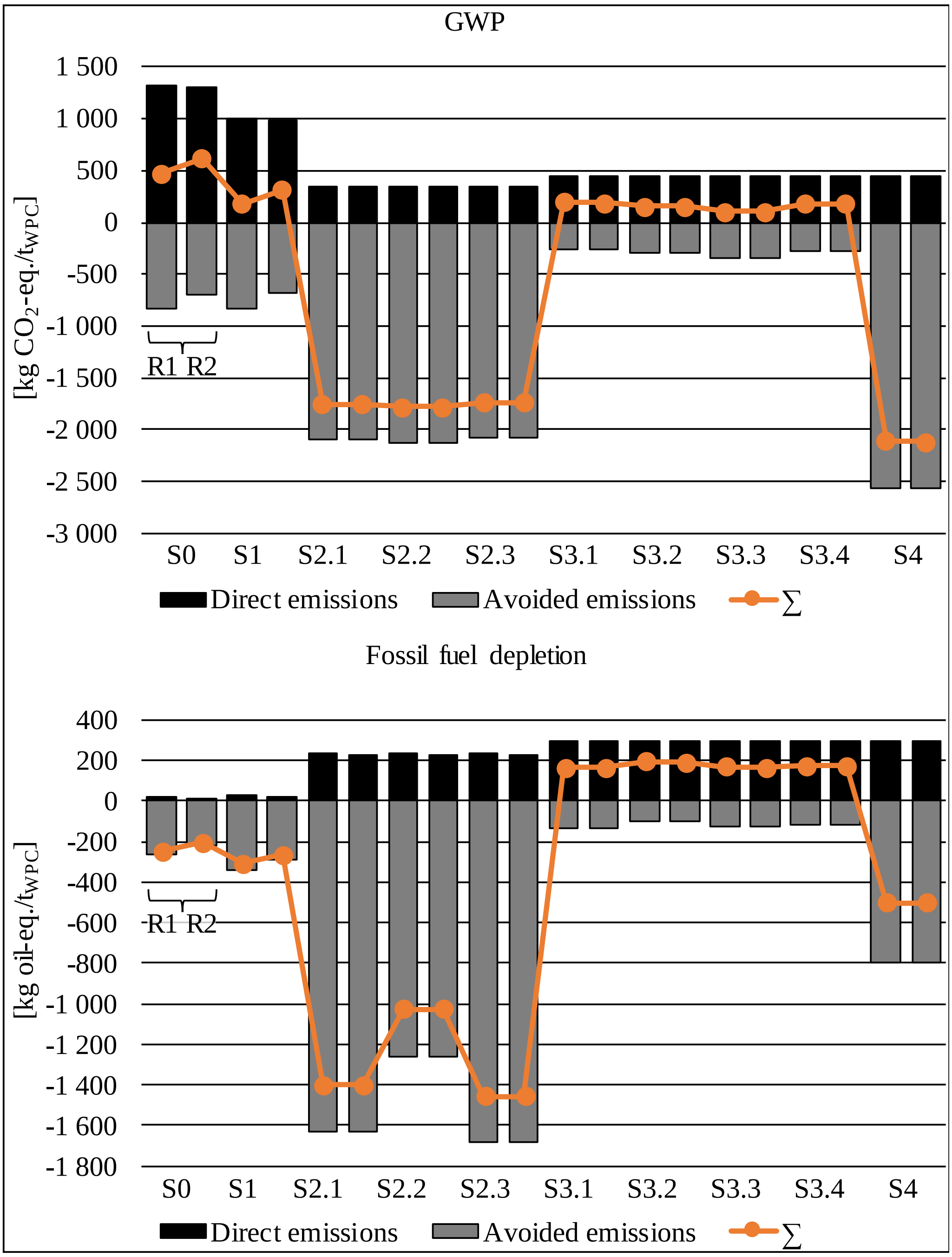

Figure 4.8. Contributions of the scenarios in the CDW management case study.

4.4.2 Sensitivity analysis

The results of the study revealed that avoided environmental impacts play an important role in regard to the total results. Avoided environmental impacts are achieved from both energy and material substitution; therefore, these aspects were further assessed in the sensitivity analysis. In Scenarios 0 and 1, it was assumed that the energy recovered and produced in the incineration of CDW fractions substitutes for average electricity and district heat production in Finland. The recovered energy could also substitute for other types of energy production, i.e. local or marginal energy production. Therefore, what was investigated is how the results would vary if the recovered energy, both electricity and district heat, were to substitute for energy produced from biomass, natural gas, hard coal and peat instead of for average energy production.

The results of this variation are presented in Figure 4.9. The results reveal that the highest emission reductions in the environmental impacts contributing to GWP are achieved by WPC production when the energy recovered from CDW fractions in Scenarios 0 and 1 substitutes for energy produced from biomass. When environmentally unfavourable energy production, such as energy produced from hard coal and peat, is substituted in Scenarios 0 and 1, the environmental benefits of WPC production (i.e. Scenarios 2-4) decrease, and vice versa. Scenarios 2-4 are environmentally speaking more preferable when the substituted energy in Scenarios 0 and 1 is derived from biomass. Therefore, the environmental performance of WPC production in particular, and material recovery in general, is higher compared to incineration, when the ‘greener’ or ‘cleaner’ energy is substituted with the energy recovered from waste. Conversely, the environmental performance of energy recovery versus material recovery improves when ‘dirty’ energy is substituted with waste-derived energy. Regarding the ADP_{fossil} impact category, more fossil resources are consumed in Scenario 3.1 compared to the other scenarios, regardless of the variations in the substituted energy production. Conversely, fewer fossil resources are consumed in Scenarios 2.3 and 4 than in Scenarios 0 and 1, despite the variations.

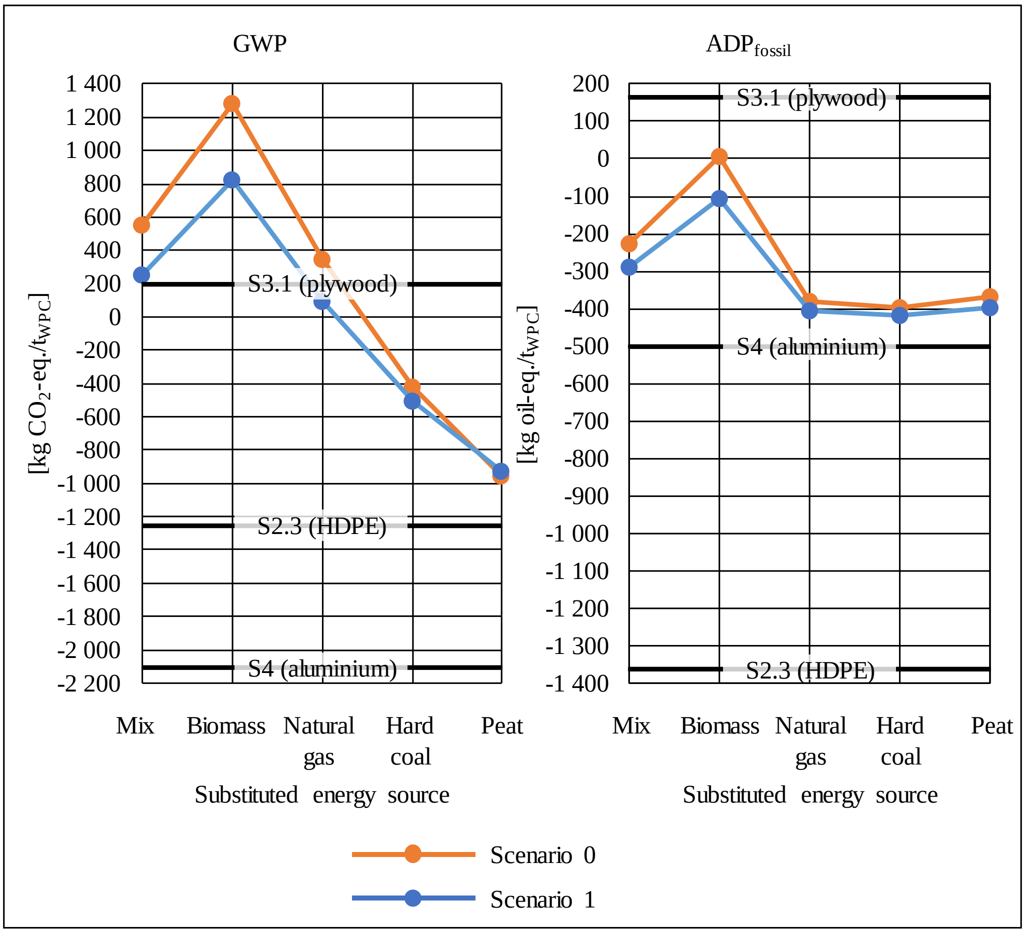

Figure 4.9. The influence of substituted energy production on the GWP and ADP$_{fossil}$ impact categories.

The study assumed that the produced WPCs substitute for plastic, wood or aluminium in a mass-based ratio of 1:1; i.e. 1 000 kg of WPC substitute for 1 000 kg of alternative material. However, since the mechanical and physical characteristics of WPCs differ from those of wood, plastic and aluminium, the substitution ratio might not reach the assumed ratio. The influence of the material substitution rate was therefore further analysed in the sensitivity analysis.

Figure 4.10 illustrates the influence of the material substitution rate on the environmental impacts of WPC production in Scenarios 2.3 (HDPE was substituted with WPC), 3.1 (plywood) and Scenario 4 (aluminium), versus the same influence findings in Scenarios 0 and 1. In the figure, the results are presented with varying material substitution rates: from a 0% substitution rate (no material substitution) to a 100% substitution rate (the assumption of the study, i.e. 1 000 kg of WPC substitute for 1 000 kg of a given material). This analysis enables one to identify the break-even points in which the environmental impacts of WPC production equal those of Scenarios 0 and 1. For Scenario 0, the baseline situation, the results of the sensitivity analysis reveal that the produced WPCs do not have to substitute for any materials in order to achieve environmental benefits. In contrast, for Scenario 1, the results show that material substitution needs to occur in order to decrease the environmental impacts of CDW management. The results indicate that in Scenarios 2.3 and 4, 6% and 8% respective material substitution rates are needed to decrease the contribution to the GWP, compared to Scenario 1. In Scenario 3.1, a considerably higher, 80%, substitution rate is required. In terms of the ADP_{fossil} impact category, considerably higher material substitution rates are required compared to the GWP impact category. In Scenario 2.3, an approximately 30% material substitution rate is required to decrease the environmental impact, while in Scenario 4, an approximately 70% substitution rate is required. If plywood (Scenario 3.1) were substituted with WPCs, the environmental impacts would always be higher compared to Scenarios 0 and 1. Therefore, the break-even point cannot be determined in Scenario 3.1.

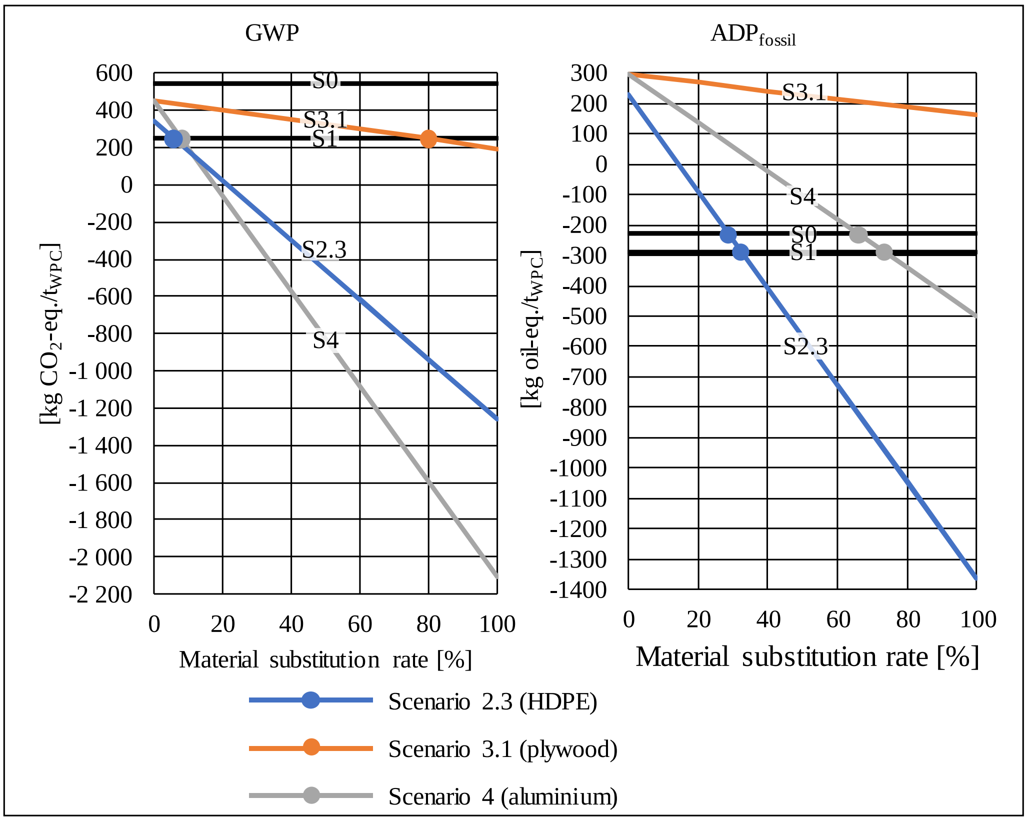

Figure 4.10. The influence of the material substitution rate (0-100%) on the GWP and ADP_{fossil} impact categories.

4.5 Exploring differences and determining factors

4.5.1 Comparison of the case studies

While the case studies included in this thesis represent distinctly different operational environments, waste management systems and waste streams, they also exhibit fundamental similarities, enabling the comparison among them. Waste management is the focal point of each case study, as the functional units of the studies demonstrated: the treatment of a given amount of waste in a given area in a reference year is the functional unit of each one.

The baseline situations of the case studies represent the current waste management practices and system in the case area in a reference year, and alternative scenarios represented potential waste treatment methods for application in the case area. In the baseline situations of all the case studies, the predominant waste treatment methods are landfilling and incineration. Alternative scenarios were selected, developed and refined case-by-case. Initially, all alternative scenarios were considered as improvement steps for the waste management system in the case area from the point of view of environmental impacts. However, as the results of the case studies demonstrate, and particularly in the São Paulo case study, all alternative scenarios do not necessarily result in lower environmental impacts compared to the baseline situation. The energy recovered from waste was assumed to substitute for other energy production using conventional fuels as energy sources. Therefore, the substitution approach (also known as crediting or the avoided-burden approach) was applied to manage the multifunctional process of simultaneously treating waste and producing energy.

As the results of the case studies demonstrate, the *type* of substituted energy production has a strong influence on the environmental impacts of waste management, because the avoided emissions are attributed as negative emissions, or credits, in the overall environmental impacts. In all case studies, the produced electricity is assumed to substitute for average electricity production in the case countries. Therefore, the modelling principles and approaches applied in the case studies are similar even though the baseline situations and alternative scenarios differ.

The case studies included in this thesis have been compared with each other using two approaches. Firstly, the Hangzhou and South Karelia case studies were directly compared with each other regarding the sensitivity of the modelling parameters for identifying how the influence of different factors varies between two very different operational environments and waste management systems. Secondly, all the results of all the case studies included in this thesis were compared with each other by further analysing the direct emissions generated in landfilling versus in incineration, as well as avoided emissions achieved from material and energy substitution.

That further analysis reveals that in landfilling, direct emissions are generated in various processes and phases, for instance, in the use of machinery, anaerobic degradation of waste, precipitation (i.e. leachate generation), and the treatment processes of LFG and leachate (e.g. consumption of electricity and chemicals as well as possible leakages). In a waste incineration plant, direct emissions are generated in the combustion process; the use of machinery; the use of auxiliary fuels (e.g. coal or natural gas); emission control (e.g. use of chemicals); as well as the treatment of ashes, rejects and slags (e.g. bottom and fly ash). Avoided emissions are those achieved through material (e.g. recovering metals from bottom ash) and energy substitution (see Section 4.5.3 for further discussion about the direct and avoided emissions).

4.5.2 Parameter sensitivity

The sensitivity analyses of the Hangzhou and South Karelia case studies demonstrate that when one analyses the sensitivity of individual input or modelling parameters, its influence on the total environmental impacts of a waste management system can be estimated in a quantitative manner with SRs. Various input parameters, such as recovery efficiencies, waste properties and emission control efficiencies, are evidently influenced by the operational environment due to the inherent relationship between waste management and the surrounding systems, as pointed out previously in the dissertation. For instance, the socio-economic and technological aspects of an operational environment influence these parameters.

The SRs determined in the Hangzhou and South Karelia case studies are compared in Table 4.1 below to discover how the sensitivity of parameters differs between these diverse operational environments. The sensitivity analyses of the case studies demonstrate that the most sensitive parameters for landfilling concern the generation, collection and treatment efficiency of LFG and leachate, though the case studies exhibit some inconsistencies in the magnitude of SRs. Regarding the most sensitive parameters of incineration, clearly standing out are the LHV of mixed waste, the $CO_{2,fossil}$ emissions of incineration, and energy production efficiencies. In addition, the NO_x and SO_2 emissions of incineration have a noteworthy influence on the results of the case studies in the AP and EP impact categories. The results of the sensitivity analyses imply that the sensitivity of a parameter might be dependent on the value of a parameter in the GWP impact category. This can be detected when the sensitivity of a given parameter increases alongside the value of a given parameter: the higher the value of a given parameter, the more sensitive the parameter is. The LHV of mixed waste and the collection rate of LFG demonstrate this phenomenon in the sensitivity analyses of the Hangzhou and South Karelia case studies.

Table 4.1. Comparison of the SRs of the input parameters applied in the South Karelia and Hangzhou case studies.

Parameter	Average SRs					
	GWP		AP		EP	
	South Karelia	Hangzhou	South Karelia	Hangzhou	South Karelia	Hangzhou
Landfilling						
LFG collection rate	-2.73	-0.21	-	-0.64	-	-0.54
LFG oxidation rate	-0.52	-0.11	-	0.07	-	0.06
LFG generation potential	0.97	0.98	-	-0.64	-	-0.54
Bulldozer diesel consumption	0.01	0.001	0.04	0.01	0.05	0.01
Bulldozer emissions	0.01	0.001	0.03	0.01	0.05	0.01
Amount of leachate	-	-	-	-	0.001	1.19
Amount of pollutants in leachate	-	-	-	-	0.01	1.19
Electricity consumption of leachate treatment	0.003	0.001	0.01	0.02	0.01	0.006
Incineration						
Electric efficiency of incineration	-2.3	-0.82	-0.85	-1.42	-1.41	-1.95
LHV of mixed waste	-6.7	-0.71	-1.47	-1.25	-3.28	-1.71
$CO_{2,fossil}$ emissions of incineration	6.29	2.16	-	-	-	-
NO_x emissions of incineration	-	-	0.28	0.2	1.64	0.63
SO_2 emissions of incineration	-	-	0.05	0.09	-	-
HCl emissions of incineration	-	-	0.01	0.01	-	-
Own electricity use in incineration	0.3	0.19	0.12	0.41	0.21	0.58
Cement consumption for residue treatment	0.13	0.07	0.03	0.05	0.07	0.1
Amount of residues (i.e. flue gas residues)	0.14	0.01	0.04	0.008	0.09	0.02
Metal recycling						
Share of metal in mixed waste	0.004	-0.01	0.001	-0.02	0.01	-0.01
Proportion of aluminium in bottom ash	-0.41	-0.01	-0.08	-0.02	-0.03	-0.01
Proportion of steel in bottom ash	-0.26	-0.002	0.004	-0.001	0.08	-0.0003
Transportation						
Transportation distance of mixed waste	0.08	0.02	0.03	0.03	0.14	0.1
Transportation distance of residues	0.002	0.002	0.001	0.01	0.0004	0.02
Transportation distance of cement	0.001	0.001	0.0003	0.002	0.001	0.006

4.5.1 Influence of an operational environment

Although the SRs of parameters yield valuable information about parameter sensitivity, i.e. the direct influence of parameters on the total results, further analysis is required to identify the influence of an operational environment on environmental impacts of waste management. Therefore, it is necessary to analyse separately the direct and avoided emissions of waste management in the case studies.

The direct emissions generated in landfilling and incineration processes and the avoided emissions resulting from the substituted material (i.e. metals) and energy substitution in the case studies are presented in Figure 4.11. In the figure, the emissions are presented per 1 kg of treated waste. The figure focuses on the emissions contributing to the GWP, because that particular impact category has been assessed in all case studies, and GWP is evidently among the most important environmental impacts of waste management and also the most commonly assessed impact category in waste LCA studies (Cleary, 2009). Additionally, since climate change is a global environmental impact, the comparison of case studies located in different corners of the globe is justified in this regard. For instance, in contrast with the GWP, acidification and eutrophication potentials have a stronger location and case specificity, and therefore a comparison of the case studies focusing on these impact categories is not as justified or informative.

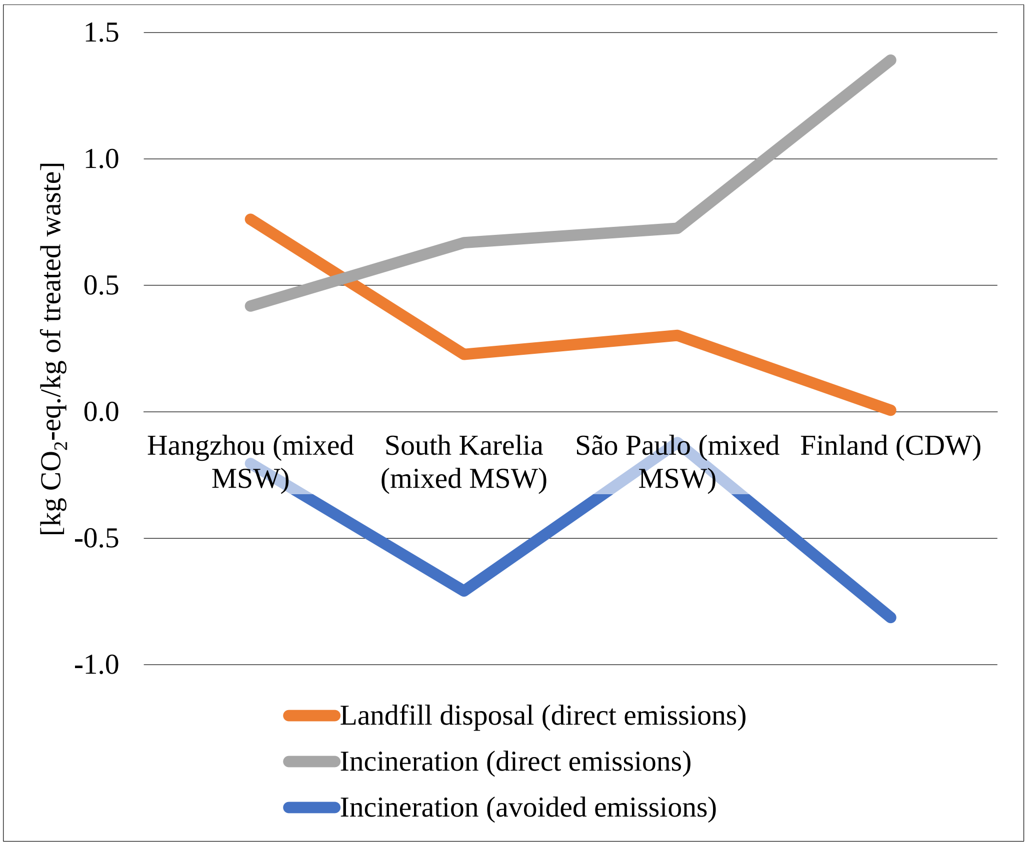

Figure 4.11. The direct and avoided emissions generated in the case studies.

The analysis of direct emissions of landfilling generated in the case studies reveals the following. In the Hangzhou case study, the direct emissions of landfilling are higher than those of incineration due to the low collection rate of LFG: 25% in the Tianziling landfill and 0% in the Liugongduan one. Due to poor or non-existent collection of LFG, environmental benefits are gained through a shift from landfilling to incineration even if the produced electricity does not substitute for conventional electricity production. In the other case studies assessed, direct emissions of incineration outweigh those of landfilling. The generation of LFG is highest in in the São Paulo case study. In the Hangzhou case study, the corresponding value is only slightly lower due to the similar characteristics of disposed waste. Since the proportion of biodegradable items in mixed waste is notably lower in the South Karelia case study, the CH_4 generation potential is also lower than that applied in the Hangzhou and São Paulo case studies. Inert waste, namely mineral wool and plasterboard, is disposed of in a landfill in the CDW management case study, and therefore the direct emissions contributing to the GWP in landfilling derive mainly from the use of machinery in landfilling operations.

The determining factors affecting the direct emissions of landfilling are highly dependent on at least two aspects of an operational environment: technological and socio-economic ones. Technological aspects of an operational environment have an influence on the maturity (i.e. the level of technological development) and efficiency of emission control and treatment. Income level, as an example of a socio-economic factor, can be justified as having an influence on the maturity of landfill operations: the higher the income level, the more the resources become available for municipal operations and responsibilities, such as waste management. Because of the influence of socio-economic aspects on waste composition, the socio-economic aspects of an operational environment also affect the CH_4 generation potential in landfilling, as CH_4 generation potential is directly influenced by the proportion of biodegradable waste fractions, such as cardboard, paper and biowaste, in the waste disposed of. The CH_4 generation potential is also influenced by landfill operations and practices, which are in turn affected by the technological aspects of an operational environment. Managed and unmanaged landfills have, therefore, different coefficients for CH_4 correction and methane oxidation factors when one determines CH_4 generation potential according to the first order decay model (IPCC, 2006). Even though the CH_4 generation potential was not the primary reason behind the high emissions of landfilling in the Hangzhou case study, it determined the ranking between the São Paulo and South Karelia case studies in this regard (see Figure 4.11), because the LFG collection rates did not vary significantly among the case studies: 64% and 80% in the São Paulo case study, and 75% in the South Karelia case study.

The direct emissions of incineration indicate that of all the case study areas, the highest amount of emissions are generated in the CDW management case study in Finland. This is due to the proportion (40%) of plastic in incinerated waste, resulting in a high amount of fossil CO_2 emissions. Since the absolute values of direct emissions of incineration are larger than those of avoided emissions originating from the substituted energy production, the net result of incineration is positive: the direct emissions of incineration outweigh the avoided ones. The same phenomenon can be detected in the Hangzhou and São Paulo case studies, albeit not as distinctly. However, the mixed waste management system in South Karelia was the only case study assessed in which the avoided emissions resulting from substituted energy production are greater than the direct emissions of an incineration process.

The direct emissions generated in waste incineration are of the same order of magnitude in the South Karelia and São Paulo case studies. Direct emissions contributing to the GWP are slightly higher in the São Paulo study than those in the South Karelia case study, because the fossil CO_2 emission factor of incinerated waste is approximately 5% higher in São Paulo than in South Karelia. The composition of mixed waste in the case studies does not account for this, but rather the type of waste incinerated does. In the South Karelia case study, mixed waste is incinerated as such, i.e. without any pre-treatment; whereas in the São Paulo case study, the incinerated waste is RDF with somewhat higher direct emissions.

Of the assessed case studies, the direct emissions of waste incineration are the lowest in the Hangzhou case study. The reason for this is that the fossil CO_2 emission factors of the waste types (i.e. mixed waste and RDF) incinerated in Hangzhou are also the lowest of the assessed case studies. For instance, the fossil CO_2 emission factor of RDF in the Hangzhou case study is approximately 19% lower compared to that applied in the São Paulo case study. Conversely, the fossil CO_2 emission factor of mixed waste in the Hangzhou case study is 29% lower compared to the emission factor of mixed waste in the South Karelia case study.

Based on the findings of the case studies, the most determining factor in the direct emissions contributing to the GWP of waste incineration is the fossil CO_2 emission factor of incinerated waste, which is directly influenced by the composition of waste. To continue this line of reasoning: the composition of waste is highly influenced by the socio-economic aspect of an operational environment. For instance, the proportion of packaging materials in mixed waste correlates with income level, due to the resulting increased consumption of commodities and pre-packed food. In contrast, the proportion of organic waste in mixed waste has an inverse correlation with income level. Even though other factors also exert an influence on the direct emissions of waste incineration contributing to the GWP, such as the use of machinery in waste incineration plants, the treatment of ashes, reject and slags, as well as the use of chemicals in processes, the influence of these did not determine the differences among the case studies in this regard. Socio-economic aspects of an operational environment can thus be considered as having a strong influence on the GWP of waste incineration.

When taking a closer look at the avoided emissions resulting from substituted energy production, one finds that the highest amount of avoided emissions is achieved in the mixed waste and CDW management case studies in Finland. Three key factors explain this and will be elaborated upon in what follows. The first key factor in the issue of avoided emissions is that the energy content (i.e. LHV) of incinerated waste differs among the case studies. It is evident why the energy content of incinerated waste is higher in the CDW management case study compared to the other case studies: wood and plastic with higher energy and lower moisture contents are incinerated in the CDW management case study, whereas mixed waste with considerably higher moisture content and lower energy content is incinerated in the other case studies. The differentiation in avoided emissions of the mixed waste management case studies is not as explicit in this regard. It is, however, clear that the proportions of two waste fractions having a strong influence on the LHV of waste, namely organic waste and plastic, are different in Finland compared to those of the mixed waste in China and Brazil. The following two assumptions can be considered as rules of thumb when dealing with the LHV of mixed waste: (1) the higher the proportion of plastic in mixed waste, the higher the LHV; and (2) the higher the proportion of organic waste in mixed waste, the lower the LHV. The proportion of plastic in mixed waste in the South Karelia case study was 21%, and corresponding proportions in the Hangzhou and São Paulo case studies were 19% and 15%, respectively. The proportion of organic waste was 24% in the South Karelia case study, and 56% and 49% in the Hangzhou and São Paulo case studies, respectively.

Since the energy content of waste is directly influenced by waste composition, the above-mentioned variations in the proportions of waste fractions in mixed waste among the case studies reflect to the LHVs of mixed waste. In the South Karelia case study, the LHV of mixed waste was approximately 15 MJ/kg, while in the Hangzhou and São Paulo case studies, corresponding values for RDF were approximately 6 and 9 MJ/kg, respectively. Because waste composition and the resulting energy content of waste are highly influenced by the socio-economic aspects, and also by the political aspects of an operational environment, the avoided emissions are also similarly affected by these aspects. The influence of political aspects on the composition and energy content of waste of an operational environment stems from the connection between these (political) aspects and the level of source separation, which in turn have an influence on waste composition (as discussed in Section 2.4.2). Thus, political aspects of an operational environment also have an influence on the avoided emissions achieved from the substituted energy production.

The second key factor affecting the environmental impacts of avoided energy production is the type of substituted energy production. As discussed previously, the type of substituted energy production varied case by case. However, a common factor among the case studies is that the electricity produced from waste is assumed to substitute for average electricity production (i.e. the average electricity grid mix) in a case country. When solely focusing on the emission factors of the average electricity grid mixes (see Figure 3.14 in Section 3.5), one assumes the highest amount of avoided emissions would be achieved when substituting waste-derived electricity for the Chinese average electricity production; the second highest amount by substituting waste-derived electricity for the average electricity grid mix in Brazil; and the lowest amount by substituting waste-derived electricity for the average electricity grid mix in Finland. However, as Figure 4.11 above demonstrates, these assumptions did not fulfil expectations: the highest amount of avoided emissions in this respect were achieved in Finland. Therefore, the type of substituted electricity production cannot be considered as a determining factor in this regard, although its importance should not be neglected, either.

The type of substituted energy production plays a more determining role when only electricity is recovered and produced from waste. In fact, the type of substituted electricity production can well be justified as being a determining factor when assessing the environmental impacts of avoided energy production between the Hangzhou and São Paulo case studies: the avoided emissions are somewhat lower in the São Paulo case study than those in the Hangzhou one, as can be expected based on the emission factors of the electricity grid mixes in the countries. Since production of electricity is highly influenced by the technical, geographical and economic aspects of an operational environment, the role of the operational environment is greatly important in this regard.

The third key factor affecting the magnitude of the avoided emissions achieved through substitution of conventional energy production is energy recovery and production efficiencies. Only electricity is recovered in the Hangzhou and São Paulo case studies, whereas both electricity and heat are recovered in the Finnish case studies. This is a result of the geographical location of the case studies: there is no need for heating in Hangzhou and São Paulo due to the considerably warmer climate conditions compared to Finland. In warmer climate conditions, the thermal energy recovered from waste could also be utilized in industrial processes as process steam in order to increase the energy recovery rate of waste incineration.

Since the energy recovery efficiency of waste incineration is higher in Finland compared to the other case countries assessed (for instance the total annual energy efficiency was 64-68% in the South Karelia case study), the avoided emissions resulting from substituted energy production are notably higher in Finland. By way of comparison, in the Hangzhou case study, the energy production efficiency of waste incineration was 16-24% in the existing waste incineration plants and 26-32% in the new, hypothetical ones. In the São Paulo case study, the energy efficiency of incineration was even lower: 18%. In addition to the geographical aspects of an operational environment, technical aspects also affect energy recovery efficiencies. Furthermore, due to the close connection between technical and economic aspects, it is also important to recognize that economic aspects of an operational environment have an influence on the energy recovery efficiency of waste incineration, albeit not necessarily a direct or evident one.

To sum up the findings of the comparison of the case studies, it was discovered that the avoided emissions of substituted energy production are inevitably a determining factor in the environmental performance of incineration versus that of landfilling. Since the direct emissions of incineration may well outweigh those of landfilling, as was the case in the South Karelia, São Paulo and CDW management case studies, the magnitude of avoided emissions achieved through substitution of other energy production determines whether incineration is more environmentally beneficial than landfilling. The Hangzhou case study differed from the other case studies: the direct emissions contributing to the GWP of incineration were lower than those of landfilling, and therefore, no energy substitution was needed to occur to gain environmental benefits in this regard.

The political aspects of an operational environment influence public acceptance of alternative waste treatment methods. In the São Paulo case study, the lack of political will to promote incineration and energy recovery from waste has inhibited the adoption of waste incineration. The regional development plan of the city thus did not have targets for promoting waste incineration. In the Hangzhou case study, public opposition against waste incineration was instead identified when determining the goal and scope of the study. Nevertheless, waste incineration has been adopted in the city due to the urgent need to decrease the volume of waste disposed of in landfills, and due to lack of space for landfill sites. As an EU member country, Finland differs from the other case countries assessed in the thesis. The ambitious targets of the EU waste policy have steered waste policy and legislation in Finland. For instance, the landfill ban is an example of an improvement step directed by the EU which has increasingly influenced the energy recovery rate of waste, while simultaneously decreasing the environmental impacts, such as GWP, of waste management. Although the influence of political aspects might not be evident from the LCI data of the case studies, political aspects did indeed affect the goal and scope phase of the studies when determining the status quo in the areas and identifying alternatives.

4.6 Reflection of the results on the research questions

This section of the thesis will explore results concerning the three central research questions. The first research question of this thesis was: What are the environmental impacts of waste management in the case areas in Finland, China and Brazil, and how might these be decreased? What follows are findings pertaining to the research question for each case.

In the Hangzhou case study, the environmental impacts of the baseline situation (i.e. the waste management practices in the reference year) are approximately 1 800 kt CO_2-eq./functional unit (0.6 kg CO_2-eq./$kg_{mixed\ waste}$), -500 t SO_2-eq./functional unit (-0.5 g SO_2-eq./$kg_{mixed\ waste}$) and -15 t PO_4^{3-}-eq./functional unit (-0.005 g PO_4^{3-}-eq./$kg_{mixed\ waste}$). The results indicate that of the assessed alternatives, the environmental impacts of waste management can be most effectively decreased by the mechanical treatment of mixed waste prior to incineration, thus decreasing the need for auxiliary fuel, namely coal. It is also important that the organic reject generated in the mechanical treatment should be anaerobically digested or utilized in ethanol production from the point of view of environmental impacts.

In the South Karelia case study, the environmental impacts of the baseline situation are approximately 5 200 t CO_2-eq./functional unit (0.2 kg CO_2-eq./$kg_{mixed\ waste}$), 6 300 kg SO_2-eq./functional unit (0.3 g SO_2-eq./$kg_{mixed\ waste}$) and 940 kg PO_4^{3-}-eq./functional unit (0.04 g PO_4^{3-}-eq./$kg_{mixed\ waste}$). The results of the study demonstrate that incineration significantly decreases the environmental impacts of mixed waste management. Though the results are unambiguous regarding the environmental performance of incineration versus landfilling, they do not clearly indicate the most environmentally favourable incineration scenario due to the differences in the type of substituted district heat production.

In the São Paulo case study, the environmental impacts of the baseline situation are approximately 1 100 kt CO_2-eq./functional unit (0.3 kg CO_2-eq./$kg_{mixed\ waste}$), -500 t SO_2-eq./functional unit (-0.1 g SO_2-eq./$kg_{mixed\ waste}$) and 20 t PO_4^{3-}-eq./functional unit (0.006 g PO_4^{3-}-eq./$kg_{mixed\ waste}$). The results indicate that the environmental impacts of waste management can be reduced with the AD of source-separated organic waste and with the MBT of mixed waste when the produced RDF is utilized in cement production, thus diminishing the need for coal.

In the CDW management case study, the environmental impacts of the baseline situation are approximately 550 kg CO_2-eq./functional unit (kg 0.6 CO_2-eq./kg_{CDW}) and -230 kg oil-eq./functional unit (-0.2 kg oil-eq./kg_{CDW}). The environmental impacts of waste management diminish, compared to the baseline situation, when the CDW fractions are utilized in WPC production. Significant environmental benefits are achieved when the produced WPCs substitute for plastic or another energy-intensive material.

The second research question of the thesis was: How do the environmental impacts of different waste treatment methods differ among the operational environments? In this case, the scenario settings varied significantly among the case studies due to the differences in baseline situations as well as in potential improvement steps. Therefore, a comparison of the results as such is not reasonable, but the direct and avoided emissions per a certain mass of treated waste have been further analysed in Section 4.2 and will be elaborated upon in what follows.

The main findings of the comparison reveal that the direct emissions of landfill disposal were higher than those of incineration in the Hangzhou case study, unlike in the other case studies, in which direct emissions of incineration outweighed those of landfilling. This resulted mainly from the significantly lower LFG collection rate in the Hangzhou case study, compared to the other case studies. The direct emissions of incineration were the highest in the CDW management case study, due to the different type of waste incinerated: CDW fractions, namely wood and plastic, versus mixed waste or RDF. Conversely, the direct emissions of incineration and landfilling were of the same order of magnitude between the South Karelia and São Paulo case studies, although the operational environments are distinctly different. Regarding the direct emissions of landfilling, the determining factor between the South Karelia and São Paulo case studies was the differences in the CH_4 generation potential of the disposed waste, since LFG was collected with a similar level of efficiency in both case studies. When considering the direct emissions of incineration, the determining factor in the Hangzhou, South Karelia and São Paulo case studies was the fossil CO_2 emission factor of incinerated waste. Amongst these case studies, the highest fossil CO_2 factor of incinerated waste was detected in the São Paulo case study, followed by the South Karelia case study. In the Hangzhou case study, the fossil CO_2 emission factor was 19% and 29% lower compared to that in the São Paulo and South Karelia case studies, respectively.

The final research question was related to the previous one: What are the most important reasons underlying the differences? Following is a discussion of the decisive factors found in the comparative analysis of the case studies.

The amount of avoided emissions achieved from substituted energy production is the most important distinctive factor among the assessed case studies. Several factors influence the amount of avoided emissions, such as the LHV of incinerated waste, type of substituted energy production, energy recovery efficiency, and rate of incineration. The comparison among the studies revealed that the amount of avoided emissions through energy substitution is clearly the highest in the Finnish context. Considering the emission factors of substituted electricity production (see Section 3.5), it can be concluded that an emission factor of substituted electricity production is not a significant factor among the case studies in this regard, but rather the higher energy recovery rate and LHV are. In order to increase the energy recovery rate of waste incineration, it is important that the recovered thermal energy is utilized in district heating or as process steam. From a broad perspective, the energy recovery of waste incineration is not very effective and requires more co-operation with the waste management sector and different industrial sectors, such as paper and cement industries. The avoided emissions of substituted energy production were slightly higher in the Hangzhou case study than in the São Paulo case study, due to the differences in the emission factors of the substituted energy production. The avoided emissions achieved from substituted material production played a significant role in the CDW management case study, whereas its role was minor in the other case studies assessed.

After analysing the role of the operational environment in the environmental impacts of waste management in the case studies, it was found that the socio-economic, technological and geographical aspects particularly had an evident influence on the differences among the case studies. The energy recovery rate of waste incineration was identified as the most important factor, affecting the results when the environmental favourability of waste incineration versus other waste treatment methods was assessed. The energy recovery rate of waste incineration is influenced by numerous factors, such as waste composition, the technological maturity of waste incineration and, most importantly, the regional need for the recovered energy. These factors are in turn influenced by the three aforementioned aspects — socio-economic, technological and geographical — of an operational environment. The thesis indeed identified the most important reasons underlying the differences among the case studies from this standpoint. Political aspects of an operational environment did not so evidently influence the environmental impacts of waste management, compared to the other aspects assessed. The political aspects, however, influenced the outlining of the scope and aims of the LCA studies through the selected scenarios and their applicability in practice.

The aspects of an operational environment should be acknowledged, particularly when exploring the differences in the environmental performance of waste treatment alternatives in different case areas. This plays a vital role, for instance, when outlining the correlation between the priority order of the waste hierarchy and environmental impacts in different areas and waste management systems.

5 Conclusions

5.1 Contribution to knowledge

This thesis explored the influence of an operational environment on the environmental impacts of waste management, through case studies. Being an ‘article’ thesis, the contributions of the publications to the existing knowledge and literature form the basis for the dissertation summary’s contribution to knowledge.

Publication I provided valuable information about the environmental impacts of mechanical treatment of waste prior to incineration, compared to the baseline situation in which waste is co-incinerated with coal without pre-treatment. Such an assessment in the Chinese context had not previously been published in the literature, and therefore the paper introduced a novel perspective for both academia and decision- and policy-making.

Publication II, in turn, compared the environmental impacts of two distinctly different waste management systems. An emphasis was placed on the sensitivity of various parameters and factors concerning the environmental impacts of waste management systems. In addition to focusing on determining factors, also factors having a minor influence on the environmental impacts were acknowledged, because they play a role when seeking possibilities to diminish the workload of waste LCA studies by applying secondary data instead of acquiring primary or site-specific data. This standpoint adds valuable information and concrete examples to the existing knowledge, with a practical emphasis.

Publication III contributed to the existing knowledge by assessing comprehensively the environmental impacts of MSW management in the city of São Paulo. Previously published LCA studies of the MSW management in the city had instead focused on specific treatment methods and their environmental impacts. With this research gap, Publication III served to provide valuable information to the public and for policy-making. From a scientific point of view, the study has placed particular emphasis on the influence of the avoided environmental impacts resulting from substituted energy production, thus adding further information to existing knowledge in the literature.

The contribution of Publication IV to the existing knowledge concerned the environmental impacts of WPCs as a material recovery option for CDW fractions. Since the main emphasis of the study was placed on the waste management standpoint (as was also the case in the other publications included in this thesis), knowledge of the environmental impacts of WPC production as part of a CDW management system was provided to the literature.

The dissertation summary discusses the influence of an operational environment on the environmental impacts of waste management through the findings of the case studies and the experience gained from them. The concept of an operational environment from the standpoint of waste management was defined in the thesis based on existing knowledge in the literature. The comparison of the case studies revealed that the avoided emissions achieved from substituted energy production play a vital role concerning the environmental impacts of a waste management system, and are a determining factor regarding the environmental performance of incineration versus landfilling. The influence of socio-economic, technological and geographical aspects of an operational environment was particularly identified as a determining factor behind the differences in the environmental impacts among the case studies. The role of political aspects was also acknowledged, yet it was not as evident as the other aspects discussed. The dissertation concluded that the political aspects particularly affect the goal and scope of waste LCA studies when determining and defining alternative treatment scenarios for the waste management system in a case area.

Due to the inherent and intricate relationship between a waste management system and its operational environment, the socio-economic, technological, political and legislative, and geographical aspects of that operational environment should be acknowledged throughout each phase of an LCA study, particularly when analysing the differences in the environmental performance of waste treatment alternatives. This plays a vital role when interpreting how the priority order of the waste hierarchy correlates with environmental impacts in different areas and waste management systems.

5.2 Recommendations for further research

This research explored the influence of an operational environment on the environmental impacts of waste management, through case studies. As mentioned previously, the case studies assessed differ distinctly from each other, yet exhibit fundamental similarities, enabling a comparison among them for identifying the influence of an operational environment from the standpoint of environmental impacts.

This research focused on the environmental impacts of waste management. To gain a more comprehensive understanding of the inherent relationship of an operational environment and a waste management system, the social and economic aspects of sustainability should also be acknowledged in further research. The diverse case studies included in this thesis made possible the identification of the determining factors of an operational environmental exerting an influence on the environmental impacts of waste management. Although the case studies were representative for this purpose, in order to conduct a more detailed analysis among them, more similar case studies could provide information about other factors having a less evident influence on the environmental impacts of waste management among different operational environments. Furthermore, future research could (1) incorporate case studies which are focused on the material recovery of mixed waste; (2) assess the environmental impacts of waste management in low-income countries, which have fewer possibilities to invest in waste treatment facilities; and (3) concentrate on other waste streams, such as commercial and industrial waste.

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Environmental impact assessment of municipal solid waste management incorporating mechanical treatment of waste and incineration in Hangzhou, China

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A B S T R A C T

Municipal solid waste (MSW) management is becoming increasingly popular around the world as a means of accommodating the increasing amounts of waste that the growing global population generates. China currently produces more MSW than any other country. As such, this area of the world is facing challenges on an unprecedented scale. MSW management in China is highly dependent on landfilling, and the development of sanitary landfills is currently a top priority for the Chinese government. Hangzhou is one of the most developed cities in China. In fact, in 2013, the amount of incinerated MSW in Hangzhou represented 56% of total MSW. MSW incineration is primarily performed via a process of co-incineration with coal because MSW has a low heating value.

This paper employs a environmental impact assessment by LCA program to determine whether refuse-derived fuel (RDF) production and incineration can have a more positive impact on the environment than the co-incineration of MSW with coal in Hangzhou, China. According to the results, RDF production and incineration could improve Hangzhou's MSW management global warming potential from −33% to 0%, the acidification potential from −90% to 34%, and the eutrophication potential from −1200%−350% in comparison to the co-incineration of MSW with coal. The treatment of organic reject material from RDF production has a significant effect on the results; as such, it should be utilized in energy production rather than landfilled.

1. Introduction

Municipal solid waste (MSW) management is an important issue for the urbanizing world, especially in developing countries, where economic development and expansion have significantly increased the generation of MSW. The vast amount of MSW generated in growing cities around the world requires sustainable management. The majority of waste is currently disposed of in landfills from where it emits landfill gas (LFG) that contains methane, a substance that makes a significant contribution to global warming. Waste and waste water together account for 3% of the global greenhouse gas (GHG) emissions, with landfill gas methane being the largest source (IPCC, 2014).

The amount of MSW in China has increased rapidly in recent years, and China is now the world's largest producer of MSW. In 2004, China generated 155 million tons (120 kg per capita) of MSW (National Bureau of Statistic of China (2005)) and by 2013 this figure had reached 172 million tons (126 kg per capita) (National Bureau of Statistic of China (2014)). These statistics do not include the waste collected by pickers, which is estimated to represent 8–10% of the total MSW generated (Chen et al., 2010).

The MSW that is generated in China is predominantly treated via landfilling and incineration. For example, in 2010, 79% of MSW was landfilled, 19% was incinerated, and 2% was composted (Dong et al., 2014a). Between 2002 and 2010, the proportion of incineration steadily increased from 3.7% to 19%. Modern landfill sites in China employ LFG collection equipment and modern leachate treatment systems to satisfy national pollution standard requirements. To be considered environmentally sound, cities in China should have safe disposal rates, which include landfilling,

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incineration, and composting, of between 85% and 90% (Chen et al., 2010).

In 2010, the share of mixed MSW disposed of in landfills in Hangzhou, which is one of the most developed areas in China, was 51%, while the rest was sent to incineration (Chi et al., 2014). The first MSW landfill (Tianziling Solid Waste Landfill) was constructed in Hangzhou in 1991, and it utilizes cement curtain technology to prevent leachate from polluting groundwater (Zhang et al., 2010).

Life cycle assessment (LCA) is used to deduce the impacts that different activities, including waste management systems, have on the environment with the intention of comparing how different configurations of the given systems vary (Cleary, 2009; Coventry et al., 2016; Ekvall et al., 2007; Fernández-Nava et al., 2014; Finnveden, 1999; Laurent et al., 2014; Turner et al., 2016; W. Zhao et al., 2009a).

Many researchers have conducted LCAs of waste management systems. Dong et al. (2013) examined the effect that source separation had on MSW management in Hangzhou and found that 23% of GHG emissions could be reduced in comparison to the base scenario. Chi et al. (2014) calculated that it was possible to achieve a 30% reduction in GHG in Hangzhou through improving source separation. More recently, Dong et al. (2014b) compared the disposal of MSW into landfills with and without LFG recovery and incineration and concluded that incineration is the most viable option for waste management in Hangzhou. Zhao et al. (2009b) used a LCA approach to estimate emissions from Hangzhou's MSW management system and found that landfills made the biggest contribution to the emissions that cause global warming, while incineration contributed the most to acidification. Zhao et al. (2011) examined MSW management in Tianjin using LCA and life cycle costing (LCC) and concluded that the operation of current and new landfills in combination with LFG recovery would represent a promising approach to waste management in Tianjin.

To date, the LCA studies that have assessed Chinese MSW management systems have focused on the incineration of mixed MSW with coal and improving source separation or improving landfill disposal practices. These studies have ignored the potential of refuse-derived fuel (RDF) production through the mechanical treatment and utilization of RDF in waste incineration. The production of RDF from mixed MSW could potentially be used to improve the fuel qualities of mixed MSW, thus removing the need to use coal as an auxiliary fuel in waste incineration plants in China. This mechanical treatment could also enable the recovery of recyclable waste from the MSW. In addition to RDF, there is also a significant organic reject fraction coming from mechanical treatment that requires treatment. To that end, the current study had two main objectives. The first of these objectives was to assess the change in the environmental impacts that transitioning from mixed MSW co-combustion with coal to the mechanical treatment of mixed MSW in combination with the incineration of RDF would have. The second objective was to determine the most environmentally sound method by which the organic reject from the mechanical treatment of mixed MSW could be processed.

2. Materials and methods

A environmental impact assessment was conducted in accordance with ISO standards 14040 and 14044 using life cycle assessment (LCA) tool GaBi 6 (ISO 14040, 2006; ISO 14044, 2006). The four major phases in the environmental impact assessment include goal and scope definition, inventory analysis, impact assessment, and the interpretation of results. Environmental impact assessment is a relative approach that requires a functional unit that defines what is being studied. All subsequent analyses are related to the selected functional unit. In the current study, the functional unit was the mass of MSW produced in the case study region in 2013. The waste management system includes processes that have multiple purposes; for example, the incineration plant produces energy, utilizes waste, and recovers materials. To avoid allocation between outputs, as is advised in Standard 14044, a system expansion was used to account for the substitution of energy and virgin material productions. An assumption of zero burden condition was used in the study, meaning that the waste enters the system boundary without any burden related to the production and use of materials. This facilitates a comparison of the treatment options that are available for a given amount of waste but is not suitable for an analysis in situations in which the quantity of waste changes; for example, as a result of waste minimization efforts (Finnveden, 1999; Hagberg et al., 2009; Laurent et al., 2014).

2.1. Goal and scope definition

The objective of the study is to investigate the change in environmental impacts that transitioning from MSW co-combustion with goal to mechanical treatment of MSW and RDF incineration would have in a Chinese city and to identify the most environmentally sound method of treating the organic rejects of RDF production. Hangzhou was selected as the case city for this research because the amount of MSW produced in Hangzhou has increased rapidly over the course of the last decade at an average annual growth rate of 10%, as presented in Fig. 1. The MSW management system in Hangzhou is based on landfilling and incineration. The

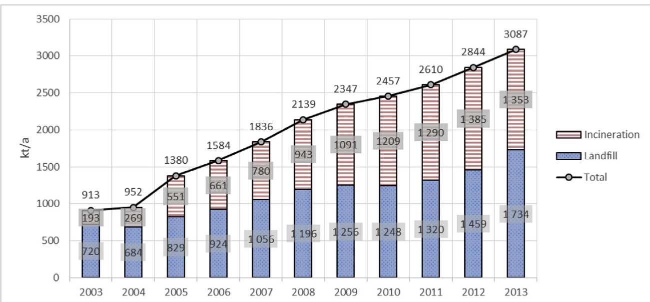

Fig. 1. The MSW mass generated and treated in Hangzhou between 2003 and 2013 (Dong et al., 2013; Hangzhou Municipal Solid Waste Disposal center, 2014).

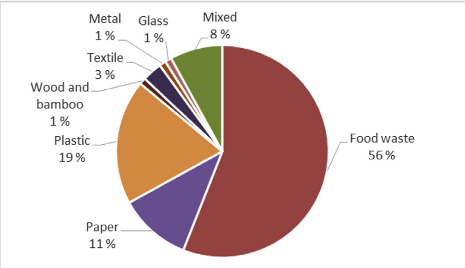

Fig. 2. Average mixed MSW composition in Hangzhou (Chi et al., 2014; Hangzhou Municipal Solid Waste Disposal center, 2014; Zhang et al., 2010).

share of waste incinerated has rapidly increased in the past ten years, reaching 44% in 2013. At the same time, the share of waste landfilled decreased to 58% in 2013. The MSW management system includes two landfills, four incineration plants, and a biogas plant for food waste treatment. The first phase of the MSW landfill at the Hangzhou Tianziling Solid Waste Landfill site was built in 1991 and closed in 2006. The second phase of the Tianziling landfill is expected to reach its full capacity in 5–6 years due to the significant increase in the amount of MSW generated and an insufficient growth in the city's incineration capacity (Hangzhou Municipal Solid Waste Disposal center, 2014).

Hangzhou MSW management includes a four-bin collection system for waste, which was established in Hangzhou in 2010 (Dong et al., 2013). Once collected, the waste is transported directly to the treatment site without the use of transfer stations. The four-bin source separation system enables hazardous waste, food waste, recyclables, and other waste to be collected separately. In addition, unofficial agents collect waste from households and bins before selling this waste to recyclers.

As is the case with the majority of MSW in China in general, Hangzhou's MSW contains a large percentage of food waste, and this varied from 47% to 64% between 2005 and 2011. The second largest waste fraction was plastic waste (14%–27%). Together with paper, the third largest component of waste, these three waste types represent between 70% and 90% of the total MSW mass. The composition of waste documented in the study is presented in Fig. 2.

A environmental impact assessment of the MSW management system in Hangzhou was conducted using three scenarios: a present conditions case scenario and two scenarios involving RDF production from MSW. The present condition scenario (Scenario 0) represented the state of the MSW management system in 2013. Scenario 1 represented RDF production and incineration at three waste incineration plants (Qiaosi, Yuhang, and Xiaoshan) replacing MSW and coal co-incineration, while the Lvneng incineration plant, where no coal is used, would continue MSW co-incineration. The fuel energy of RDF in Scenarios 1 and 2 was assumed to be the same as the total fuel energy of coal and MSW in Scenario 0. In Scenario 2, the same RDF mass (same fuel energy) as that applied in Scenario 1 was assumed to be directed to several new waste incineration plants (hypothetically replacing the old plants in Qiaosi, Yuhang, and Xiaoshan) with higher electric efficiency plants. Table 1 summarizes the MSW mass directed to incineration, RDF production, or landfilling in the scenarios studied.

Four treatment possibilities for the organic reject from RDF production were considered in Scenarios 1 and 2: disposal to landfill (1), biodrying (2), anaerobic digestion (3), or ethanol production (4). It was assumed that the organic fraction of waste was disposed of to the Tianziling landfill site. It was also assumed that the biodried organic reject was incinerated in the same plant as that in which the RDF was produced. The incineration of the organic reject after biodrying reduced the mass of the RDF compared to the scenarios without the incineration of the organic reject because the total fuel energy that went to incineration was assumed to remain the same in all Scenarios 0, 1 and 2. It was assumed that the digestate from anaerobic digestion of the organic fraction was directed to pile composting and then used as landfill cover material.

The system boundary of the study (Fig. 3) included the transport of MSW to the incineration plants and landfills, the unit operations at the treatment sites, and the unit operations required to produce the energy need for transportation and unit operations. The direct emissions from the operations and the indirect emissions produced during the process of procuring fuels and electricity were both accounted for. The present anaerobic digestion of source-separated food waste at the Tianziling landfill site was excluded from the environmental impact assessment because the only data obtained from this plant were the mass flow of food waste to the plant (200 t/d) and the share of reject from that food waste stream (one-third is rejected). The emissions that were avoided by displacing the average electricity production in China with the electricity produced from the MSW incineration and as a result of the gas turbine utilizing LFG from the Tianziling landfill were also accounted for. There is no district heat demand in Hangzhou; therefore, only electricity was recovered.

2.2. Inventory analysis

Inventory analysis is one of the most resource-intensive processes involved in an environmental impact assessment. In the current study, the inventory data were collected using a process-

Table 1
The MSW mass directed to incineration, RDF production, and landfill disposal in the scenarios studied, including method for organic reject treatment (Landfill = LF, anaerobic digestion = AD, ethanol production = EtOH and biodrying).

Scenario		0	1.1 & 1.3 & 1.4	1.2	2.1 &2.3 & 2.4	2.2
Org. Reject treatment		–	LF, AD or EtOH	Biodrying	LF, AD or EtOH	Biodrying
Lvneng	kt/a	204	204	204	204	204
Qiaosi	kt/a	411	490	394	0	0
Yuhang	kt/a	256	343	276	0	0
Xiaoshan	kt/a	416	604	487	0	0
New plants	kt/a	0	0	0	1494	1201
Landfill Liugongduan	kt/a	366	13	293	0	248
Landfill Tianziling	kt/a	1432	1432	1432	1388	1432
Total	kt/a	3086	3086	3086	3086	3086

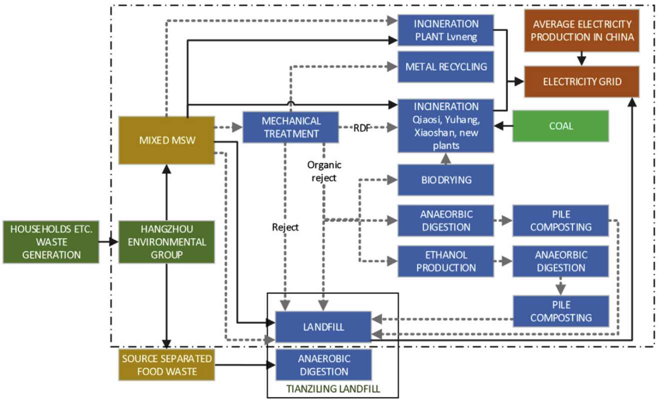

Fig. 3. Hangzhou MSW management Scenario 0 (solid line) and Scenarios 1 and 2 (dotted line).

LCA technique that included collecting data from the inputs and outputs of the unit processes.

2.2.1. Landfilling

In 2013, 56% of the mixed MSW in Hangzhou was disposed of in the Liugongduan and Tianziling landfills. The landfill disposal was calculated according to the information gathered during previous studies (Chi et al., 2014; Dong et al., 2014a, 2014b; Nielsen et al., 2008) and the information received during site visits to the Tianziling landfills, as summarized in Table 2. LFG was only collected at the Tianziling landfill site, and the gas collected was directed to a gas engine. Diesel was consumed by the bulldozer used at the landfill, and the emissions of this bulldozer are summarized in Supplementary material SI2.

2.2.2. The incineration of MSW

MSW incineration in Hangzhou is primarily based on fluidized bed technology because three of the four incineration plants utilize fluidized bed technology (Qiaosi, Yuhang, and Xiaoshan), with only

Table 2
Parameters for calculating MSW disposal into landfill.

Parameter	Value	Unit
Diesel use	0.00014	kg diesel/kg MSW
LFG	0.120	m^3/kg MSW
LFG CH4 content	50	%
LFG CH4 oxidation	10	%
LFG collection		
Liugongduan	0	%
Tianziling	25	%
LFG combustion in gas engine		
Electric efficiency	39	%
CH_4 emission	0.000323	kg CH_4/MJ
N_2O emission	0.0000005	kg N_2O/MJ
Leachate treatment		
Leachate	0.2	kg/kg MSW
Electricity use	0.0015	MJ/kg MSW
CH_4 emission	0.00006	kg/kg MSW
Leachate pollutants after treatment		
NH_3	0.00081	kg/kg leachate
P tot	0.000012	kg/kg leachate

Lvneng using grate technology. Data on the incineration plants were collected by visiting the incineration plants and through surveys with the plant operators. The three fluidized bed boiler incineration plants utilized coal as an auxiliary fuel. It was assumed that the coal was transported from a coal mine that was located 600 km away from Hangzhou. The operations data of the four plants are summarized in Table 3, and these data were used to calculate Scenario 0. More information about the waste incineration plants is presented in Supplementary material SI3. The lower heating value of MSW as received (LHV_{ar}) in Hangzhou was low, varying between 4 and 5 MJ/kg. In all of the incineration plants, a small share of metal (0.07% of the utilized MSW mass) was recovered in a pretreatment phase and then recycled. It was assumed that the composition of this metal was 50% steel and 50% aluminum (Dong et al., 2014a). The metal recycling emission credits are presented in Supplementary material SI3. It was assumed that the coal came from the Huaibei coal mine, which is approximately 600 km from Hangzhou.

The calculation accounted for the flue gas emissions and the treatment of solid residues. The average fossil carbon content of MSW was calculated to be 12.5% of the mixed MSW mass. This calculation is presented in Supplementary material SI3. The other emissions were obtained from the plant operators and are summarized in Table 4. The bottom ash was assumed to be made into bricks by adding cement to 30% of the bottom ash mass and sold directly from the plant. The transportation of any sold bricks was not included in the calculations. The fly ash was solidified by adding water 5% and cement 3% to the fly ash mass. The solidified fly ash was then transported to the landfill. It was assumed that the cement was transported from a nearby cement mill in Hangzhou. The distances the mixed MSW, fly ash, and cement were transported are summarized in Supplementary information SI1.

2.2.3. Refuse derived fuel production and incineration

In Scenarios 1 and 2, the MSW was assumed to be directed to a mechanical treatment center located near the incineration plants for the production of RDF. The modeled mechanical treatment line included shredding, screening for organic fraction separation, magnetic separation of ferrous metals, eddy current separation of non-ferrous metals, and air separation for heavy fraction separation. The recovery rates (shares removed from the material stream

Table 3
Operations data for the four existing incineration plant in Hangzhou.

Operation data			Lvneng	Qiaosi	Yuhang	Xiaoshan
MSW	Mass	kt/a	204	411	256	416
	LHV_{ar}	MJ/kg	3.9	4.2	4.2	4.6
Coal	Mass	kt/a	0	2	8	23
	Share	% of MSW	0	0.57	3.2	5.5
	LHV_{ar}	MJ/kg	0	21	21	21
Fuel energy	MSW	GWh/a	220	480	299	531
	Coal	GWh/a	0	14	46	134
	Share	% of total	0	3	13	20
	Total	GWh/a	220	494	345	666
Electricity efficiency		%	24	20	16	19
Own use		% of produced	19	29	25	21
Ash		of MWS	20	16	18	22
Bottom ash		of ash	85	63	55	55
Fly ash		of ash	15	38	45	45

going to RDF) of the machinery used were calculated according to the approach employed by Nasrullah et al. (2015), who previously studied the production of RDF from MSW. The process is summarized in Supplementary material SI4. The energy consumption of the RDF production line was assumed to be 70 kWh/t MSW (Nasrullah et al., 2015). According to the calculation, 68% of the MSW ended up as RDF, and approximately 30% was removed as organic reject. The LHV_{ar} of the RDF was calculated by assuming that the RDF contained 86% of the energy content of the MSW (Nasrullah et al., 2015). The outputs of the mechanical treatment line for the scenarios in which the organic reject was not incinerated (i.e., Scenarios 1.1, 1.3, 1.4, 2.1, 2.3, and 2.4) are summarized in Table 5.

2.2.4. Organic reject treatment

Four methods of treating the organic reject that was generated during the RDF production were considered: landfilling, biodrying, anaerobic digestion, and ethanol production. The first method involved directing the organic reject to Tianziling landfill, where the LFG generation was estimated to be 0.12 m^3/kg (Dong et al., 2014a).

The second treatment method involved biodrying, which was assumed to occur close to the incineration and RDF plants. The biodrying process was assumed to be an aerobic process through which the moisture content was reduced, thereby improving the combusting properties of wet organic reject (Zhang et al., 2009) and making it suitable for co-incineration with the produced RDF. The life cycle inventory (LCI) data for the biodrying process and its associated inputs and outputs are summarized in Supplementary material SI5.1.

Because the fuel energy directed to incineration was assumed to be same in the plants and the incineration of the dried organic reject reduces the need for RDF, the mass of RDF and organic reject was calculated so that the fuel energy of a given plant remained the same in all scenarios. The outputs of the mechanical treatment and subsequent biodrying of organic reject are summarized in Table 6. The LHV_{ar} of the RDF that was produced were the same in Scenarios 1 and 2.

The third method was wet mesophilic anaerobic digestion. The biogas was directed to a gas turbine for the purposes of generating electricity and heat. The electricity that was produced was directed to the electricity grid, and part of the produced heat was directed for heating the reactor. It was assumed that any residual heat that was generated was wasted because there is no district heat grid in Hangzhou. The generated digestate was assumed to be dewatered by a centrifuge and treated by pile composting. It was also assumed that the rejected water was directed to dilute the incoming feedstock in order to reduce the need for fresh water. The LCI data of the anaerobic digestion and pile composting processes are summarized in Supplementary material SI5.2. It was assumed that the compost was used as a landfill cover at the Tianziling landfill site.

The fourth method was ethanol production. This method was considered because the biowaste contains sugars that can be fermented into ethanol. The ethanol production was assumed to occur

Table 4
Flue gas emissions of existing incineration plants in Hangzhou.

	Lvneng	Qiaosi	Yuhang	Xiaoshan
Dust (kg/a)	7000	27 000	18 000	39 000
SO_2 (kg/a)	26 000	27 000	17 000	38 000
HCl (kg/a)		6800	4400	9600
NO_2 (kg/a)	110 000	150 000	94 000	200 000
CO (kg/a)		62 000	40 000	85 000
Hg (kg/a)		7	4	9
Cd (kg/a)		3	2	4
Pb (kg/a)		3	2	4
Dioxin (mg/a)		740	480	1000

Table 5
Mechanical treatment outputs in Scenarios 1.1, 1.3, 1.4, 2.1, 2.3, and 2.4.

	Unit	Qiaosi	Yuhang	Xiaoshan	New plants
RDF	kt/a	335	235	413	1022
RDF LHV_{ar}	MJ/kg	5.3	5.3	5.8	5.3
Fuel energy	GWh/a	494	345	666	1505
Organic reject	kt/a	149	105	184	456
Metal	kt/a	3	2	4	10
Non-magnetic metal	kt/a	1	0	1	2
Heavy reject	kt/a	1	1	2	5

Table 6
Mechanical treatment outputs in Scenarios 1.2 and 2.2.

	Unit	Qiaosi	Yuhang	Xiaoshan	New plants
RDF	kt/a	270	189	333	822
RDF LHV_{ar}	MJ/kg	5.3	5.3	5.8	5.3
Dried org. reject	kt/a	58	41	72	177
Dried org. reject LHV_{ar}	MJ/kg	6.0	6.0	6.5	6.0
Fuel energy	GWh/a	494	345	666	1505
Metal	kt/a	3	2	3	8
Non-magnetic metal	kt/a	0	0	1	1
Heavy reject	kt/a	1	1	1	4

Table 7
The parameters applied in the sensitivity analysis.

Parameter	Scenario	Values applied			
		Low	High	Default	
LFG collection rate	All scenarios	15	35	25	%
LFG yield	All scenarios	0.1	0.14	0.12	m^3/kg MSW
Electric efficiency	Scenario 0, 1	15	24	19	%
Electric efficiency	Scenario 2	26	32	29	%
LHV MSW	Scenario 0	3.5	5.0	4.3	MJ/kg
LHV RDF	Scenario 1.1–0.4	4.6	6.4	5.5	MJ/kg
LHV RDF	Scenario 2.1–2.4	4.4	6.2	5.3	MJ/kg
LHV dried org. reject	Scenario 1.2	5.4	7.0	6.2	MJ/kg
LHV dried org. reject	Scenario 2.2	5.2	6.8	6.0	MJ/kg

close to the RDF plant. The LCI data of the ethanol production process are summarized in Supplementary material SI5.3. The production of ethanol requires alpha-amylase enzymes and glucoamylase enzymes, and the emissions from the enzyme production were calculated according to approach recommended by Nielsen et al. (2007). The stillage was assumed to be directed to dry anaerobic digestion, and the digestate was further directed to dewatering and pile composting, similar to the digestate from the anaerobic digestion of organic reject. The LCI data related to the dry anaerobic digestion are summarized in Supplementary material SI5.3. It was assumed that the compost was used as a landfill cover at the Tianziling landfill site.

2.2.5. *Uncertainty analysis*

The parameters of the mixed MSW management system studied were expected to vary. Therefore, an uncertainty analysis was performed to assess how different parameters affected the results of the present environmental impact assessment study. The range of values for the parameters used in the analysis was selected from

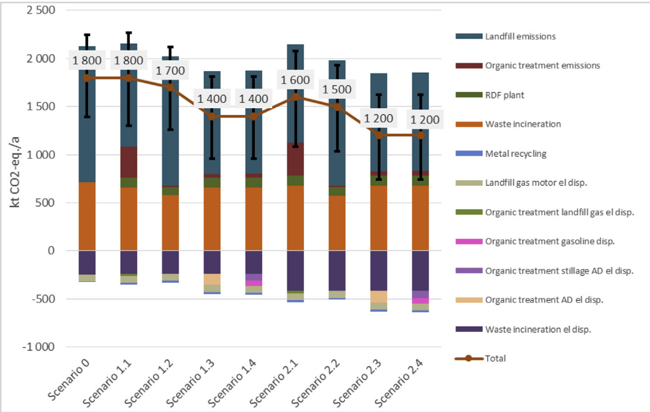

Fig. 4. Annual global warming potential ($kt_{CO2,eq.}$/a) of Hangzhou MSW management.

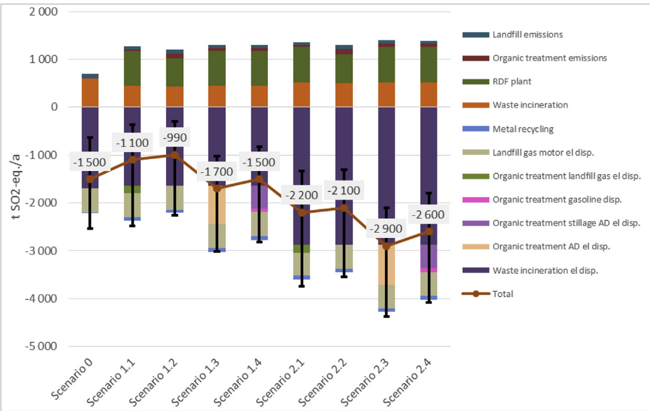

Fig. 5. Annual acidification potential ($t_{SO2,eq.}$/a) of Hangzhou MSW management.

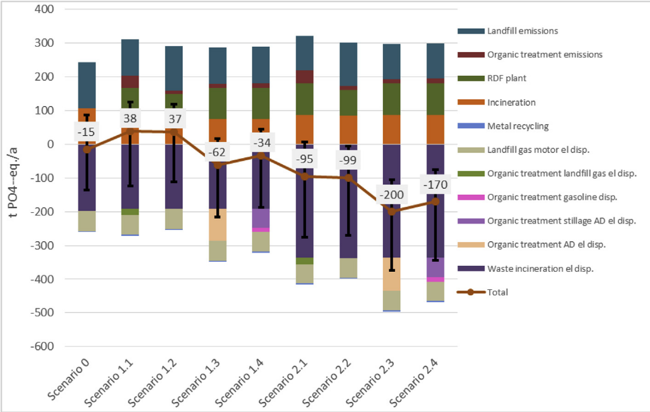

Fig. 6. Annual eutrophication potential ($t_{PO4,eq.}$/a) of Hangzhou MSW management.

the references used to obtain the default values for these parameters. These default values were then used to calculate the minimum and maximum values of the net impact assessment results for all scenarios. The highest uncertainty in the data was related to the LHV of waste. In the uncertainty analysis, the LHV of MSW was changed within a reasonable range of values appropriate for MSW in China, which, in turn, changed the LHV of RDF and the dried organic reject. Another significant factor was the electric efficiency, which also varied. Landfilling mixed MSW was anticipated to have a significant impact on the results; as such, the LFG yield in the Tianziling and Liugongduan landfills were changed, as was the rate at which LFG was collected in the Tianziling landfill. The parameters used in the analysis are summarized in Table 7.

2.3. Impact assessment

In the impact assessment phase, the potential environmental impacts were evaluated using the inventory analysis data. In this phase, the inventory data is associated with impact categories and indicators (ISO 14040, 2006; ISO 14044, 2006). In the current study, the environmental impact categories included global warming potential (GWP), acidification potential (AP), and eutrophication potential (EP). The impact assessment was conducted until the characterization phase, and there was no normalization or weighting of the results. The impact assessment was performed using GaBi 6.0 software and the CML2010 April 2013 life cycle impact assessment methodology (Thinkstep, 2015).

3. Results and discussion

The results of the life cycle assessment of the study scenarios are presented as total values of GWP (Fig. 4), AP (Fig. 5), and EP (Fig. 6). The figures also include the results of the uncertainty calculation, which are represented by the error bars. The order of scenarios remained unchanged during the uncertainty analysis. According to the uncertainty calculation, the highest uncertainty was related to the EP values with the changes related to the default value being, on average, between −322% and 198%. The change in GWP results was between −31% and 29%, and the change with the AP results between −81% and 46%. The results highlighted that the annual GWP was lower in the alternative MSW management systems represented by Scenarios 1 and 2 than it was in the reference scenario. The GWP was reduced by between 0 and 33%, with the best scenarios being 2.3 and 2.4, where the MSW was directed to RDF production and the resulting RDF was incinerated in new plants that had a higher electric efficiency, with the resulting organic reject directed to energy recovery by anaerobic digestion (Scenario 2.3) or ethanol production (Scenario 2.4). In all of the scenarios, the net GWP was positive, which was mainly due to the high emissions from landfilling and the lack of emission reductions from displacing fossil heat production because there was no heat recovery from waste incineration.

In all scenarios, the landfill emissions caused the highest share of GWP, representing 48–67% of the total GWP. Incineration, the second most polluting activity, accounted for 29–37% of the total GWP. The organic treatment emissions in Scenarios 1.1 and 2.1 were 7–23 times higher than those in other scenarios due to the LFG emissions from directing organic reject into landfill, being 15% and 16% of the emissions in Scenarios 1.1 and 2.1 respectively. The emissions avoided from electricity displacement (el disp.) generated from waste incineration compared to the emissions from waste incineration were 36–41% in old plants and 62%–73% in new plants, where the electric efficiency was higher.

Electricity use and avoided electricity production are playing a greater role in emissions that cause acidification than it does in

Table 8
Calculated net emission factors of utilizing MSW, RDF, and organic reject in Hangzhou in relation to the processed waste mass.

	GWP	Acidification	Eutrophication
	$kg_{CO2,eq.}$/t	$g_{SO2,eq.}$/t	$g_{PO4,eq.}$/t
Landfill	710	−280	36
Incineration			
MSW	360	−860	−72
RDF (old plants)	400	−1300	−120
RDF (new plants)	230	−2400	−250
RDF + dried reject (old plants)	340	−1300	−130
RDF + dried reject (new plants)	140	−2400	−250
RDF plant	74	500	63
Org reject landfill	690	−280	40
Org reject AD	−190	−1700	−190
Org reject EtOH	−180	−1100	−120

Table 9
Net emission factors of MSW and RDF management LCA studies.

	GWP	Acidification	Eutrophication	Reference
	kgCO2,eq./t	gSO2,eq./t	gPO4,eq./t	
Landfill				
MSW	−69−162	–	–	(Manfredi et al., 2009)
MSW	490	−440	–	(Arena et al., 2003)
OFMSW	843	104	–	(Evangelisti et al., 2014)
MSW	234	–	–	(Hupponen et al., 2015)
MSW	595−1311	127−374	86	(Cherubini et al., 2009)
Incineration				
MSW	−844−−126	–	–	(Astrup et al., 2009)
MSW	390	–	–	(Arena et al., 2015)
MSW	46	−4600	–	(Arena et al., 2003)
RDF	95	−3660	–	(Arena et al., 2003)
RDF co-combustion	−1512−563	–	–	(Astrup et al., 2009)
MSW	−92−140	–	–	(Hupponen et al., 2015)
MSW	273	2370	354	(Chaya and Gheewala, 2007)

emissions that cause GWP. In the current study, the net AP was negative in all scenarios due to the high emissions reductions caused by displacing the average electricity production with the electricity produced from waste. The emissions from the incineration caused the main emissions in Scenario 0 (86%); however, in the remaining scenarios, the electricity used to produce RDF caused the highest emissions (46%–57%), followed by incineration (34–39%). Scenarios 1.1 and 1.2 had a higher net AP than the reference scenario because of the emissions from electricity use for RDF production. In Scenarios 1.3 and 1.4, the organic reject treatment produced additional electricity, which resulted in a higher emission displacement than that observed in Scenarios 1.1 and 1.2. The scenario that involved a new incineration plant that combined RDF incineration and anaerobic digestion of organic reject proved to be the best option in terms of the net AP.

The emissions causing eutrophication are similar to the GWP caused primarily by the emissions that resulted from the disposal of MSW into landfills. In Scenario 0, they are more important and are responsible for 56% of all emissions, while in other scenarios, they are slightly less significant and cause 35%–45% of all emissions. The use of electricity in RDF plants is also a significant source of emissions and is responsible for 25%–35% of the total emissions. The net EP is negative in most of the scenarios due to the emissions that are avoided as a result of electricity displacement. Scenarios 1.1 and 1.2 have higher net emissions than the reference Scenario 0 because of the increased emissions caused by the RDF plant's electricity use, while the avoided emissions were not increased as much. The net emissions from Scenarios 1.3 and 1.4 were lower than those in Scenario 0 because of the increased emission reductions caused by electricity production from the organic reject.

The calculated net emission factors of different treatment possibilities are summarized in Table 8. Landfilling waste results in higher net GWP potential than incineration due to the methane emissions contained in LFG and the emissions that are avoided from electricity produced by MSW or RDF incineration. The importance of utilizing organic reject from RDF production in energy recovery rather than landfilling is highlighted. As an example, the net GWP from landfilling organic reject can be as high as 690 $kg_{CO2,eq}$/t, whereas energy production from organic reject would result in negative net GWP.

The focus of previous LCA studies on MSW management in the literature has been primarily on GWP, and less attention has been given to AP and EP. A short review of the existing literature that describes incineration and landfilling, which are the two main treatment methods assessed in this study, is presented in Table 9. The net GWP of incineration of MSW and RDF from this study is at the higher end in comparison to that presented in the literature. This is most likely the result of the lack of heat recovery from incineration in Hangzhou. Similarly, the net GWP of landfilling waste is greater than the average in the literature, which can be primarily attributed to the relatively low LFG collection rate in the Tianziling landfill and the lack of LFG collection in the Liugongduan landfill.

4. Conclusions

The life cycle assessment study presented in this paper demonstrated that the environmental situation in Hangzhou could be improved by changing the MSW management system that is currently employed in the area. The highest improvement potential arises from producing RDF that is of a higher quality than the original MSW for energy recovery. However, the benefits gained may be easily diminished if the organic reject from RDF production is landfilled. To prevent this, the organic reject from RDF production should, instead, be used in energy recovery; e.g., by anaerobic digestion. In addition, the environmental situation in Hangzhou could be improved even further if the waste incineration plants combined heat and power production to produce heat for industry. Still, the main problem in Hangzhou is linked to inefficient source separation, especially in relation to food waste, which deteriorates the quality of any MSW that is directed to incineration. Reducing the share of food waste would mean that the heating value of MSW would be higher, more electricity from the incineration could be produced, the auxiliary coal in waste incineration could be reduced, less reject from the RDF would be produced, and RDF production energy demand would be lower. In conclusion, technologically advanced systems could partly improve the environmental situation, while officials from Hangzhou's MSW management should treat educating the general public about the benefits of source separation as a top priority.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2016.09.146.

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Publication II

Liikanen, M., Havukainen, J., Hupponen, M., and Horttanainen, M.

Influence of different factors in the life cycle assessment of mixed municipal solid waste management system – A comparison of case studies in Finland and China

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Influence of different factors in the life cycle assessment of mixed municipal solid waste management systems – A comparison of case studies in Finland and China

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A B S T R A C T

The life cycle assessment (LCA) of municipal solid waste (MSW) management systems is typically rather arduous due to extensive data acquisition needed to calculate the direct and avoided emissions of the systems. A possibility to diminish the workload of the LCA studies is to utilise default or generic data instead of direct and case-specific data. However, it is crucial to know when this is justified. Direct and case-specific data should be applied at least to the key processes and parameters which have the strongest influence on the total results, whereas default data can be applied to the processes and parameters which have only a minor influence on the total results.

Mixed MSW management systems in the South Karelia region, Finland, and the city of Hangzhou, China, were compared in this study in terms of the influence of different factors on the LCA results of the systems. The comparison focused particularly on the influence of individual parameters on the global warming, acidification and eutrophication potentials of the LCA studies. According to the study, parameters directly related to the generation and collection of landfill gas, the energy and fossil carbon content of mixed MSW, energy production efficiencies, as well as the nitrogen oxide and sulfur dioxide emissions of incineration had the highest influence on the total results in both case studies, and therefore direct, case-specific data should be applied particularly to them. The use of machinery in landfilling, the electricity and chemical consumption in leachate treatment, the transportation of auxiliary materials (e.g. chemicals and incineration residues) as well as the electricity consumption and the use of machinery in bottom and boiler ash treatment had instead only a minor influence on the total results. Default or generic data could be applied to them to diminish the workload of the LCA studies. It is worth mentioning that the findings of the study apply merely to these particular case studies. Further research and corresponding comparisons are required to draw more profound and general conclusions.

1. Introduction

Waste is a worldwide issue. Particularly due to population growth and urbanisation in developing countries, the generation of municipal solid waste (MSW) has increased significantly over the past decades. For instance, the global MSW generation rate is expected to double by 2025 from the generation rate in 2012 (World Bank, 2012). Alongside the increase in MSW generation, the environmental impacts of MSW have been more comprehensively identified globally. The growing awareness of the negative environmental impacts of MSW has increased the use of life cycle assessment (LCA) methodology in the MSW management sector. By means of LCA, the potential environmental impacts of MSW management systems can be evaluated (EN ISO 14040, 2006; EN ISO 14044, 2006). LCA enables taking into account both direct (i.e. emissions from treatment processes) and avoided (i.e. emissions avoided due to energy or material substitution) emissions of MSW management processes (Ekvall et al., 2007). Laurent et al. (2014) conducted a comprehensive review of the application of LCA to MSW management systems. According to the study, LCA was first conducted on MSW management systems in the 1990s, and currently it is a widely used method in the assessment of the environmental impacts of MSW management systems. The LCA of MSW management systems has been primarily applied in high

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income countries, particularly in Europe. It has also gained popularity in lower income countries during the past decade due to increased MSW generation and urbanisation. For instance, several MSW LCA studies have been conducted in China in recent years.

LCA studies of MSW management systems are typically highly case-specific, depending on the objective of the study and local conditions and features. Nevertheless, the purpose of most LCA studies is the comparison of different treatment and management options for MSW. For instance, De Feo and Malvano (2009) assessed the environmental impacts of 12 different management options for MSW in a region in South Italy to select the best MSW management system for the region. LCA has also been used to compare different source separation and collection systems: for instance, Larsen et al. (2010) assessed five scenarios with alternative collection systems for recyclables by means of LCA, and Rigamonti et al. (2009a) utilised LCA in the optimisation of collection systems for recyclables. Additionally, LCA has widely been used as a decision support tool for policy making in the field of MSW management. For instance, Turner et al. (2016) and Lazarevic et al. (2012) introduced different approaches to how the LCA of MSW management systems can be utilised as a decision support tool.

The intricacy of MSW management systems poses challenges for LCA studies. Of the main phases of LCA (i.e. goal and scope definition, inventory analysis, impact assessment and interpretation) (EN ISO 14040, 2006), particularly inventory analysis is highly time and resource-consuming due to the comprehensive data acquisition needed to calculate the direct and avoided emissions of the system. Various approaches have been developed to facilitate and simplify LCA (e.g. Fleischer et al., 2001). A simple and straightforward way to diminish the workload of MSW LCA studies is to use default or generic data (i.e. secondary data) instead of direct and case-specific data (i.e. primary data) in inventory analysis. In order to do that without reducing the reliability of the results, it is important to know the influence of an individual parameter on the total results. Therefore, the following straightforward rule of thumb should be retained: one can apply default or generic data to parameters with a minor influence on the total results while simultaneously applying direct and case-specific data to other parameters in order to maintain the reliability of the LCA study.

The influence of an individual parameter on the total results can be identified by sensitivity analysis, which assesses the effect of input parameters' changes on the total results. The more sensitive the result is to a given parameter, the more case-specific and reliable the data concerning the parameter should be. Direct data should be used at least concerning the key parameters which have the highest influence on the overall environmental performance of MSW management systems. Regarding the LCA of MSW management systems, the key processes and parameters have been rather well recognised in literature (see Table 1). The environmental impacts of surrounding systems, e.g. electricity and heat production, often override the environmental impacts of the MSW management system itself (Ekvall et al., 2007). Parameters related to energy and material recovery and substitution (e.g. electricity and heat production efficiencies, material recovery efficiency) are therefore particularly important in MSW LCA studies. While previous research has particularly focused on the key processes and parameters of MSW management LCA studies, little research has been conducted to identify the processes and parameters which have only a minor influence on the total results. Nevertheless, they are crucial in terms of the above-mentioned simplification possibility, i.e. using default or generic data instead of direct and case-specific data.

Two different case studies are compared in this study: the South Karelia region in Finland and Hangzhou city in China (see Fig. 1). South Karelia is a region in South-East Finland, and it consists of nine municipalities. Hangzhou is the capital city of the Zhejiang Province in Eastern China. In both case studies, mixed MSW (i.e. the remaining part of MSW after the source separation of different waste fractions) management system of the area is investigated by means of LCA. The case studies have been initially reported by Hupponen et al. (2015) and Havukainen et al. (2017). The comparison of the case studies focuses particularly on different input parameters used in the LCA of the mixed MSW management systems. The objective of the study is to determine the most and least important (i.e. sensitive) input parameters of the case studies in order to identify possibilities to simplify their LCA by using default or generic data instead of direct and case-specific data.

Table 1
Typical key factors in the LCA of MSW management systems presented in literature (literature studies particularly focusing on the subject are listed as references).

MSW management phase	Key factor	Reference
MSW generation	Waste composition	Slagstad and Brattebø, 2013
	Source-separation efficiency	Rigamonti et al., 2009b
Landfilling	Collection of landfill gas (LFG) and leachate	Manfredi and Christensen, 2009
Incineration	Energy recovery and substitution	Burnley et al., 2015
Recycling	Material recovery and substitution	Rigamonti et al., 2009b

The research questions are the following:

- What are the key factors, i.e. processes and input parameters, in the case LCA studies on South Karelia, Finland, and Hangzhou, China?
- Which factors have instead only a minor influence on the total results in the case areas?
- How could the LCA of the case studies be simplified by using default or generic data instead of case-specific, direct data?

2. Materials and methods

2.1. Description of the case areas

The South Karelia region in Finland and Hangzhou city in China were selected as the case areas for the study to analyse both high income and lower income countries' mixed MSW management systems (see Supplementary material A for further information). They represent distinctly different areas (e.g. population, geographical location, income level) and mixed MSW management systems, however with some similarities, which enable the comparison between them. For instance, incineration is a treatment method for mixed MSW in both areas. Since the case studies differ from each other in many respects, the similarities between them can be an indication of a more extensive phenomenon. In other words, if the influence of a given parameter on the total results is similar in both case studies, the same phenomenon can be valid in other mixed MSW management systems, too.

Key data (i.e. population, MSW generation rate, the composition of mixed MSW and collection system) concerning the case areas' MSW management systems are presented in Fig. 2. In South Karelia, all mixed MSW generated in the region was landfilled until 2013. The incineration of mixed MSW started in 2013 and has increased in stages. Currently, all mixed MSW generated in the region is incinerated. Since there is no waste incineration plant in the region, mixed MSW is transported to a waste incineration plant in Riihimäki which is located approximately 220 km from the region. (Etelä-Karjalan Jätehuolto Oy, 2016.) In Hangzhou, incineration and

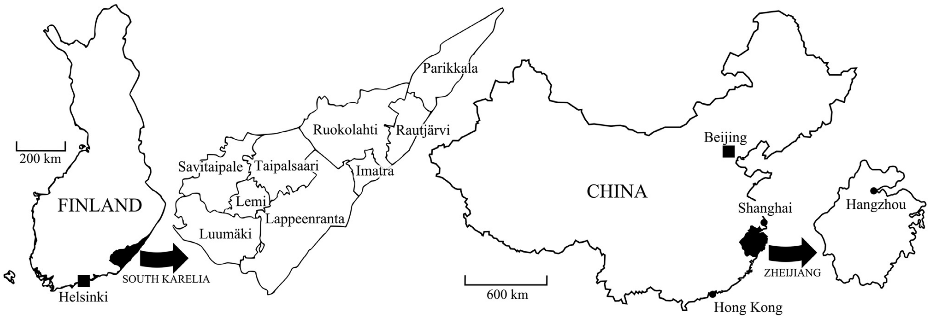


Fig. 1. Case study areas.

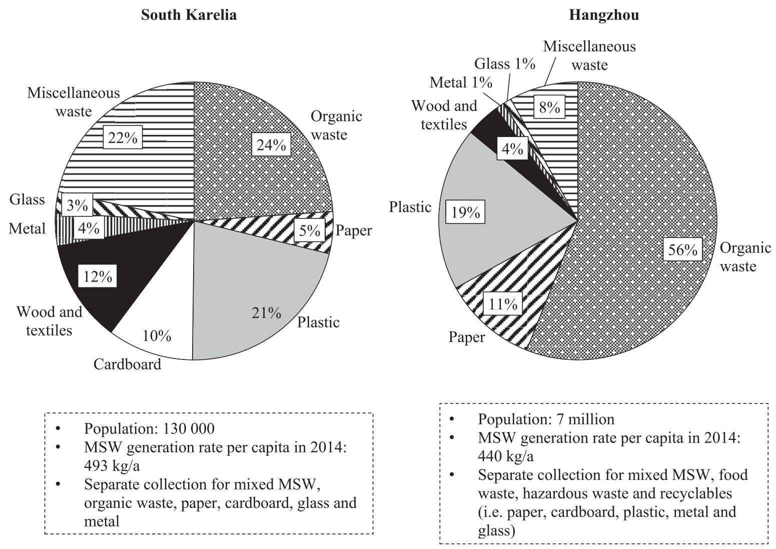


Fig. 2. Key data of the case areas' MSW management systems (Horttanainen et al., 2013; Regional Council of South Karelia, 2016; Lappeenranta, 2016; Eurostat, 2016; Havukainen et al., 2016; Dong et al., 2013).

landfilling are the main treatment methods for mixed MSW. In 2013, 58% of mixed MSW was landfilled and the rest incinerated. At present, there are two landfills and four incineration plants in Hangzhou (Havukainen et al., 2017).

2.2. Life cycle assessment

As mentioned previously, the case studies have been initially reported by Hupponen et al. (2015) and Havukainen et al. (2017) in peer-reviewed literature. Thus, the validity of the case studies has already been checked. However, the South Karelia case study (Hupponen et al., 2015) was significantly modified in this study to enable the comparison between them (see Chapter 2.3.1 for further information). The LCAs of both cases, South Karelia and Hangzhou, were carried out according to the ISO standards 14040 and 14044 (EN ISO 14040, 2006; EN ISO 14044, 2006). The GaBi 6.0 LCA modelling software was used in both studies (Thinkstep, 2016). CML 2001 – November 2010 was used for impact assessment in the South Karelia case study, and CML 2010 – April 2013 was used in the Hangzhou case study. These particular versions of CML were used in this study since they were also applied by Hupponen et al. (2015) and Havukainen et al. (2017). The impact categories were the global warming potential (GWP) for a 100 year time span, acidification potential (AP) and eutrophication potential (EP) in both studies. According to a review by Cleary (2009), these impact categories have been most commonly applied in the LCA of MSW management systems. The functional unit of the LCA studies was the same in both studies, i.e. the treatment of mixed MSW generated in the areas during a year: 22 500 t in the South Karelia study (Etelä-Karjalan Jätehuolto Oy, 2013; Statistics Finland, 2016) and 3086 kt in the Hangzhou study (Havukainen et al., 2017).

2.3. Scenarios and calculation principles

2.3.1. South Karelia

The scenarios of the South Karelia case study are the same as in a study by Hupponen et al. (2015), i.e. the regional mixed MSW management situation in 2012 is assessed (see Fig. 3). There are two main scenarios: landfilling (Scenario 0) and incineration (Scenario 1). Additionally, there are three different sub-scenarios in the incineration scenario: Riihimäki (Scenario 1.1; the situation in 2012, i.e. without plastic and bio refineries which currently operate in the plant), Kotka (Scenario 1.2) and Leppävirta (Scenario 1.3), which are cities rather close to South Karelia and represent different treatment options for mixed MSW generated in South Karelia. The sub-scenarios are rather different from each other. First of all, the incineration scenarios have different transportation distances: 220 km (Scenario 1.1), 120 km (Scenario 1.2) and 210 km (Scenario 1.1). Another distinct difference between the scenarios is the incineration technology. Mixed MSW is incinerated in a grate furnace in Scenarios 1.1 and 1.2, whereas in Scenario 1.3, refuse-derived fuel (RDF) is produced from mixed MSW and incinerated in a fluidised bed boiler. Additionally, the substituted heat production differs between the scenarios. In Scenarios 1.1 and 1.2, the produced district heat substitutes heat produced by natural gas, whereas produced heat substitutes biofuels (72% of the heat production), plastic waste (19%), heavy fuel oil (7%) and coal (2%) in Scenario 1.3.

Hupponen et al. (2015) assessed the GWP of the management of mixed MSW from regional collection points (approximately 3100 $t_{mixed\ MSW}$/a). In this study, the mixed MSW management of the entire region is assessed instead of mere regional collection points. Additionally, the AP and EP of the mixed MSW management is assessed in this study in addition to GWP. The GWPs of the management scenarios have been calculated similarly as in a study conducted by Hupponen et al. (2015). The data used to calculate the APs and EPs of the scenarios are presented in Supplementary material B. The environmental impacts of capital goods (e.g. trucks, buildings, equipment, etc.) were not taken into account in the study, although according a recent study by Brogaard and Christensen (2016), the capital goods of waste management systems can have a significant influence on the total results. This results from the system boundaries of the case study initially defined by Hupponen et al. (2015).

2.3.2. Hangzhou

The scenarios of the Hangzhou case study are the same as in a study by Havukainen et al. (2017). There are three main scenarios in the Hangzhou LCA study (see Fig. 4): the actual mixed MSW management situation in 2013 (Scenario 0), the production and incineration of RDF at three incineration plants (Qiaosi, Yuhang and Xiaoshan) to replace MSW and coal co-incineration (Scenario 1), and the production and incineration of RDF at new plants with a higher electricity production efficiency (Scenario 2). Additionally, there are four different treatment options for the organic reject generated from mechanical treatment in the LCA study (i.e. four different sub-scenarios): landfill (Scenarios 1.1 and 2.1), biodrying (Scenarios 1.2 and 2.2), anaerobic digestion (Scenarios 1.3 and 2.3)

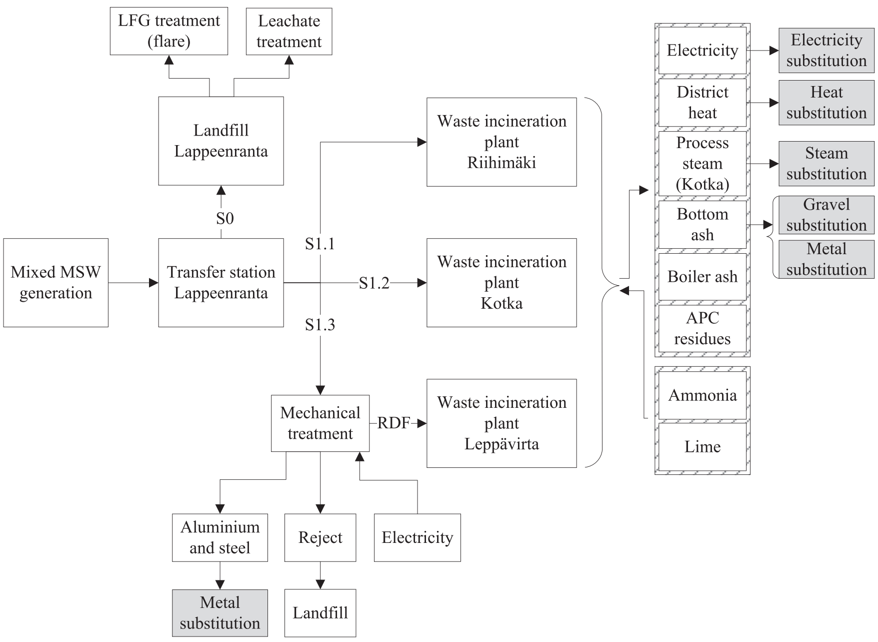
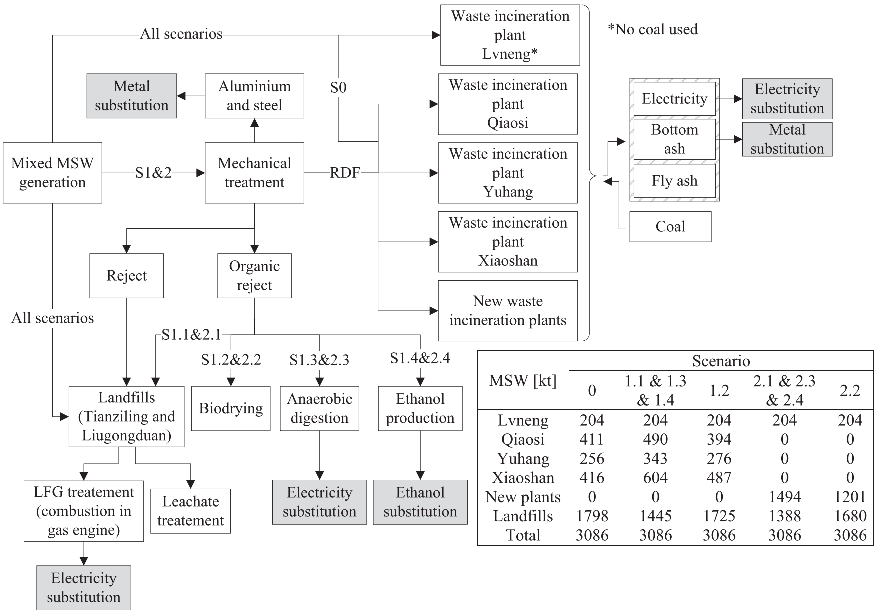


Fig. 3. Mixed MSW management scenarios in the South Karelia case study.



MSW [kt]	Scenario 0	1.1 & 1.3 & 1.4	1.2	2.1 & 2.3 & 2.4	2.2
Lvneng	204	204	204	204	204
Qiaosi	411	490	394	0	0
Yuhang	256	343	276	0	0
Xiaoshan	416	604	487	0	0
New plants	0	0	0	1494	1201
Landfills	1798	1445	1725	1388	1680
Total	3086	3086	3086	3086	3086

Fig. 4. Mixed MSW management scenarios in the Hangzhou case study.

and ethanol production (Scenarios 1.4 and 2.4). As in the South Karelia study, the environmental impacts of capital goods were not taken into account in this study, either, due to the initial system boundaries of the study.

2.4. Comparison of the case studies

2.4.1. Contribution analysis

Contribution analysis is a commonly used method to present the results of LCA studies (Heijungs and Kleijn, 2001). In addition to presenting the results, it is a sensitivity analysis method even though not always identified as one (Clavreul et al., 2012). In contribution analysis, the total LCA result is decomposed into individual process contributions, i.e. the net result, either positive or negative, is presented as a sum of direct and avoided emissions of individual processes. The positive and negative impacts of individual processes are typically separately presented in contribution analysis in order to identify where direct and avoided emissions result from. Therefore, the most and least important processes can be identified through a contribution analysis.

2.4.2. Perturbation analysis

The influence of an individual parameter on the total result can be determined by means of a perturbation analysis, where input parameters are individually varied and the total result is calculated for each variation (Heijungs and Kleijn, 2001). The influence of each variation on the total result can be determined by the following equation:

$$SR = \frac{\frac{\Delta result}{initial\ result}}{\frac{\Delta parameter}{initial\ parameter}} \quad (1)$$

where SR is the sensitivity ratio. As presented in the equation, SR is determined by proportioning the relative change of the total result to the relative change of an individual parameter. (Clavreul et al., 2012.) Thus, the change of a parameter results in an SR-fold change in the total result. For instance, if a parameter has an SR of 5, then a 20% increase in the parameter's value results in a 100% increase in the total result. If the SR of a parameter were negative, the total result would decrease when increasing the value of a parameter. Therefore, the sign of an SR indicates what kind of influence a parameter has on the total result: parallel or reverse. By determining the SRs for the input parameters, the most and least important parameters of the LCA study can be identified. According to Heijungs and Kleijn (2001), parameters with SRs (as absolute values) higher than 0.8 are important. When the absolute value of an SR is higher than 1.0, the parameter can be regarded as particularly important. If the SR of a parameter is less than 0.2, the parameter's influence on the total result is rather minor. These definitions are however only approximate since the magnitude of an SR is highly dependent on the impact category. Therefore, different impact categories' SRs should not be compared with each other, and the sensitivity of parameters should be evaluated within an impact category (Bisinella et al., 2016).

It is worth to mention and emphasise that the South Karelia and Hangzhou case studies present actual mixed MSW management systems. Thus, they include case-specific, direct data derived from different operators in the case areas. In previous literature, perturbation analysis has been conducted in hypothetical MSW LCA studies (Clavreul et al., 2012; Bisinella et al., 2016). Various parameters were tested in the South Karelia and Hangzhou case studies. Approximately 50% of them were applied in both case studies. The list of the tested parameters is presented in Supplementary material C.

3. Results and discussion

3.1. Contribution analysis

3.1.1. South Karelia

The GWPs, APs and EPs of the mixed MSW management scenarios in the South Karelia region are presented in Supplementary material D. According to the results, incineration (Scenarios 1.1, 1.2 and 1.3) is better option than landfilling (Scenario 0) in all impact categories. Heat substitution made a significant contribution to the results. In Scenario 1.3, the substituted heat is produced mainly by biofuels, whereas substituted heat is produced by natural gas in Scenarios 1.1 and 1.2. Therefore, due to the higher amount of avoided emissions resulting from substituting heat produced by natural gas, Scenarios 1.1 and 1.2 had negative GWPs. The GWP of Scenario 1.3 was instead positive due to the lower amount of avoided emissions from substituting heat produced by biofuels. On the other hand, Scenario 1.3 had the lowest AP and EP mainly due to avoided emissions resulting from heat substitution and a higher electricity production efficiency.

The collection and transportation of mixed MSW accounted for a larger proportion of the direct emissions in the AP and EP impact categories than in GWP (see Table 2 presenting the main processes' contributions to the direct and avoided emissions). Landfilling made a similar contribution to the total results in all impact categories: it accounted for the vast majority of the direct emissions. The incineration of mixed MSW accounted for a larger proportion of the direct emissions regarding GWP than the other impact categories. The treatment of boiler ash, air pollution control (APC) residues and metals generated relatively more emissions concerning AP and EP than GWP. The treatment of bottom ash and the use of chemicals in incineration made a minor contribution to the direct emissions in all impact categories. As for the avoided emissions of Scenarios 1.1–1.3, the most noteworthy difference between the scenarios is that metal substitution accounted for less emissions concerning EP compared to GWP and AP, whereas the proportion of energy (i.e. electricity, heat and process steam) substitution of the avoided emissions was similar between the impact categories, i.e. it accounted for the vast majority of the avoided emissions. Gravel substitution made only a minor contribution to the total results in all impact categories.

3.1.2. Hangzhou

The GWPs, APs and EPs of the mixed MSW management scenarios in Hangzhou are presented in Supplementary material E. Scenarios 0 and 1.1 had the highest GWPs, whereas Scenarios 1.1 and 1.2 had the highest APs and EPs. Scenarios 2.3 and 2.4 had the lowest emissions in all impact categories. It is noteworthy that the GWPs of the scenarios were positive. The APs and EPs were instead negative with the exception of the EPs of Scenarios 1.1 and 1.2. The negative APs and EPs resulted from electricity substitution.

The transportation of mixed MSW contributed relatively more to the total APs and EPs of the scenarios compared to the GWPs of the scenarios, as in the South Karelia case study (see Table 3 where the direct and avoided emissions of the main processes are presented). Landfilling accounted for a significantly lower proportion of the direct emissions concerning AP than EP and GWP. Incineration generated relatively more emissions regarding AP compared to the other impact categories. Bottom ash treatment also accounted for a larger proportion of the direct emissions concerning AP and EP than GWP. It is noteworthy that bottom ash treatment made a more significant contribution to the total results in the Hangzhou case study compared to the South Karelia case study. Boiler ash treatment made only a minor contribution to the direct emissions in all impact categories. The treatment of organic reject made a similar contribution to direct emissions in all impact categories. Mechanical treatment to produce RDF made a greater contribution to the direct emissions concerning AP and EP than GWP due to electricity consumption. The division of the avoided emissions was rather similar in all impact categories: electricity substitution generated most of the avoided emissions, whereas metal recycling did not yield a significant amount of avoided emissions. The contribution of electricity substitution from the combustion of LFG and the energy substitution from organic reject to the avoided emissions was noteworthy in all impact categories.

Table 2
The contributions (%) of treatment processes to the total direct and avoided emissions in the South Karelia case study.

Impact category	GWP				AP				EP			
Scenario	0	1.1	1.2	1.3	0	1.1	1.2	1.3	0	1.1	1.2	1.3
Direct emissions												
Transportation of mixed MSW	1.8	2.2	1.6	2.0	4.4	11.9	10.0	9.4	6.4	12.3	9.7	13.5
Landfill emissions	98.2	0.0	0.0	0.0	95.6	0.0	0.0	0.0	93.6	0.0	0.0	0.0
Incineration	0.0	93.3	93.8	90.8	0.0	75.4	74.4	68.1	0.0	78.2	79.1	66.7
Bottom ash treatment	0.0	0.1	0.1	0.0	0.0	0.5	0.8	0.0	0.0	0.4	0.8	0.0
Boiler ash treatment	0.0	1.2	1.2	4.4	0.0	5.4	6.0	16.9	0.0	3.0	3.1	12.8
Pretreatment of metals	0.0	1.8	1.9	1.7	0.0	6.0	7.2	5.0	0.0	4.4	4.9	5.1
Use of chemicals in incineration	0.0	1.4	1.5	1.0	0.0	0.8	1.6	0.6	0.0	1.6	2.5	1.9
Avoided emissions												
Electricity substitution	0.0	21.0	15.5	60.3	0.0	51.7	42.8	55.4	0.0	38.8	30.1	52.6
Heat substitution	0.0	68.9	28.9	27.4	0.0	38.4	18.1	40.1	0.0	59.0	26.0	46.3
Steam substitution	0.0	0.0	46.4	0.0	0.0	0.0	29.0	0.0	0.0	0.0	41.8	0.0
Metal substitution	0.0	10.0	9.1	12.3	0.0	9.7	10.0	4.5	0.0	2.0	1.9	1.1
Gravel substitution	0.0	0.1	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.2	0.2	0.0

Table 3
The average contributions (%) of treatment processes to the total direct and avoided emissions in the Hangzhou case study.

Impact category	GWP			AP			EP		
Scenario	0	1.1–1.4	2.1–2.4	0	1.1–1.4	2.1–2.4	0	1.1–1.4	2.1–2.4
Direct emissions									
Transportation of mixed MSW	1.0	1.0	1.0	18.7	9.5	8.4	12.8	9.9	9.0
Landfill emissions	65.9	57.1	55.5	1.2	0.6	0.5	47.8	33.9	31.6
Incineration	31.4	31.2	32.3	67.9	31.0	34.0	34.3	23.3	26.4
Bottom ash treatment	1.6	0.9	0.9	11.2	3.3	3.1	4.5	2.1	2.0
Boiler ash treatment	0.1	0.0	0.0	1.0	0.2	0.2	0.6	0.2	0.2
Organic reject treatment	0.0	5.0	5.3	0.0	5.3	5.2	0.0	4.8	4.9
RDF production	0.0	4.7	4.9	0.0	50.1	48.6	0.0	25.9	26.0
Avoided emissions									
Electricity substitution of incineration	76.1	61.2	73.0	76.3	64.1	75.5	76.7	65.2	76.3
Electricity substitution of LFG combustion	22.7	18.9	12.5	22.7	19.7	12.9	22.9	20.1	13.1
Energy substitution of organic reject treatment	0.0	15.0	11.0	0.0	13.3	9.6	0.0	13.7	9.9
Metal substitution	1.3	4.9	3.5	0.9	2.9	2.0	0.4	1.0	0.7

3.2. Perturbation analyses

3.2.1. South Karelia

The most important, i.e. sensitive, parameters concerning the GWP, AP and EP of landfilling in the South Karelia case study are presented in Fig. 5. As can be seen, due to the dependency of an SR on an impact category, the most important parameters vary significantly between the impact categories. When it comes to the least important parameters, certain ones stood out in all impact categories. The electricity and chemical consumption of leachate treatment had only a minor influence on the total results in all impact categories (SRs<0.01). Additionally, the NH_3 emissions of landfilling had only a minor influence on the AP and EP of landfilling (SRs<0.007). Otherwise, there were no distinct similarities between the impact categories in terms of the least important parameters.

The most important parameters concerning the GWPs, APs and EPs of the incineration scenarios (1.1–1.3) are presented in Fig. 6. Among the least important parameters, particularly those related to the transportation of auxiliary materials (i.e. other materials than waste) stood out in all impact categories (SRs<0.01). Furthermore, certain parameters regarding bottom ash treatment, metal recycling as well as the treatment of boiler ash and APC residues had only a minor influence on the total results in all impact categories. These parameters concerned the electricity consumption in the treatment of boiler ash and APC residues, the use of machinery (i.e. wheel loaders) in bottom ash treatment and the pretreatment of metals for recycling (SRs<0.01).

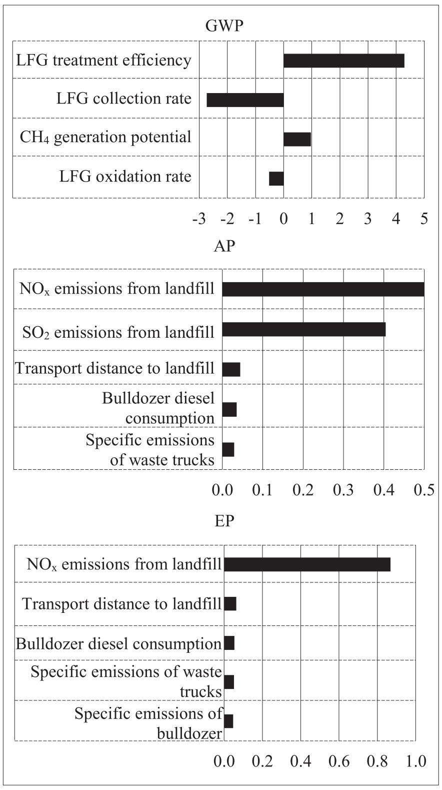


Fig. 5. The most important parameters and their SRs regarding landfilling in the South Karelia case study.

3.2.2. Hangzhou

The most important parameters regarding landfilling (Tianziling and Liugongduan landfills) in the Hangzhou case study are presented in Fig. 7. With regard to the least important parameters, the electricity consumption of leachate treatment had only a minor influence on the total results regarding all impact categories (SRs<0.02), as in the South Karelia case study. Parameters concerning the use of bulldozers in landfilling (i.e. diesel consumption and emissions generated during use) had only a minor influence on the total results in all impact categories (SRs<0.02).

The most important parameters regarding the GWPs, APs and EPs of the incineration of mixed MSW in the Hangzhou case study are presented in Fig. 8. The transportation of auxiliary materials

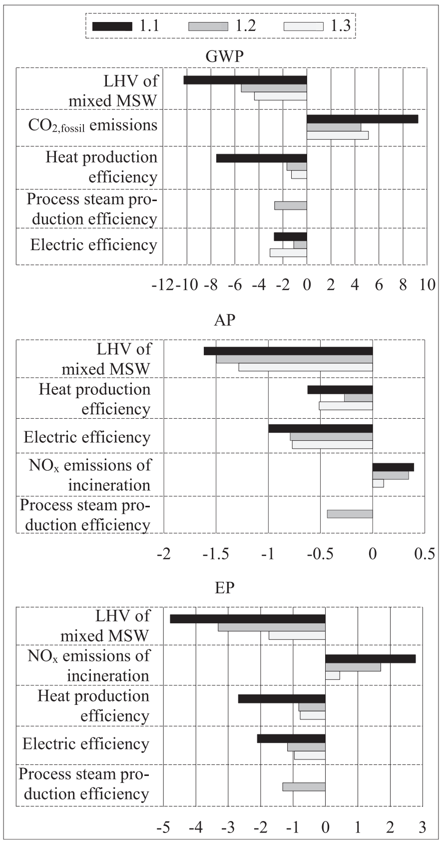


Fig. 6. The most important parameters and their SRs regarding incineration (Scenarios 1.1–1.3) in the South Karelia case study.

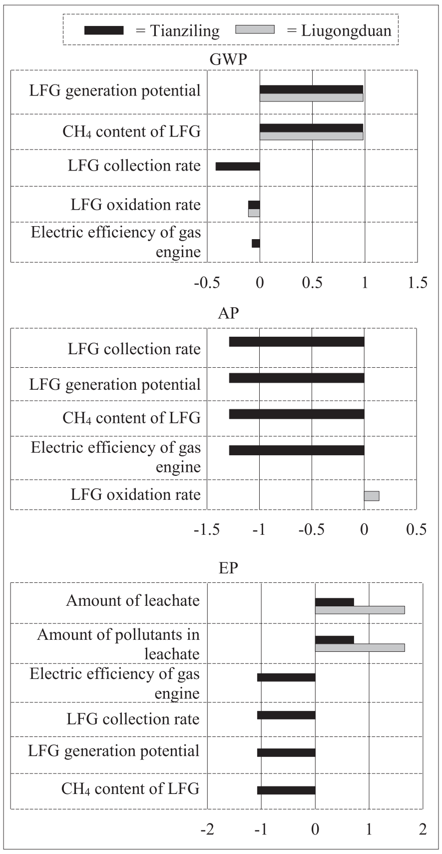


Fig. 7. The most important parameters and their SRs regarding landfilling in the Hangzhou case study.

proved to have only a minor influence on the total results in all impact categories (SRs<0.01), as in the South Karelia case study. Additionally, certain parameters concerning the treatment of boiler ash and metal recycling were among the least important parameters. These parameters concerned the share of aluminium and steel in bottom ash, the amount and water content of boiler ash, and the cement consumption of boiler ash treatment (SRs<0.02).

3.2.3. Comparison of the case studies

The case studies are compared to each other in terms of the SRs of the parameters that were applied in both studies. These parameters' mean values and coefficients of variation (CV) are presented in Table 4. The mean values and CVs were calculated from the parameters' SRs in the different scenarios of each case study, i.e. case by case. Furthermore, the Mann-Whitney *U* test on a 95% confidence level test was applied to identify significant differences among the case studies (see Supplementary material F where the results of the test are presented) (Brunner and Puri, 1996). A simple guideline when interpreting the results of the test: the smaller the *p*-value is, the more significant the difference among the case studies is. The range of *p*-values is 0–1 (*p*-value less than 0.05 indicates a significant difference). The analysis was carried out with SPSS Statistics Version 23.

The parameters concerning landfilling could not be analysed by the Mann-Whitney *U* test due to lack of data: the South Karelia case study included one landfilling scenario and the Hangzhou case study included two. Therefore, these parameters' SRs among the case studies are compared solely based on the data presented in Table 4. As can be seen, among the parameters related to landfilling, the collection rate of LFG was considerably more sensitive in the South Karelia case study than in the Hangzhou case study with regard to GWP. The collection rate of LFG was 75% in the South Karelia case study, whereas it was 25% in the Hangzhou case study. Based on this, the higher the collection rate is, the more sensitive it is. The collection rate of LFG also had an influence on the total AP and EP in the Hangzhou case study due to the electricity production from LFG. The oxidation rate of LFG was also more sensitive in the

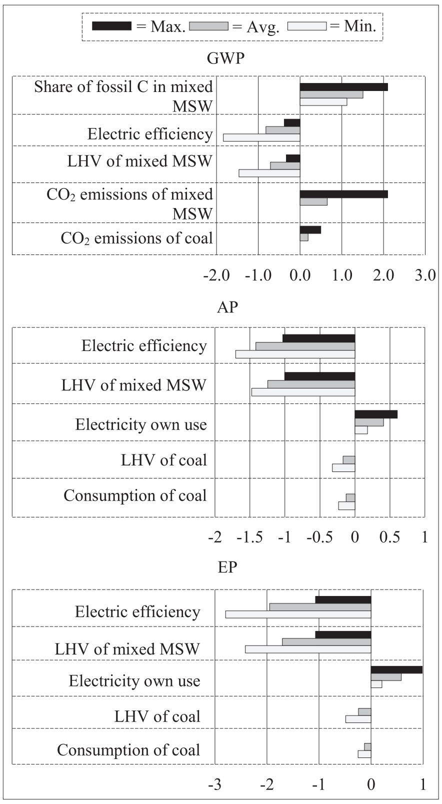


Fig. 8. The most important parameters and their SRs regarding incineration in the Hangzhou case study.

South Karelia case study in terms of GWP. The oxidation rate was approximately four times higher in the South Karelia case study, which indicates the same phenomenon as the collection rate of LFG. The LFG generation potential was equally sensitive concerning the GWPs of both case studies. In the Hangzhou case study, it also had an effect on the total APs and EPs due to electricity substitution. In terms of the parameters concerning the use of bulldozers, the parameters were more sensitive with regard to GWP in the South Karelia case study than in the Hangzhou case study. However, such distinct differences were not identified in the other two impact categories. The influence of parameters concerning the generation of leachate and the concentration of pollutants in it differed between the case studies due to the higher concentration of pollutants in leachate in the Hangzhou case study.

As for the parameters concerning the incineration of mixed MSW, it can be seen that the electric efficiency of incineration was more sensitive regarding the GWP in the South Karelia study due to a higher energy production rate, which correspondingly results from the higher energy content of mixed MSW in South Karelia (the *p*-value of the Mann-Whitney *U* test was 0.071 → a rather significant difference among the case studies). The parameter was more sensitive regarding the AP in the Hangzhou case study (*p*-value 0.036 → a significant difference). In terms of EP, there was no distinct difference between the case studies (*p*-value 0.393). Concerning GWP, the LHV of mixed MSW was substantially more sensitive in the South Karelia case study (*p*-value 0.036). The parameter itself was also substantially higher in the South Karelia study. This again indicates the correlation between parameter's value and sensitivity: the higher the LHV of mixed MSW is, the more sensitive it is. There were no such distinct differences between the case studies in terms of the parameter in the AP and EP impact categories (*p*-values 0.143 and 0.250). As did the LHV, the $CO_{2,fossil}$ emissions of incineration was significantly more sensitive regarding GWP in the South Karelia case study (*p*-value 0.036): its SR was multiple times higher than in the Hangzhou case study due to the higher fossil carbon content in the mixed MSW. There were no distinct and noteworthy differences between the case studies regarding the SRs of NO_x (*p*-values 0.393 and 0.250 in the AP and EP impact categories), SO_2 (*p*-value 0.571 in the AP impact category) and HCl (*p*-value 0.114 in the AP impact category) emissions. The electricity own use in incineration was more sensitive in terms of AP in the Hangzhou case study (*p*-value 0.036). In terms of the other impact categories, the sensitivity of the parameter did not vary significantly among the case studies. The SRs of cement consumption in residue treatment did not vary substantially between the case studies (*p*-values 0.571, 0.571 and 0.786 in terms of GWP, AP and EP). The SRs of the amount of residues varied significantly between the case studies (*p*-values 0.036 in all impact categories).

Parameters concerning metal recycling had rather a minor influence on the total results in both case studies, and their SRs did not vary significantly between the studies (*p*-values 0.095 in all impact categories). However, as can be noticed in Table 4, parameters concerning metal (i.e. aluminium and steel) recycling were more sensitive in the South Karelia case study, particularly in terms of GWP. Parameters regarding transportation were similar from their sensitivity point of view in both case studies with one exception. The transportation of mixed MSW was significantly more sensitive in South Karelia case study in terms of GWP (*p*-value 0.036).

3.3. Factors influencing the LCA of the case studies

The contribution analyses of the case studies demonstrated how critical energy substitution was in the case studies. In the South Karelia case study, heat substitution had a remarkably strong influence on the total results, and it determined the order of the incineration scenarios in all impact categories. Electricity substitution also had a major influence on the total results. In the Hangzhou case study, the influence of energy recovery and substitution was not as obvious due to only electricity recovery from mixed MSW. Nevertheless, energy recovery and substitution was evidently the most critical individual process influencing the total results of both case studies. It is therefore highly recommendable to use case-specific and direct data regarding parameters concerning energy recovery and substitution (e.g. energy content of mixed MSW, energy production efficiencies, etc.).

The perturbation analyses of the case studies demonstrated that the most critical parameters concerning landfilling were directly related to LFG (i.e. generation potential, collection rate, treatment efficiency) and leachate (i.e. generation potential, concentration of pollutants), even though there were some inconsistencies between the case studies, as presented in Table 4. The LHV of the mixed MSW, $CO_{2,fossil}$ emissions of incineration, and energy production efficiencies were clearly the most critical ones of the parameters related to incineration in the case studies. Additionally, the NO_x and SO_2 emissions of incineration had a notable influence on the total

Table 4
Comparison of the SRs of the parameters that were applied in both studies.

Parameter	SRs' mean values and CVs (in brackets)					
	GWP		AP		EP	
	South Karelia	Hangzhou	South Karelia	Hangzhou	South Karelia	Hangzhou
Landfilling						
LFG collection rate	-2.73 (-)	-0.42 (-)	-	-1.29 (-)	-	-1.07 (-)
LFG oxidation rate	-0.52 (-)	-0.11 (0.1%)	-	0.14 (-)	-	0.12 (-)
LFG generation potential	0.97 (-)	0.98 (0.1%)	-	-1.29 (-)	-	-1.07 (-)
Bulldozer diesel consumption	0.01 (-)	0.001 (18%)	0.04 (-)	0.01 (40%)	0.05 (-)	0.01 (40%)
Bulldozer emissions	0.01 (-)	0.001 (18%)	0.03 (-)	0.01 (40%)	0.05 (-)	0.01 (40%)
Amount of leachate	-	-	-	-	0.001 (-)	1.19 (40%)
Amount of pollutants in leachate	-	-	-	-	0.001 (-)	1.19 (40%)
Electricity consumption of leachate treatment	0.003 (-)	0.001 (18%)	0.01 (-)	0.02 (40%)	0.005 (-)	0.01 (40%)
Incineration						
Electric efficiency of incineration	-2.30 (37%)	-0.82 (64%)	-0.85 (12%)	-1.42 (17%)	-1.41 (35%)	-1.95 (32%)
LHV of mixed MSW	-6.70 (38%)	-0.71 (58%)	-1.46 (9%)	-1.25 (16%)	-3.28 (38%)	-1.70 (30%)
$CO_{2,fossil}$ emissions of incineration	6.29 (33%)	2.16 (25%)	-	-	-	-
NO_x emissions of incineration	-	-	0.28 (45%)	0.20 (58%)	1.64 (58%)	0.63 (65%)
SO_2 emissions of incineration	-	-	0.05 (50%)	0.09 (57%)	-	-
HCl emissions of incineration	-	-	0.01 (44%)	0.01 (60%)	-	-
Electricity own use in incineration	0.30 (32%)	0.19 (23%)	0.12 (35%)	0.41 (43%)	0.21 (49%)	0.58 (53%)
Cement consumption for residue treatment	0.12 (57%)	0.07 (50%)	0.03 (21%)	0.05 (62%)	0.07 (18%)	0.09 (69%)
Amount of residues (i.e. flue gas residues)	0.14 (56%)	0.01 (61%)	0.04 (20%)	0.01 (72%)	0.09 (20%)	0.02 (81%)
Metal recycling						
Proportion of aluminium in bottom ash	-0.41 (35%)	-0.01 (23%)	-0.12 (2%)	-0.02 (40%)	-0.02 (20%)	-0.01 (53%)
Proportion of steel in bottom ash	-0.26 (35%)	-0.002 (25%)	0.01 (0.2%)	-0.001 (40%)	0.08 (19%)	-0.0003 (52%)
Transportation						
Transportation distance of mixed MSW	0.10 (69%)	0.02 (28%)	0.04 (34%)	0.03 (47%)	0.17 (72%)	0.10 (59%)
Transportation distance of residues	0.002 (85%)	0.002 (70%)	0.001 (73%)	0.005 (80%)	0.004 (80%)	0.01 (86%)
Transportation distance of cement	0.001 (72%)	0.001 (63%)	0.0003 (44%)	0.002 (74%)	0.001 (39%)	0.01 (81%)

results. In terms of metal recycling, the most important parameters were related to the recoverable amount of metal in mixed MSW, or rather in bottom ash. Of the transportation-related parameters, parameters concerning the transportation of mixed MSW had the strongest influence on the total results.

As presented in Table 4 and previously discussed, parameters with only a minor influence on the total results were identified in all the main mixed MSW management phases: transportation, landfilling and incineration. In terms of the least important parameters concerning landfilling, the use of a bulldozer in landfilling,

Table 5
Possibilities to simplify (i.e. apply default or generic data instead of case-specific, direct data) the LCA of the South Karelia and Hangzhou case studies.

	Simplification possibility
Landfilling	– Electricity and chemical consumption in leachate treatment – The use of machinery in landfilling (i.e. the diesel consumption of a bulldozer)
Incineration	– The treatment of boiler ash and APC residues: electricity consumption, the use of machinery (i.e. the diesel consumption of a wheel loader) – Bottom ash treatment and metal recycling: the use of machinery in bottom ash treatment, the pretreatment of metals for recycling (i.e. the use of machinery and electricity consumption)
Transportation	– The transportation of auxiliary materials, such as chemicals, APC residues, boiler and bottom ash (i.e. diesel consumption and transportation distances)

and the electricity and chemical consumption in leachate treatment had a fairly minor influence on the total results. Certain parameters related to bottom and boiler ash treatment were the least important ones related to incineration. For instance, the electricity consumption during the treatment did not have a notable influence on the total results. Additionally, the transportation of auxiliary materials had a rather minor influence on the total results, regardless of the impact categories.

The perturbation analyses of the case studies also demonstrated how the magnitude of an SR is dependent on the value of a parameter. With regard to certain parameters (e.g. the LHV of mixed MSW and the collection rate of LFG), the correlation between the magnitude of an SR and the value of a parameter was the following: the higher the value of a given parameter is, the more sensitive the parameter is.

A possibility to simplify the LCA of the case studies is to apply default or generic data instead of direct, case-specific data. Default data should be applied with caution, i.e. to parameters which have only a minor influence on the total results. The exclusion of certain processes (e.g. the transportation of auxiliary materials) from the assessment is also a possibility to simplify the LCA of the case studies. However, it requires particular caution. Possibilities to simplify the LCA of the South Karelia and the Hangzhou case studies are presented in Table 5.

The possibilities to simplify the LCA of the South Karelia and Hangzhou case studies were identified and discussed in this study. However, one should identify the limitations of the study. First of all, only three impact categories were assessed in the case studies. Therefore, the simplification possibilities concern only the GWP, AP and EP impact categories. Secondly, only two different case studies were compared in this study. The findings of the study apply solely to the case studies, and more case studies are required in order to draw more extensive and general conclusions. Thirdly, it should be noticed that the study focused on individual parameters and their sensitivity, and the identified simplification possibilities concerned merely them. In other words, this study did not concern process, modelling or scenario uncertainties, although they can also have a strong influence on the overall uncertainty of LCA studies (Clavreul et al., 2012). This is due to the fact that parameter sensitivity can be computationally quantified (i.e. by determining SRs), and thus utilised in the comparison of different case studies. Fourthly, it is worth noting that the differences between the LCAs of the case studies (e.g. system boundaries, modelling principles, etc.) can influence the magnitude of SRs. Therefore, the most and least important parameters were identified case by case based on the ranking of the SRs rather than focusing merely on the magnitude of SRs. For instance, although the SR of a parameter would vary notably (e.g. 1.5 and 2.5) between the case studies, if the parameter were among the most important parameters in both studies based on the case-specific ranking of SRs, the parameter would be identified as an important one, and vice versa. Regardless of the limitations of the study, the study introduces a novel perspective to the LCA of MSW management systems. When default or generic data is enough instead of direct, case-specific data? It is not easy to draw the line between them. However, the study presents examples on how the particular case studies could be simplified in this manner. It should be acknowledged that the simplification possibilities presented in this study are rather conservative due to the above-mentioned limitations. Therefore, they may well be applicable in other case studies, too.

4. Conclusions

Mixed MSW management systems in the South Karelia region, Finland and the city of Hangzhou in China were compared in this study in order to find out the similarities and differences between the case studies in terms of the influence of different factors on the total results of the LCA studies. The comparison of the case studies focused particularly on the influence of various input parameters on the total results, i.e. the GWPs, APs and EPs of the systems. After the comparison, possibilities to simplify and thus diminish the workload of the case studies were discussed and introduced in the study.

Even though there were differences in the influence of individual parameters on the total results of the case studies, certain factors stood out. Energy recovery and substitution were the most critical individual processes influencing the results of the case studies. In terms of individual input parameters, those directly related to the generation and collection of LFG, the energy and fossil carbon content of mixed MSW, energy production efficiencies, as well as the NO_x and SO_2 emissions of incineration had a significant influence on the total results in both case studies. Therefore, direct and case-specific data should be particularly applied to these parameters. Parameters related to the use of machinery in landfilling, the electricity and chemical consumption in leachate treatment and the transportation of auxiliary materials were not that crucial regarding the total results of the case studies. Additionally, certain parameters related to boiler and bottom ash treatment had a minor influence on the total results. To diminish the workload of the LCA of the case studies, default or generic data could be applied to these parameters instead of case-specific, direct data. It is worth noting that the findings of the study apply only to these particular case studies. Therefore, to draw more general conclusions, further research on the subject is required.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2017.04.023.

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Publication III

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Steps towards more environmentally sustainable municipal solid waste management – A life cycle assessment study of São Paulo, Brazil

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Steps towards more environmentally sustainable municipal solid waste management – A life cycle assessment study of São Paulo, Brazil

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ABSTRACT

Landfill disposal has thus far been the predominant treatment method for municipal solid waste (MSW) throughout Brazil, including São Paulo city. Environmentally sustainable development of MSW management in São Paulo necessitates a stepwise reduction of landfilling. However, ever increasing MSW generation poses the challenge of managing increasing MSW volumes while simultaneously modernizing the MSW management system. In this study, the environmental impacts of the current MSW management system and future alternatives in the city were assessed by means of life cycle assessment (LCA) to determine a pathway towards more environmentally sustainable MSW management. The assessed impact categories were global warming, acidification and eutrophication potentials. Potential future alternatives included the stepwise reduction of landfilling by the introduction of composting, anaerobic digestion and mechanical-biological treatment (MBT). The results of the study indicated that the environmental impacts of MSW management in São Paulo can be most effectively diminished by anaerobic digestion of source separated organic waste and MBT of MSW, on condition that the produced refuse-derived fuel (RDF) is utilized in cement production as a substitute for coal. The other utilization option for RDF, incineration, would increase the environmental impacts of MSW management due to the low amount of avoided emissions resulting from electricity substitution since average electricity production in Brazil is dominated by hydropower. Sensitivity analyses indicated, however, that the environmental impacts of incineration might decrease with different modeling assumptions, e.g. the modeling assumption regarding the kind of electricity production substituted by electricity production from MSW. Nevertheless, the main findings of the study remained the same and they are in line with the previous literature.

1. Introduction

Ever greater generation of municipal solid waste (MSW) is becoming an increasingly pressing issue globally, particularly in emerging economies and developing countries, where the local infrastructure and MSW management systems cannot always keep up with the larger MSW volumes resulting from rapid population and economic growth as well as increased urbanization (Guerrero et al., 2013). The World Bank (2012) has forecast that global MSW generation will double by 2025 (2012 as a reference year). Emerging economies, in particular, China, India and Brazil, will play a crucial role in addressing this global issue. In volume, China is the world's largest generator of MSW, and India and Brazil are third and fourth largest, respectively (Waste Atlas, 2017).

Brazil generates approximately 63 million tonnes of MSW annually (Waste Atlas, 2017). Annual MSW generation per capita is approximately 380 kg, which is lower than typically found in upper middle income countries (approximately 440 kg according to World Bank (2012)). Taking into account the strong correlation between MSW generation and income level, MSW generation per capita can be expected to increase over the following years, should Brazil, as forecast, recover from the current recession (World Bank, 2017a). Despite uncertainty surrounding the economic prospects of the country, total MSW generation will however continue to increase due to population growth (World Bank, 2012). Statistics for the national economy for 2013 to 2016 show that while gross national income (GNI) per capita decreased by 30.6%, the total population increased by 2.6%. Although population growth has

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decelerated slightly, due to the poor economic situation, the growth curve has nevertheless remained positive (World Bank, 2017b). Brazil will thus face the tough challenge of managing increasing MSW generation while simultaneously developing its MSW management systems in a more sustainable direction.

São Paulo is the most populous city in Brazil and the case area of this study (see Fig. 1). The city is the capital of São Paulo state, which is the world's largest MSW generator at a regional level, generating approximately 20 million tonnes of MSW annually, of which 4.7 million tonnes are generated in São Paulo city (CCAC, 2015a; Waste Atlas, 2017). It has been estimated that 97.8% of MSW generated in São Paulo is under the formal MSW management system. In addition to the formal MSW management system, individual pickers and picker organizations collect recyclables informally. Formally collected MSW is disposed of in the city's two sanitary landfills (i.e. landfills with landfill gas (LFG) and leachate collection systems). In addition, São Paulo has two mechanical sorting plants for separately collected recyclables such as plastic, paper, cardboard, metal and glass. Formally collected recyclables constitute, however, only a minor proportion of total MSW generated (approximately 1%, 50 000 tonnes/a) (CCAC, 2015a; CCAC, 2015b).

The future strategy for MSW management in São Paulo is described in PGIRS (Plano de Gestão Integrada de Residuous Solidos), the MSW management plan of the city, launched in 2014 (Prefeitura de São Paulo, 2014). One of the main priorities of PGIRS is reduction of the volume of MSW disposed of in landfills. Organic waste management can play an important role in achieving this objective, since approximately half of MSW is organic waste. Consequently, source separation and separate treatment of organic waste would efficiently reduce the volume of MSW landfilled, and thus the environmental impacts. Both composting (including home composting) and anaerobic digestion (AD) are proposed in PGIRS as potential treatment methods for organic waste. Some small-scale initiatives promoting home composting already exist (Composta São Paulo, 2017) but no composting or AD plants are currently operational in São Paulo, based on information from 2016, when the data for this study was collected. In addition to the above-mentioned treatment methods, mechanical-biological treatment (MBT) is proposed in PGIRS as a potential treatment method (CCAC, 2015a; Prefeitura de São Paulo, 2014).

São Paulo city
- Population: 11-12 million, depending on the source
- Surface area: ~1500 km^2
- MSW generation: ~400 kg/capita/year

Fig. 1. Background information about São Paulo.

Life cycle assessment (LCA) is a method for estimating the potential environmental impacts of products or systems (EN ISO 14040, 2006; EN ISO 14044, 2006). LCA has been utilized in the field of MSW management since the 1990s, and it is currently a widely used technique to assess the environmental impacts of MSW management systems (Laurent et al., 2014). LCA enables comparison of different MSW management strategies and treatment methods in terms of their environmental impacts, which makes it a useful tool for decision- and policy-making (e.g. Karmperis et al., 2013; Turner et al., 2016).

A vast number of studies have been published that investigate different aspects of MSW management in different parts of the world – and Brazil is no exception. For example, the electricity production potential of MSW in Brazil was the focus of Leme et al. (2014), Lino and Ismail (2011) and Mambeli Barros et al. (2014). LCA studies focusing on comparison of alternative MSW treatment methods in a given case area are also common. For instance, Goulart Coelho and Lange (2016) and Bernstad Saraiva et al. (2017) recently assessed the environmental impacts of MSW management alternatives in Rio de Janeiro by means of LCA. LCA studies of MSW management in São Paulo, the case area of this study, have also been published. Mendes et al. (2003) assessed the environmental impacts (namely global warming, acidification and nutrient enrichment) of organic waste treatment in São Paulo using LCA. They compared three different treatment methods for organic waste: landfilling, composting and AD. Both composting and AD had lower environmental impacts than landfilling, with one exception: composting had the highest acidification potential of the three treatment methods. AD had lower environmental impacts than composting (conventional composting without biofiltration).

In another study, i.e. Mendes et al. (2004), the same authors used LCA to compare the environmental impacts (for the same impact categories as above) of landfilling and MSW incineration for São Paulo. It was found that landfilling had higher environmental impacts than incineration. However, the differences were not great due to the structure of the electricity production sector in Brazil, where a vast majority of electricity is produced by hydropower (94% in Mendes et al. (2004)). Therefore, electricity production from MSW did not yield a significant amount of avoided emissions. More recently, Soares and Martins (2017) conducted a gate-to-grave LCA for 1 tonne of MSW received in CTVA Caieiras landfill, one of the sanitary landfills in São Paulo. In the study, the environmental impacts of various waste-to-energy options - LFG combustion with energy recovery, MBT combined with AD, and incineration – were evaluated. It was found that MBT combined with AD had the lowest environmental impacts of the scenarios assessed in the study. Being a site-specific gate-to-grave LCA, the study did not cover the entire life cycle of MSW, i.e. from MSW generation to its final treatment or disposal. As evident from the above examples, previous LCA studies of the MSW management system of São Paulo have focused more on specific treatment methods and their environmental impacts, rather than assessing comprehensively the environmental impacts of the MSW management system as a whole consisting of different treatment options and methods for different MSW flows (e.g. organic and residual fraction of MSW).

This study uses LCA to assess the environmental impacts of different management alternatives for MSW in São Paulo in order to determine a pathway towards more environmentally sustainable MSW management in the city. The study takes into account the entire life cycle of MSW: from MSW generation to its final treatment or disposal. Both organic and residual fractions of MSW are assessed in the study. The scenarios the study are based on the development proposals and objectives of PGIRS, and they present potential improvement steps for the system. The research questions of the study are the following:

- What are the environmental impacts of MSW management in São Paulo?
- In which direction should the system be developed in order to diminish the environmental impacts of MSW management in the city?

2. Materials and methods

2.1. MSW management in São Paulo

Household waste, street cleaning waste as well as waste from markets and commercial activities generating less than 50 kg per day are regarded as MSW in São Paulo. Total MSW generation in São Paulo is approximately 4.7 million tonnes/a (CCAC, 2015a). The vast majority of MSW in São Paulo is mixed MSW from households; in 2015, 3.8 million tonnes of mixed MSW was collected and treated (AMLURB, 2016). Mixed MSW refers to the remaining part of MSW after the source separation of recyclables. This study focuses on this particular MSW flow, i.e. formally collected and treated mixed MSW from households, since formal information is available and such waste constitutes the majority of total MSW generated (henceforth in this study MSW refers solely to mixed MSW from households due to above-mentioned reasons). Formally collected and treated recyclables are not taken into account in the study since they compose only a minor proportion of the total MSW generation, and they are already treated in an environmentally sustainable manner, on the presumption that the collected materials are truly reclaimed. Informally collected recyclables are not assessed in the study since no formal information was available.

The MSW management authority of São Paulo, AMLURB, has contracted out MSW management activities to two private companies: Loga and Ecourbis (CCAC, 2015a). The division between the companies is geographical. Loga is responsible for management of MSW generated in the northern and western parts of São Paulo, whereas Ecourbis is in charge of the MSW generated in the southern and eastern of the city (AMLURB, 2016). There are two landfills in São Paulo: the Central de Tratamento Leste (CTL) landfill (Ecourbis) and the Central de Tratamento e Valorização Ambiental (CTVA) Caieiras landfill (Loga). MSW is transported either directly or via transfer stations to the landfills. There are three transfer stations in the city; Ecourbis owns and operates two of them (Santo Amaro and Vergueiro), and Loga one (Ponte Pequena). Both companies have their own collection and transportation fleets. There are two mechanical sorting plants in the city. However, only a minor proportion of MSW is treated in these plants, and the vast majority of MSW is disposed of in landfills, as mentioned earlier (CCAC, 2015a). Different MSW streams and the average composition of mixed MSW in São Paulo are presented in Fig. 2. As can be seen, organic waste predominates, forming 49% of the composition of mixed MSW.

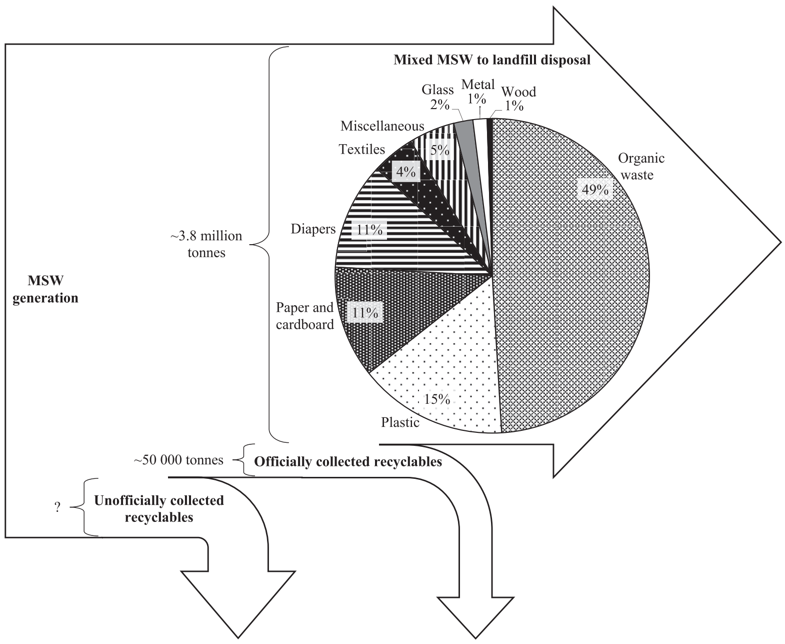


Fig. 2. Different MSW streams and the composition of mixed MSW in São Paulo (AMLURB, 2016; CCAC, 2015b).

2.2. Life cycle assessment

The LCA was conducted in accordance with the ISO standards 14040 (2006) and 14044 (2006). The impact categories assessed in the study were global warming potential (GWP) for a 100 year time span, acidification potential (AP) and eutrophication potential (EP), since the required life cycle inventory (LCI) data was available (see Section 2.2.2 and Supplementary material A-D for further information about the LCI data). Sufficiently comprehensive LCI data about other possibly relevant impact categories (e.g. human toxicity potential) was not available, and they were therefore excluded from the study. The modeling for the study was done with GaBi LCA modeling software (version 7) (Thinkstep, 2017), and the CML 2001 (April 2015) was used for impact assessment. The functional unit is the treatment of formally collected mixed MSW in São Paulo in one year. 2015 was selected as the reference year: 3.8 million tonnes of MSW were formally collected and treated in that year (AMLURB, 2016). The study takes into account the entire life cycle of MSW in São Paulo, i.e. generation, collection, treatment and the final disposal of waste. The relevant unit processes regarding the different life cycle phases are included in the system boundaries of the study (see Section 2.2.2 for further information). The study has a consequential approach, i.e. in addition to the direct emissions resulting from the unit processes, the study takes into account avoided emissions resulting from energy and material substitution (i.e. energy recovery from MSW and recycling). The context situation of the study was considered as a micro-level decision support, i.e. the study was assumed to have no large-scale consequences on the background system (e.g. the national electricity production market) (EC-JRC, 2010). Therefore, the produced electricity from MSW was assumed to substitute average electricity production in Brazil. Hydro (75.2%), natural gas (8.5%), biomass (6.3%), heavy fuel oil (3.5%) and nuclear (2.9%) are the main energy sources for electricity production in Brazil (Thinkstep, 2016).

2.2.1. Scenarios

The scenarios considered in the study represent potential treatment methods for MSW in São Paulo and are stepwise improvements towards a more environmentally sustainable MSW management system in the city. The strategic MSW management development plans of the city are taken into account in the scenarios: the treatment methods employed in the scenarios are proposed in PGIRS as potential treatment methods for MSW in São Paulo − composting, AD and MBT.

There are five main scenarios in the study (Scenarios 0–4). Additionally, there are sub-scenarios in Scenarios 2, 3 and 4. The sub-scenarios indicate the treatment method for separately collected organic waste: composting (X.1) or anaerobic digestion (X.2). In the baseline scenario (Scenario 0), which is the status quo, 100% of collected MSW is disposed of in landfills. Scenario 1 is a combination of home composting and landfilling: 5% of organic waste (i.e. 2.5% of the total MSW) is home composted and the residual MSW (i.e. the remaining MSW after the separation of organic waste) is disposed of in landfills. A 5% home composting rate was chosen since the objective regarding home composting in PGIRS is highly ambitious: 33% home composting rate of organic waste by 2033 (Prefeitura de São Paulo, 2014). However, in a shorter time span the objective is not realistic since approximately 5% of organic waste is composted in Brazil at present (CCAC, 2015a). Therefore, we employed a more realistic rate for home composting in the study.

In Scenario 2, the home composting of organic waste is accompanied by the separate collection and treatment of organic waste: 20% of organic waste (i.e. 9.8% of the total MSW) is either composted (Scenario 2.1) or anaerobically digested (Scenario 2.2) and the residual MSW is landfilled. We selected a 20% separate collection rate based on the PGIRS strategy objective of establishing new organic waste treatment facilities. The aim is that the total treatment capacity of the facilities would be 19% of total MSW generation, and 40% of organic waste generation by 2023 (CCAC, 2015a). In view of the current situation, no operating composting or AD plants, a more realistic mid-term goal of a 20% separate collection and treatment rate for organic waste was employed in the study.

In Scenarios 3 and 4, MBT of the residual MSW is included in the assessment: 20% of the residual MSW (i.e. 17.6% of the total MSW) is treated in MBT plants, while the rest of the MSW is disposed of in landfills. PGIRS contains no specific objectives regarding the MBT of MSW. Therefore, a realistic mid-term MBT capacity was employed, as with the organic waste treatment plants. Scenarios 3 and 4 differ in the utilization of the refuse-derived fuel (RDF) produced. The RDF is incinerated in waste-to-energy plants in Scenario 3, whereas in Scenario 4, it is utilized in cement production as a substitute for coal, which is typically used as the primary fuel in cement kilns. The treatment method for the generated organic reject is either composting or AD, depending on the sub-scenario (Scenarios 3.1 and 4.1 → composting; Scenarios 3.2 and 4.2 → AD). The scenarios are hierarchical (see Fig. 3), i.e. the improvement steps taken in previous scenarios are also included in the following scenarios. The MSW mass flows of the scenarios are presented in Table 1.

2.2.2. System boundaries and calculation principles

The system boundaries of the study (see Fig. 4) include direct emissions from transportation and treatment of MSW (including electricity and diesel consumption) as well as avoided emissions resulting from material and energy substitution. The entire MSW management system of São Paulo was assessed in the study, i.e. the study takes into account the MSW management operations of both Ecourbis and Loga.

São Paulo city has 32 districts, of which 19 are in the operation area of Ecourbis and 13 in the operation area of Loga (AMLURB, 2016). The collection and transportation of MSW was modeled for each district individually (see supplementary material A). The calculation principles for transportation distances are presented in Fig. 5. The following assumptions were made. The payload capacity of a truck was either 11 or 25 tonnes depending on whether MSW was directly transported to a MSW management site or via a transfer station. Trucks carried a full payload to a MSW treatment site and returned back empty. Organic waste was directly transported to a MSW treatment site due its high moisture content. All MSW treatment plants (i.e. composting, AD and MBT plants) were assumed to be located at the current landfill sites, CTL and CTVA Caieiras. Using the above-mentioned assumptions, the emissions from transportation were calculated with the GaBi software (i.e. the unit emissions of trucks were derived from GaBi's database).

Landfilling of the MSW was modeled mainly based on information received during visits to the CTL and CTVA Caieiras landfills (site-specific data). The modeling was complemented with literature data if necessary. The methane (CH_4) generation potential (L_0) of MSW is one of the most critical parameters in terms of the environmental impacts of landfilling, particularly GWP (Liikanen et al., 2017). CH_4 generation potential was calculated using the following equation:

$$L_0 = DOC \times DOC_f \times MCF \times F \times \frac{16}{12} \tag{1}$$

where $L_0 = CH_4$ generation potential $[Gg_{CH4}/Gg_{MSW}]$

$DOC =$ Fraction of degradable organic carbon $[Gg_C/Gg_{MSW}]$

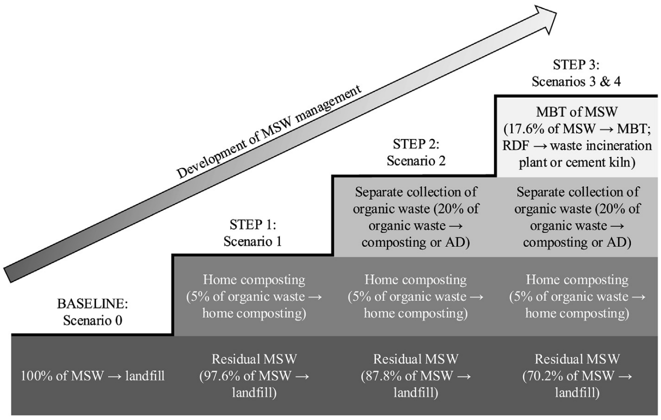


Fig. 3. Scenarios of the study.

Table 1
Mass flows of MSW in the scenarios.

Scenario	Landfill [kt]	Treatment of organic waste [kt]			MBT [kt]		∑
		Home composting	Composting plant	AD	Incineration	Cement kiln	
0	3800	–	–	–	–	–	3800
1	3707	93	–	–	–	–	3800
2.1	3335	93	372	–	–	–	3800
2.2	3335	93	–	372	–	–	3800
3.1	2668	93	372	–	667	–	3800
3.2	2668	93	–	372	667	–	3800
4.1	2668	93	372	–	–	667	3800
4.2	2668	93	–	372	–	667	3800

DOC_f = Decomposable fraction of DOC [%]
MCF = CH_4 correction fraction [-]
F = Share of CH_4 in landfill gas (LFG) [%]
16/12 = The molecular weight ratio between CH_4 and C [-] (IPCC, 2006).

L_0 is the total amount of CH_4 generated in the decomposition of MSW. Since CH_4 generation continues for decades after the disposal of MSW (IPCC, 2006), L_0 was employed in modeling in order to take into account the entire life cycle of MSW. Data used in the modeling of landfilling, including the above-described L_0, is presented in supplementary material B.

The home composting of organic waste was modeled based on literature data (Andersen et al., 2012; Boldrin et al., 2009). It was assumed that the generated compost substituted multinutrient fertilizers in domestic use. The substituted multinutrient fertilizer was a NPK fertilizer containing nitrogen, phosphorus and potassium at the same proportion (each 15% of the total content). Data used to model the environmental impacts of home composting is presented in supplementary material C.

Since there were no operational composting or AD plants in São Paulo in 2016, when the data for the study was collected, both treatment processes were modeled based on literature data. The studies of Boldrin et al. (2009), Brown et al. (2008) and Pagans et al. (2006) were utilized in modeling of the composting process (see supplementary material C for further information). Windrow composting was assumed for the composting technology. It was considered as a potential composting technology in the case area since it is a rather simple and consequently low-cost composting technology, and it has been applied in Brazil (Santos et al., 2017). The generated compost was assumed to substitute similar multinutrient fertilizer as in the home composting process. The AD process was modeled based on the studies of Angelidaki et al. (2006), Berglund and Börjesson (2006), Havukainen et al. (2017), Møller et al. (2002) and Nielsen et al. (2010) (see supplementary material C for further information). The generated digestate was assumed to be windrow composted. As in the composting process, the generated compost from pile composting was assumed to substitute conventional NPK fertilizer.

MBT of the MSW was also modeled based on literature data due to a lack of site-specific data - there were no operating MBT plants in São Paulo in 2016. The studies of Damgaard et al. (2009), Leme et al. (2014) and Nasrullah et al. (2015) were employed in the modeling of MBT (see supplementary material D for further information). Compost generated from mechanically separated organic reject (approximately 28% of input MSW) was assumed to be used as a landfill cover material instead of fertilizer due to the lower compost quality. Mechanically separated organic reject contains more harmful substances (e.g. heavy metals) and other unwanted materials (e.g. plastic) than source separated organic waste (Di Lonardo et al., 2012), which may restrict the utilization of compost in fertilizing, soil improvement and landscaping purposes. The treatment of organic reject (i.e. composting or AD) was

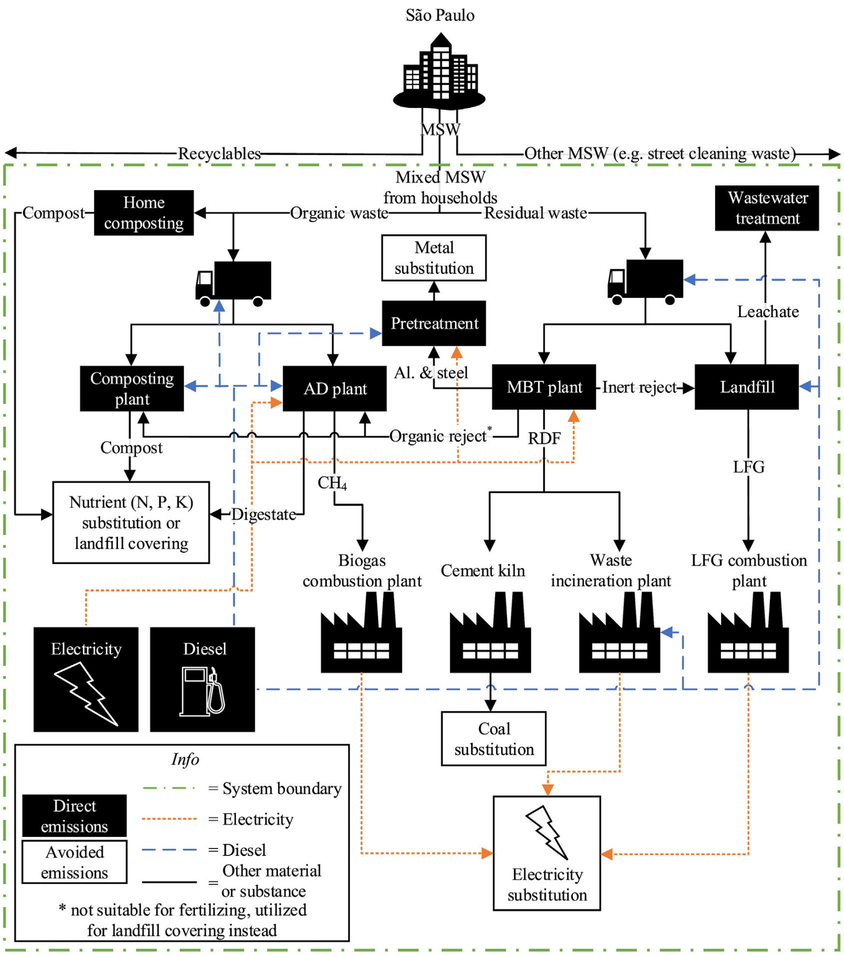


Fig. 4. System boundaries of the study.

otherwise modeled similarly as described above.

Incineration of the RDF was modeled based on the studies of Astrup et al. (2009), Birgisdóttir et al. (2006), Consonni et al. (2005), Havukainen et al. (2017), Hupponen et al. (2015), Leme et al. (2014) and Mendes et al. (2004) (see supplementary material D for further information). The utilization of RDF in cement production as a substitute fuel for coal was calculated based on the energy content (lower heating value, LHV) of RDF. Avoided acquisition (i.e. mining, processing and transportation) and combustion of hard coal in cement production were taken into account in the study, whereas other emissions from cement production were excluded from the assessment.

3. Results and discussion

3.1. Contribution analysis

In contribution analysis, the total result is decomposed into individual process contributions (Clavreul et al., 2012). The GWPs of the scenarios are presented in this manner in Fig. 6. Scenarios 4.2 and 4.1 clearly had the lowest GWPs, whereas the GWPs of the Scenarios 3.1 and 0 were the highest. The GWPs of the other

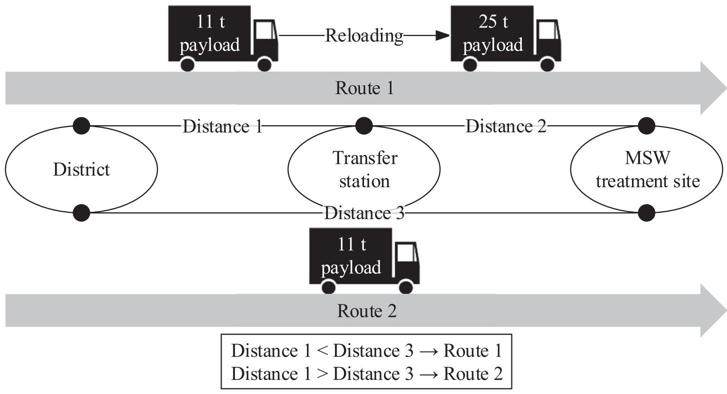


Fig. 5. Calculation principles for transportation distances.

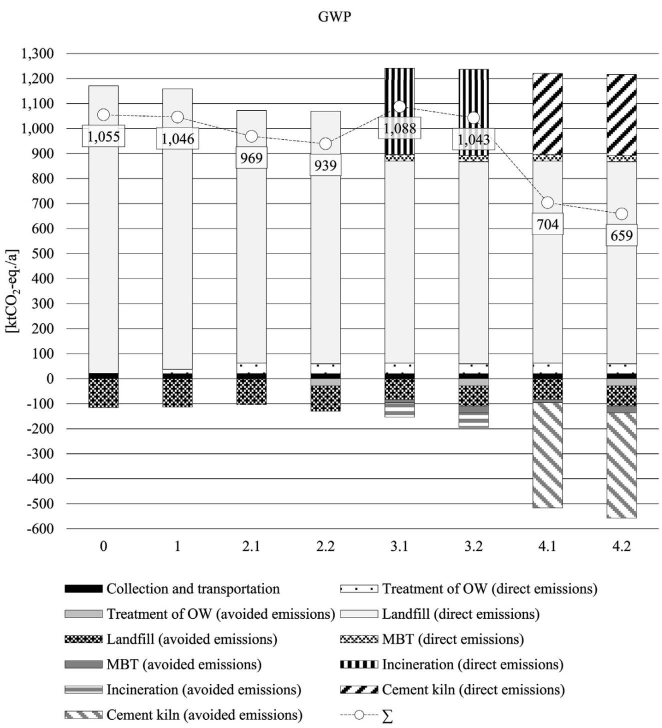


Fig. 6. GWPs of the studied scenarios.

scenarios (Scenarios 1, 2.1, 2.2 and 3.2) were lower compared to the baseline scenario (Scenario 0) but not significantly. It is noteworthy that the MBT of MSW and incineration of RDF (Scenario 3) did not decrease the GWP of MSW management in São Paulo due to the average electricity production in Brazil (approximately 75% hydropower) — quite the contrary. Indeed, electricity production from

RDF generated notably more emissions compared to the average electricity production in Brazil. In other words, the direct emissions of electricity production from RDF incineration outweighed the avoided emissions from electricity substitution (average electricity production in Brazil), which led to an unfavorable outcome for utilization of the RDF in waste incineration plants. Therefore, RDF should rather be utilized in cement production (Scenario 4). As regards different treatment options for separately collected organic waste, the GWP results indicate that AD (Scenarios 2.2, 3.2 and 4.2) is better option than composting (Scenarios 2.1, 3.1 and 4.1). Home composting of organic waste (Scenario 1) decreased the GWP of MSW management slightly.

The APs of the scenarios are presented in Fig. 7. Scenarios 4.2 and 4.1 had the lowest APs, while the APs of Scenarios 3.1 and 3.2 were the highest. It is noteworthy that the APs of all scenarios, except Scenarios 3.1 and 3.2, were negative, i.e. avoided emissions were greater than direct emissions, mainly due to electricity substitution. The results matched observations found in the GWP category – the MBT of MSW and incineration of RDF are not beneficial in this impact category either due to the low amount of avoided emissions resulting from electricity substitution and, respectively, the high amount of direct emissions generated in the incineration process. Combustion in a cement kiln is therefore a better utilization option for RDF than waste incineration for electricity generation in this regard, too. The AP of the baseline scenario was also negative, i.e. beneficial for the environment. The landfill processes inflicted considerably less direct emissions (e.g. the use of bulldozers) compared to the avoided emissions achieved in electricity production from LFG. As in the GWP impact category, it was more beneficial for the separately collected organic waste to be anaerobically digested rather than composted. The composting of organic waste, including home composting, increased the AP of MSW management compared to Scenario 0.

The EPs of the scenarios are presented in Fig. 8. As can be seen, the results were consistent with the previous main findings in the GWP and AP impact categories: AD is a better treatment option for

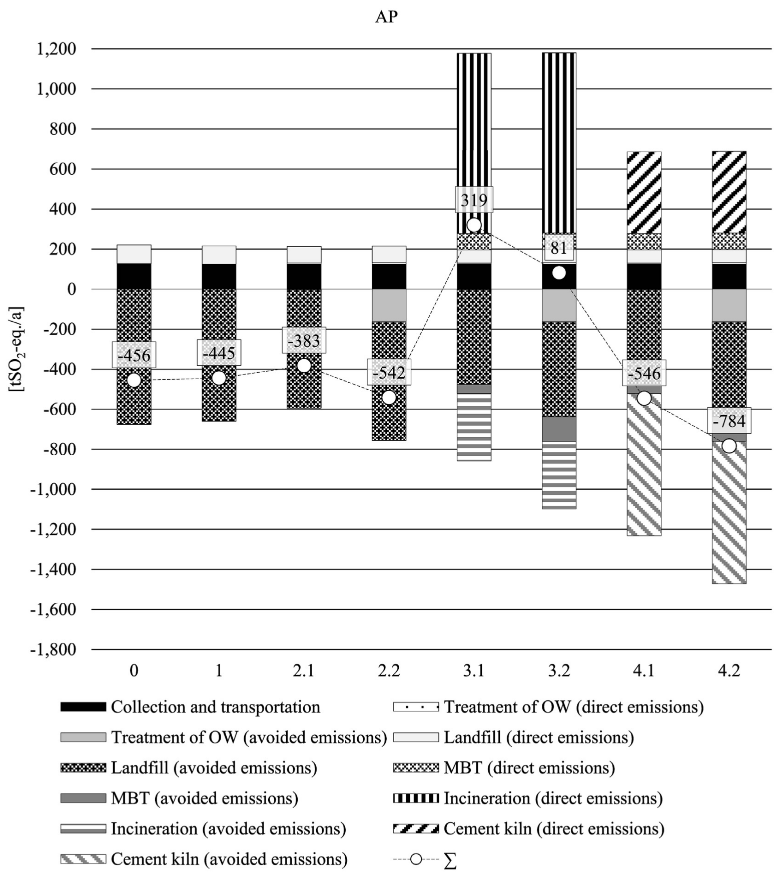


Fig. 7. APs of the studied scenarios.

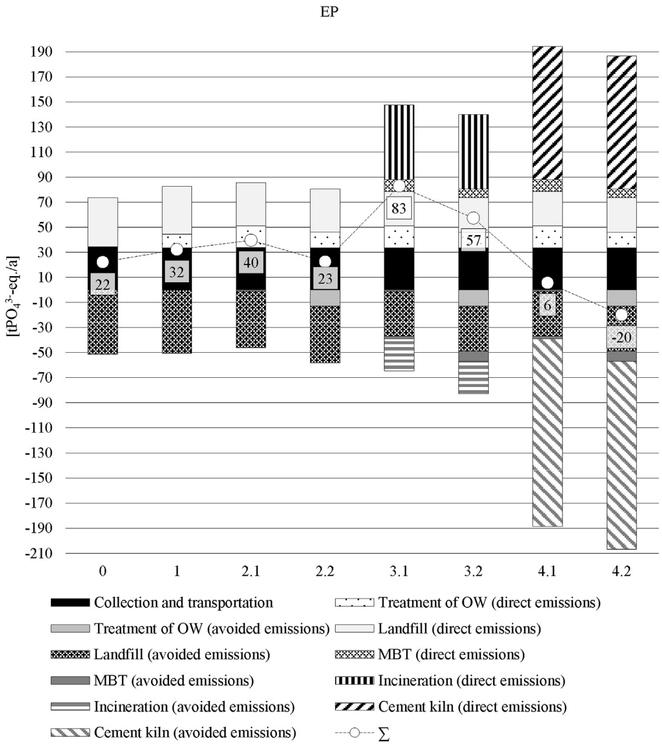


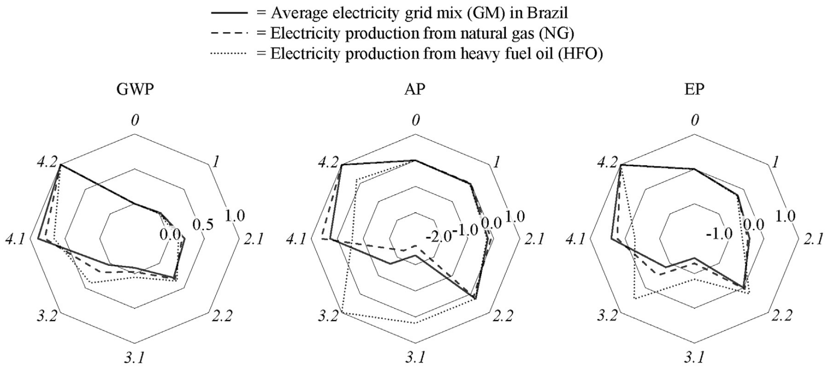
Fig. 8. The EPs of the scenarios.

separately collected organic waste, and usage in cement production is a better utilization option for RDF than incineration. As in the AP impact category, the home composting of organic waste increased the EP of MSW management. However, in this regard, the results were not consistent across the impact categories because home composting decreased the GWP of MSW management.

3.2. Sensitivity analysis

The modeling assumption regarding the kind of electricity production substituted by electricity production from MSW was crucial in terms of the total results in all the impact categories. In the study, it was assumed that the electricity produced would substitute average electricity production in Brazil, which is mainly (~75%) hydropower. Instead of average electricity production, the produced electricity could also substitute marginal energy production. Natural gas is regarded as the most likely fuel for marginal electricity production in Brazil in the foreseeable future (Bernstad Saraiva et al., 2017). Natural gas, as the second largest energy source after hydropower, constitutes a rather large proportion (8.5%) of the average electricity production mix in Brazil. Therefore, it is reasonable to select it for sensitivity analysis as an alternative substituted electricity production. In addition to natural gas, heavy fuel oil was chosen as another alternative energy source for electricity substitution in sensitivity analysis. Heavy fuel oil is the fourth largest energy source (3.6%) in the average electricity production mix in Brazil and is environmentally the most unfavorable (GWP, AP and EP impact categories) alternative of the main energy sources in average electricity production in Brazil. Therefore, it presents simultaneously the worst-case scenario for electricity production and the best-case scenario for electricity substitution — the more avoided emissions resulting from electricity substitution, the more favorable the outcome for electricity production from MSW.

The influence of different electricity substitution options on the results was investigated by determining the weighted results of the scenarios with alternative energy sources (average electricity production mix versus natural gas and heavy fuel oil). Thus, the ranking between the scenarios with different electricity substitution assumptions can be identified. The aim was to find out whether different electricity substitution assumptions make certain treatment methods (e.g. incineration) environmentally more favorable relative to the baseline scenario. The results were weighted relative to the result of the baseline scenario, Scenario 0 (see Fig. 9). The result of Scenario 0 is zero (0.0). If the relatively weighted result (RWR) of a given scenario is > 0.0, the scenario is a better option than Scenario 0. Respectively, if the RWR of a given scenario is < 0.0, the scenario is worse option than Scenario 0. Thus,



Scenario	RWR								
	GWP			AP			EP		
	GM	NG	HFO	GM	NG	HFO	GM	NG	HFO
0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1	0.02	0.02	0.01	-0.03	-0.01	-0.05	-0.24	-0.24	-0.26
2.1	0.22	0.18	0.13	-0.22	-0.10	-0.26	-0.41	-0.47	-0.62
2.2	0.29	0.32	0.36	0.26	0.15	0.25	-0.01	0.05	0.24
3.1	-0.08	-0.03	0.05	-2.36	-2.73	0.24	-1.44	-1.29	-0.84
3.2	0.03	0.18	0.40	-1.64	-2.35	1.00	-0.83	-0.51	0.45
4.1	0.89	0.79	0.66	0.27	0.62	-0.58	0.39	0.22	-0.29
4.2	1.00	1.00	1.00	1.00	1.00	0.19	1.00	1.00	1.00

Fig. 9. Relatively weighted results (RWRs) of the study with different choices for substituted electricity.

the maximum of RWRs is 1.0 in each impact category. If the RWR of a given scenario is 1.0, this implies that the scenario is environmentally the most favorable of all the scenarios in a given impact category. The RWRs were calculated using the following equation:

$$\textit{Relatively weighted result } (RWR) = \frac{Result(Scenario_0) - Result(Scenario_i)}{Max\Delta} \quad (2)$$

where Result ($Scenario_0$) = The net result of the baseline scenario (Scenario 0) in a given impact category;

Result ($Scenario_i$) = The net result of a given scenario in a given impact category;
MaxΔ = The maximum difference between the result of the baseline scenario (Scenario 0) and the results of the scenario with the lowest environmental impacts in a given impact category.

The GWPs of the scenarios did not vary significantly when employing different choices for electricity substitution. When the substituted electricity was produced from natural gas instead of the average grid mix in Brazil, the ranking between the scenarios did not vary. In terms of heavy fuel oil, the GWPs of Scenarios 3.1 and 3.2 decreased somewhat, making incineration a more reasonable treatment option for RDF. Nevertheless, Scenarios 4.1 and 4.2 remained as the most favorable scenarios regardless of the electricity substitution choices. Electricity substitution had significantly more influence on the APs of the scenarios. The ranking of the scenarios remained the same when the substituted electricity was produced by natural gas. However, the ranking of the scenarios changed substantially when the substituted electricity was produced by heavy fuel oil. The APs of Scenarios 3.2 and 3.1 decreased significantly, making incineration a more favorable utilization option for RDF than cement production. In terms of the EPs of the scenarios, variations in the electricity substitution assumptions did not have such a notable influence on the ranking of the scenarios. When substituted electricity was produced with heavy fuel oil, the EP of Scenario 3.2 was the second lowest and Scenario 4.2 remained the most viable option. The different modeling assumptions used for electricity substitution did not have an influence on the organic waste treatment options: AD was the better treatment option for organic waste than composting regardless of the changes. Similar sensitivity analysis has been carried by Goulart Coelho and Lange (2016). In their study, variations in the electricity grid mix did not have a notable influence on the results: the ranking of the scenarios remained the same despite the changes. However, it should be noted that the changes employed in their study were more subtle since they varied the shares of different energy sources in the average electricity grid mix. Therefore, the two studies are not fully comparable in this regard.

The collection rate of LFG is a key parameter of landfilling; it typically has a major influence on the total results particularly in the GWP impact category (Liikanen et al., 2017). Therefore, this study employed different LFG collection rates in the sensitivity analysis in order to discover whether the results would vary if the LFG collection rates were lower. In the modeling, the LFG collection rate was 80% in the CTL landfill and 64% in the CTVA Caieiras landfill. These values are somewhat higher than typically found in Brazil. For instance, Mendes et al. (2004) and Bernstad Saraiva et al.

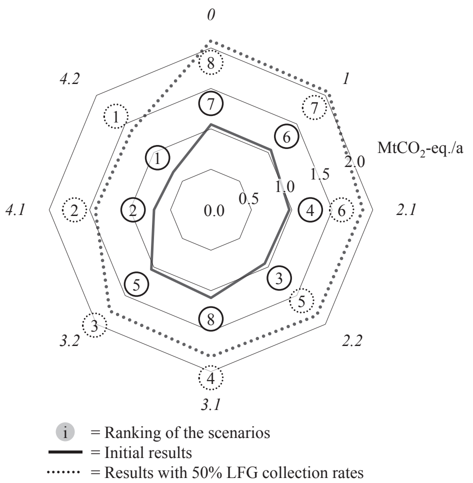


Fig. 10. Sensitivity analysis results with 50% LFG collection rates in the GWP impact category.

(2017) employed 50% LFG collection rates in their LCA studies (São Paulo and Rio de Janeiro were the case areas in these studies). Consequently, a 50% collection rate for LFG in both landfills was also applied in the sensitivity analysis (see Fig. 10, where the results of the sensitivity analysis are presented for GWP).

Lower LFG collection rates had a noteworthy influence on the GWPs of the scenarios. The incineration of RDF (Scenario 3) improved substantially in this regard – the GWPs of Scenarios 3.2 and 3.1 were the third and fourth lowest, respectively. However, the main findings of the results remained the same: (1) AD is a more favorable treatment option for organic waste than composting, and (2) RDF usage in a cement kiln is a more favorable utilization option than incineration. It should also be noted that the GWP of every alternative scenario was lower than the baseline scenario. In terms of the AP and EP impact categories, lower LFG collection rates did not have a notable influence on the results – the ranking of the scenarios remained the same.

3.3. Discussion

The results indicated that of the scenarios assessed in the study, the environmental impacts of MSW management in São Paulo can be most effectively decreased by AD of the organic waste and MBT of the residual MSW on condition that the produced RDF is utilized in cement production. The home composting of organic waste was beneficial from the GWP point of view. Home composting had, however, the opposite effect in the AP and EP impact categories, indicating a need to analyze the results further by for example weighting (EN ISO 14044, 2006) or multi-criteria decision analysis, which has been applied in similar studies (e.g. Angelo et al., 2017). Nevertheless, home composting can diminish the costs of MSW management activities if it decreases collection frequencies since collection is typically a significant cost factor in MSW management (Oliveira et al., 2017). Therefore, current activities promoting home composting in São Paulo (Composta São Paulo, 2017) are justified.

Case area-specific conditions and characteristics have a significant influence on the environmental impacts of different MSW treatment methods since the environmental impacts of the surrounding systems often override the environmental impacts of the treatment processes (Ekvall et al., 2007). Electricity production and substitution is an example of this phenomenon. In Brazil, where average electricity production is dominated by hydropower, which has rather low environmental impacts, the incineration of MSW is not particularly favorable in terms of the environmental impacts assessed in the study due to the low amount of avoided emissions resulting from electricity substitution. Instead of the average electricity grid mix, electricity produced from MSW could also substitute marginal electricity production (e.g. electricity production from natural gas). The effect of different assumptions regarding electricity substitution was assessed, and it was found that the incineration of MSW is more favorable relative to landfilling when other electricity production (more precisely electricity production from natural gas or heavy fuel oil) is substituted. One exception was that the AP of incineration increased in relation to landfilling when electricity produced by natural gas was substituted instead of average grid mix in Brazil. Therefore, in terms of the environmental performance of different MSW treatment methods (particularly incineration), electricity substitution can be a determining factor in LCA studies.

Data uncertainty and variability is inherently part of MSW management LCA studies (Clavreul et al., 2012). This study is no exception. Data uncertainty was assessed by way of an example – LFG collection efficiency. The LFG collection rates employed in the study were higher than typically employed in LCA studies with similar characteristics. Therefore, it was investigated whether lower LFG collection rates would have a significant influence on the results of the study. The ranking of the scenarios in the GWP impact category changed notably when 50% LFG collection rates were applied instead of the initial rates (80% and 64%). The main reason for this change was that the incineration of RDF became more favorable when the environmental impacts of landfilling increased. Nevertheless, the main findings of the study remained the same regardless of the change.

The results of the study are in line with previous literature. Based on this study and other literature studies (e.g. Goulart Coelho and Lange, 2016; Mendes et al., 2003; Soares and Martins, 2017) a consensus can start to be formed regarding the environmentally most favorable treatment method for organic waste. In terms of the environmental impacts of incineration and landfilling, this study and other studies clearly indicate that incineration is not as favorable in Brazil as in other countries (e.g. China) due to the rather low amount of avoided emissions resulting from electricity substitution (e.g. Goulart Coelho and Lange, 2016; Mendes et al., 2004). However, as the sensitivity analysis of the study demonstrated, various factors have an influence on the environmental performance of incineration. It should be particularly kept in mind when interpreting the results of MSW management LCA studies.

4. Conclusions

Landfill disposal has thus far been the predominant treatment method for MSW in São Paulo city. Environmentally sustainable development of MSW management in São Paulo, however, necessitates a stepwise reduction of landfilling. Stepwise improvements towards more environmentally sustainable MSW management in São Paulo were introduced and their environmental impacts investigated. The results indicated that of the proposed treatment alternatives, environmental impacts of MSW management in São Paulo can be most effectively decreased by anaerobic digestion of source separated organic waste and MBT of MSW, on condition that the produced RDF is utilized in cement production as a substitute for coal. The study focused solely on potential treatment alternatives for generated MSW, and therefore other viewpoints of environmentally sustainable MSW management, such as waste prevention and reuse, were not taken into account. These viewpoints are, however, important part of environmentally sustainable MSW management, and should be prioritized before conventional MSW treatment methods. The results of the study provide guidelines for decision- and policy-making from the environmental point of view. The results of the study can be utilized in further studies together with social and economic impact assessment to find the overall sustainability of different MSW management alternatives in the case area, and to provide insight into developing MSW management in other areas, too.

Acknowledgements

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.jclepro.2018.06.005.

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Publication IV

Liikanen, M., Grönman, K., Deviatkin, I., Havukainen, J., Hyvärinen, M., Kärki, T.,
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Construction and demolition waste as a raw material for wood polymer composites – Assessment of environmental impacts

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Construction and demolition waste as a raw material for wood polymer composites – Assessment of environmental impacts

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ABSTRACT

The European Commission's ambitious construction and demolition waste (CDW) material recovery target has placed pressure on Finland to increase its CDW material recovery rate. It has been identified that using CDW fractions, e.g. waste wood, plastic, mineral wool and plasterboard, as raw materials for wood polymer composites (WPCs) may help in reaching the CDW material recovery target. The objectives of this study were to assess the environmental impacts of WPC production using specific CDW fractions, namely wood, plastic, plasterboard and mineral wool, as raw materials, and to compare these impacts with the baseline situation where these CDW fractions are treated with conventional waste treatment methods such as landfilling and incineration. The study focused primarily on the depletion of fossil hydrocarbons and climate change. The results indicate that, when compared to the baseline situation, the environmental impacts of CDW management can be decreased when CDW fractions are used in WPC production. By substituting WPCs for plastic or aluminium, considerable environmental benefits can be achieved in terms of the aforementioned impact categories. Due to the differences in the physical and mechanical properties of WPCs compared to plastic and aluminium, WPCs cannot necessarily substitute them in a mass-based ratio of 1:1. This was acknowledged in the study by identifying minimum substitution rates for different materials. For instance, the produced WPCs should substitute at least 6% of plastic and 8% of aluminium in order to decrease the impact on climate change compared to the advanced waste management scenario. Therefore, in applications where WPCs can be used as a substitute for these materials, WPC product design and development should be prioritised.

1. Introduction

In recent decades, global waste generation has increased significantly due to the strong correlation between urbanisation and economic development and waste generation. There is no forecasted slowdown; indeed, quite the contrary (World Bank, 2012). Waste presents a continuous challenge in the modern world which must be tackled to guarantee a sustainable future. Therefore, measures to find suitable and more sustainable treatment methods for different waste fractions must be identified in order to minimise the environmental impacts of generated waste in different corners of the globe.

Another driver for waste management development is resource scarcity. The inevitable trend is to shift from waste management to resource management (Arm et al., 2017). Use of renewable materials and a shift from a single-use linear economy towards a circular economy are both means for tackling resource scarcity. The European Union (EU) has taken steps towards a circular economy with its Circular Economy Action Plan. This plan includes various ambitious measures for different phases of a product's life cycle, from production (e.g. measures that improve the durability, reparability and recyclability of product design) to end-of-life (e.g. material recovery targets for different waste streams) (European Commission, 2018). This calls for both concrete product development actions and effective treatment technologies for various waste streams.

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In the EU, construction and demolition waste (CDW) accounts for approximately 30% of total waste generation. Annually, this equates to 800 million tonnes. (European Commission, 2016a, 2016b.) Additionally, CDW contains a range of valuable materials such as minerals, plastics, metals and wood. For these reasons, the EU has set an ambitious material recovery target for its member states. By 2020, the material recovery rate of non-hazardous CDW in the EU (achieved through re-use, recycling or alternative methods of material recovery) should reach 70% (European Commission, 2016a). Even though the definitions for CDW are different across EU member states, thus hindering cross-country comparisons (Deloitte, 2015), it is evident that the CDW material recovery rate varies significantly between EU member states. Some, such as Austria and Germany, have already reached the target, while others, such as Finland, lag behind (European Commission, 2016a). In Finland, the current CDW material recovery rate is 58% (Salmenperä et al., 2016). In addition to material recovery rates, the composition of CDW also varies notably within the EU. Some common characteristics can, however, be identified. Minerals typically compose a large portion of CDW. In Finland, for instance, it has been assessed that minerals typically compose 35% of CDW. The portion of wood in CDW distinguishes Finland from other EU member states. In Finland, the portion of wood in CDW (36%) is notably higher compared to some other EU member states, where the share is as low as 2–4% (Dahlbo et al., 2015). Metals and other materials such as plastic and cardboard, compose typically a lower portion of CDW. Rocks, soil and gravel are not considered in these composition proportions.

Various waste materials have been identified as potential raw materials for wood polymer composites (WPCs) (e.g. Kazemi Najafi, 2013; Keskisaari and Kärki, 2016; Sommerhuber et al., 2015; Vidal et al., 2009). As some of the mechanical properties of WPCs, such as strength and stiffness, are lower than those of solid wood (Sain and Pervaiz, 2008), they are most commonly used in applications that do not require good structural performance. WPCs are commonly used in building materials, such as decking boards (Bolin and Smith, 2011; Sun et al., 2017) and panels (Suoware et al., 2019), and automotive components (Ashori, 2008). Apart from these, some more specific uses have been identified and tested for WPCs. For instance, pallets have been manufactured using WPCs (Korol et al., 2016; Soury et al., 2009). Compared to other composite materials such as cement bonded composites, WPCs can be regarded as value-added materials due to their versatile uses. Cement bonded composites, for instance, have been predominantly used as building materials (Ashori et al., 2012; Li et al., 2019). CDW has shown to be a potential raw material feedstock for WPC production (Keskisaari et al., 2016). In addition to the wood and plastic fractions of CDW, mineral wool has also been found to be a suitable raw material for WPCs (Väntsi and Kärki, 2014). Consequently, WPC production could help in reaching the material recovery target for CDW. When using CDW in WPC production, conventional waste treatment activities and methods such as landfilling and incineration are avoided. Furthermore, use of CDW as a raw material for WPC production is a concrete step towards resource efficiency when virgin materials such as plastic and wood are substituted (Osburg et al., 2016; Teuber et al., 2016).

Keskisaari et al. (2016) studied how use of CDW as raw material in WPCs impacts the mechanical properties of the material. They discovered both negative and positive outcomes. The use of CDW decreases the modulus values and flexural strength of WPCs, whereas it increases its impact strength. Without impacting WPC-derived products, this should be acknowledged in product and structural design. For instance, WPCs made of CDW should primarily be used in applications where these negatively-affected mechanical properties do not inhibit their utility. Further, use of waste materials can influence the compatibility of WPCs due to the potential for remaining impurities. This risk can be controlled by adding the correct amount of the right coupling agent to the composite mixture (Gao et al., 2010; Wang et al., 2017).

Life cycle assessment (LCA) is a method for evaluating the potential environmental impacts of a product or system (EN ISO 14040, 2006; EN ISO 14044, 2006) and has been applied in the environmental impact assessment of numerous materials, including WPCs (e.g. Bolin and Smith, 2011; Feifel et al., 2015; Sommerhuber et al., 2017; Väntsi and Kärki, 2015). Previously published literature on the LCA of WPCs can be grouped into two categories: (1) studies comparing the environmental impacts of WPCs and other materials (e.g. wood); (2) studies assessing the environmental impacts of WPCs made with different raw materials (virgin materials versus recycled materials).

The studies of Bolin and Smith (2011) and Feifel et al. (2015) fall within the first category. Bolin and Smith (2011) compared the environmental impacts of decking made of alkaline copper quaternary (ACQ) treated wood and WPC which was produced using both virgin and recycled raw materials (50% recycled wood, 25% recycled high-density polyethylene (HDPE) and 25% virgin HDPE). In their study, the environmental impacts of ACQ wood decking were found to be lower than those of WPC decking as, among other reasons, they had lower fossil energy consumption. Feifel et al. (2015) compared the environmental impacts of decking made from two different types of WPC (mixtures of PE and wood and of polyvinyl chloride (PVC) and wood, with all raw materials assumed to be virgin) to those of decking made from tropical wood (bilinga) or pressure-impregnated pine. They discovered that the environmental impacts of WPC decking were higher than those of pine decking but lower than those of bilinga decking. This raised the question of whether the results would be different if recycled materials had been used instead of virgin materials.

The studies of Sommerhuber et al. (2017) and Väntsi and Kärki (2015) fall into the second category, emphasising raw material selection. Sommerhuber et al. (2017) assessed the environmental impacts of WPCs made from both virgin and recycled (waste) materials. The raw materials were virgin wood with HDPE and waste wood with recycled HDPE. They found that the environmental impacts of WPCs made from waste and recycled materials were lower than those of WPCs made from virgin materials. Väntsi and Kärki (2015) assessed the environmental impacts of different types of WPCs: WPCs made of virgin wood and virgin glass fibre or recycled mineral wool and WPCs made of virgin wood and virgin or recycled polypropylene (PP). They found that by using recycled mineral wool instead of virgin glass fibre, the environmental impacts of WPCs were reduced in all assessed categories; these categories were global warming, acidification, eutrophication and abiotic depletion potential. Use of recycled PP was found to decrease the potential for global warming and abiotic depletion. Previous literature about the environmental impact assessment of WPCs, including the above-mentioned studies, has assessed the environmental impacts of WPCs from a product perspective rather than considering the environmental impacts of WPC production as part of a CDW management system. Therefore, the environmental impacts of using CDW in WPC production rather managing it in a conventional manner have not yet been comprehensively assessed.

This study intends to assess the environmental impacts of WPC production as a material recovery option for CDW. The CDW fractions assessed in the study are wood, plastic, mineral wool and plasterboard. These have been identified as suitable raw materials for WPC and henceforth, references to CDW in this study will specify these particular CDW fractions. The geographical location for the study is Finland. In the baseline or reference situation, CDW fractions are treated with conventional waste treatment methods, such as landfilling and incineration. The primary objective of the study is to discover how the environmental impacts of CDW management would change if CDW were used as raw materials in WPC production. The following research questions will be explored in this study:

- How does using CDW as raw materials in WPC production compare to the current situation, where CDW is treated as waste (i.e. composite production versus conventional waste treatment)?
- What CDW fractions should be preferred as raw materials for WPCs?
- What is the influence of substituting virgin materials by WPCs (i.e. which virgin materials should be substituted and in what quantities)?

2. Materials and methods

2.1. Wood polymer composites

WPCs typically contain a specific combination of filler material (most commonly wood), thermoplastic (plastic that can be repeatedly softened by heating) and additives (e.g. coupling agents and lubricants) (Teuber et al., 2016). Other filler materials such as mineral wool have also been used (Väntsi and Kärki, 2014). The proportion of raw materials results from the desired physical and mechanical properties of the WPC as well as its production technique. Plastic and filler are the two main raw materials, each constituting 30–70% of the total WPC mass. The proportion of filler in a WPC mixture strongly affects its mechanical properties. For instance, an increase in the proportion of wood (from 30% to 50% of the total mass) increases the tensile stiffness of the WPC while decreasing its elongation at break and impact strength (Sain and Pervaiz, 2008.)

Additives such as lubricants and coupling agents are used to enhance WPC performance or to facilitate their manufacture. Lubricant is used to improve the rheology of WPCs or, in other words, how the mixture behaves in processing. This therefore facilitates the production process. Typically, stearates and esters are used as lubricants. Coupling agent is used to improve the homogeneity of filler and polymer materials. Typically, maleated polyolefins are used as coupling agents. A lack of homogeneity in composites can result in unsatisfactory structural and mechanical properties. Therefore, coupling agents are commonly used in WPCs. Additives constitute approximately 5% of the total WPC mass (Satov, 2008.)

2.1.1. Construction and demolition waste as a raw material for wood polymer composites

Wood is used in a fibrous form, such as flour, and can be either virgin or recycled (Keskisaari and Kärki, 2016). In WPC production, PE and PP are widely-used polymers (Clemons, 2008; Keskisaari and Kärki, 2016). Both plastic types are also commonly used in construction materials (Turku et al., 2017). Both virgin and recycled plastics have been used in WPC production (Keskisaari and Kärki, 2016). Mineral wool and plasterboard have been identified as potential raw materials for WPCs (Keskisaari et al., 2016; Väntsi and Kärki, 2014). While they are alternative filler materials, they can only substitute wood to a certain extent. For instance, in a study by Keskisaari et al. (2016), in a WPC containing mineral wool and plasterboard, the composition was as follows: 40% mineral wool and gypsum from plasterboard (also known as gypsum board, drywall, or gypsum panel), 30% PP, 24% wood and 6% additives. In Finland, both mineral wool and plasterboard are common construction materials. Therefore, they are also common CDW fractions.

The WPC recipes assessed in this study are presented in Fig. 1. Recipe 1 is a conventional WPC containing waste wood, plastic and additives and can be regarded as a baseline recipe for WPCs in this study. Recipe 2 presents an alternative to Recipe 1. In Recipe 2, mineral wool and plasterboard substitute a proportion of the wood. In both recipes, the proportions of filler material (wood, mineral wool and plasterboard) and plastic are the same. While Recipe 1 represents a conventional WPC, Recipe 2 is a somewhat experimental recipe. Nevertheless, the raw materials used in both recipes are all suitable for WPCs. Additionally, wood, plastic, mineral wool and plasterboard are common construction materials in Finland and, therefore, are also common CDW fractions. For these reasons, these CDW fractions were identified as potential raw materials for WPCs. Since the study focuses on the environmental impacts of WPC production and potential raw materials are represented in the recipes, these recipes were deemed to be representative. Therefore, this study performs no further analysis on variations in the proportions of the raw materials. The recipes were adapted from the studies of Turku et al. (2017) (Recipe 1) and Keskisaari et al. (2016) (Recipe 2).

2.1.2. Production technology description

Production processes for WPCs, including the machinery used, originate from plastic production (Pritchard, 2004). The most common processes are extrusion and injection moulding. Fig. 2 illustrates the WPC production process. As pre-processing methods, crushing and hammer mill are used to reduce the size of raw materials. Additionally, magnets are used to remove any remaining metal items (e.g. nails), preventing machinery wear. The intended feedstock (e.g. wood, plastic, mineral wool and gypsum) is assumed to have no negative influence on the machinery. Next, particles are agglomerated into compounds. In agglomeration with a hot/cooling batch mixer, raw materials such as polymers, wood fibres/flour and additives are blended together to provide a homogenous mixture. (Gardner et al., 2015.) After the agglomeration, the material mixture is typically presented as granules or pellets to simplify its further processing in an extruder or injection moulding machine.

Injection moulding is a commonly-applied technology that is used to manufacture high quantities of products with complex geometries (Mali and Rautiainen, 2005). Using extrusion technology, linear profiles are produced by forcing a molten composite mixture through a die (Migneault et al., 2009). Extrusion, which is the method used in this study, can be divided into single or two-stage extrusion processes. In single-stage extrusion, material mixing and profile extrusion are performed in a single step whereas, in

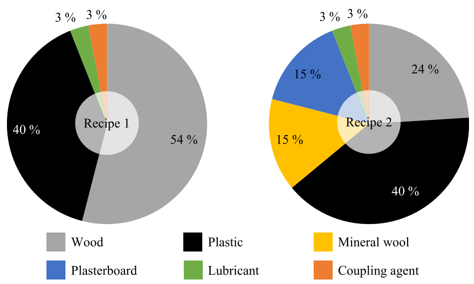

Fig. 1. WPC recipes assessed in the study (Keskisaari et al., 2016; Turku et al., 2017).

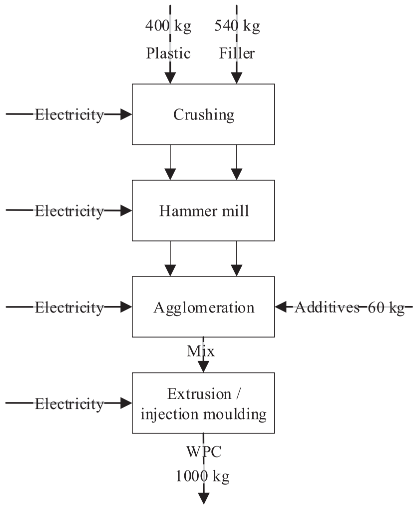

Fig. 2. WPC production process.

two-stage extrusion, the compounding and profile extrusion follow two separate processes.

An extruder is equipped with a hopper, a barrel and a single or twin screw. Raw materials, usually as granules or pellets, are loaded into the hopper and then fed into the extruder either as pre-compounded pellets or separate non-compounded materials such as powder blends. (Gardner et al., 2015.) The extruder produces an easily-modifying and homogeneous material mixture using friction, pressure and heat and pushes the mixture through a die. The produced profile is then calibrated, cooled and cut to a certain length (Wagner et al., 2014).

2.2. Life cycle assessment

Following the principles and requirements of the ISO standards 14040 (2006) and 14044 (2006), LCA was used to assess environmental impacts. ISO-standardised LCA has been recognised by the international scientific community as a tool for identifying and enhancing the environmental performance of products and systems. To compare the environmental performance of different materials, LCA is considered to be the most suitable method (e.g. Arena and de Rosa, 2003; Ortiz et al., 2009.) Furthermore, in Europe, LCA is the most commonly-applied systems analysis method in the field of waste management (Pires et al., 2011).

This study focuses principally on the impact categories of climate change (excluding biogenic carbon) and the depletion of fossil resources (fuels). These impact categories are relevant due to the emissions generated during waste treatment activities, WPC production and the production of virgin materials, which is avoided when these are substituted with WPCs. Additional impact categories (in total 19 impact categories such as eutrophication and acidification) are also assessed, albeit in less detail (see Supplementary Material A for further information). The modelling was carried out using GaBi LCA modelling software (version 8.7.0.18) (Thinkstep, 2017). ReCiPe 2016 v.1.1 (midpoint, hierarchist time-frame) was used to assess impact (RIVM, 2018; Thinkstep, 2018).

The functional unit for the study is the treatment of 940 kg of CDW, which, according to the specified recipes, corresponds to 1,000 kg of produced WPC. Therefore, the reference flow of the study is 1t of produced WPC.

2.2.1. Scenarios and calculation principles

This study begins with the generation of 940 kg of CDW using the so-called zero-burden approach; that is, it presupposes that the environmental impacts of CDW from previous life cycle phases are excluded from the assessment (Ekvall et al., 2007). The generated CDW is either treated via conventional waste management methods or used in WPC production. In the baseline scenario, Scenario 0, wood and plastic are incinerated with energy recovery in a waste incineration plant. This is currently the most common treatment method for CDW wood and plastic in Finland. The energy produced is assumed to substitute the average district heat (Statistics Finland, 2018) and electricity produced in Finland. It can also substitute other energy production such as that which uses natural gas. This is further analysed in a later sensitivity analysis. The distance for transportation to a waste incineration plant is 120 km. Landfill disposal is no longer a potential treatment method for wood and plastic due to the landfill ban on organic waste that has been in force in Finland since 2016. Therefore, the scenarios do not include landfilling with wood and plastic. Mineral wool is sent to landfill since no widespread material recovery techniques have yet been established in Finland. Due to the low organic carbon content of mineral wool, its landfilling is permitted. Finland also landfills plasterboard but recovers gypsum that has been separated from it. Plasterboard material recovery is not yet widespread. Therefore, in the baseline scenario, plasterboard is sent to landfill. Scenario 0 was mainly modelled using the GaBi LCI data (Thinkstep, 2017) (see Supplementary Material B for further information on the Scenario 0 LCI data).

In Scenario 1, more advanced waste treatment techniques are used on plastics and plasterboard. 30% of plastics are recovered conventionally as material (so-called mono-material recovery); that is, plastic granulates are manufactured from the waste-derived plastics. The plastic granulates then substitute virgin HDPE granulates. The remaining 70% are used as energy in a waste incineration plant. As in Scenario 0, the produced energy substitutes the average district heat and electricity production in Finland. The split between material and energy recoveries is based on the assumption that plastics are first collected from a construction or demolition site with a limited sorting efficiency and accuracy and then separated mechanically with a limited separation rate. As a result of this, the proportion of plastics recovered as material is assumed to be 30%. The recovered plastics substitute virgin HDPE granulates in a substitution ratio of 0.73:1 (Andreasi Bassi et al., 2017). Plasterboard consists of 96% gypsum and additives and 4% paper (Jimenez Rivero et al., 2016). Plasterboard waste is mechanically treated (e.g. through crushing and sieving) in order to separate gypsum and paper. The separated gypsum is used in the production of new plasterboard. Thus, conventional gypsum (more precisely, flue gas desulphurisation (FGD) gypsum) is substituted in a market- and mass-based ratio of 0.19:1. According to Fisher (2008), this is estimated to be the maximum proportion for recycled gypsum in new plasterboard. The separated paper contains impurities (i.e. gypsum and additive residues) and is therefore incinerated in a waste incineration plant. Mineral wool and wood are treated in a similar manner to that in Scenario 0; mineral wool is sent to landfill and wood is incinerated. As mentioned above, there is no well-established and widespread material recovery system for mineral wool in Finland and, therefore, it is disposed of in landfill. Its material recovery is excluded from Scenario 1. Material recovery methods for wood waste do exist in Finland; however, energy recovery has thus far been the most feasible and predominant treatment method used (Piippo, 2013). Therefore, energy recovery was also selected as a treatment method for wood in Scenario 1. Scenario 1 was modelled using data found in both GaBi's database (Thinkstep, 2017) and in the literature (Andreasi Bassi et al., 2017; Fisher, 2008; Jimenez Rivero et al., 2016) (see Supplementary Material C for further information on the Scenario 1 LCI data).

In scenarios 2–4, CDW is used as raw material for WPCs. The WPC production process was modelled using the data presented in Table 1. Scenarios 2, 3 and 4 differ from one another in terms of the material substituted by the produced WPC. In Scenario 2, the produced WPC substitutes virgin plastic. There are three sub-scenarios in Scenario 2, each different in terms of its substituted plastics. The following types of plastic are substituted in the sub-scenarios: PP in Scenario 2.1, PVC in Scenario 2.2 and HDPE in Scenario 2.3. Different types of plastics were selected in order to identify the influence of substituted plastic type on the environmental impacts of WPC production.

In Scenario 3, CDW is also used in WPC production. Instead of substituting plastic, the WPC substitutes wood. There are four different sub-scenarios in Scenario 3, each different in terms of its substituted wood materials. The following materials are substituted by WPC in the sub-scenarios: plywood in Scenario 3.1, solid timber in Scenario 3.2, laminated wood in Scenario 3.3 and particle board in Scenario 3.4. Different wood materials are substituted to determine the effect of the substituted wood material on the environmental impacts of WPC production – untreated wood versus further processed wood-based materials.

Since plastic and wood are the main raw materials in WPCs, they are also materials that can, most likely, be substituted by WPCs due to their somewhat similar properties; for instance, the strength and stiffness of WPCs, as examples of mechanical properties, are between those of plastic and wood (Sain and Pervaiz, 2008). In addition to wood and plastic, WPCs can substitute other materials in specific applications. This study also assesses the substitution of aluminium profiles with WPCs. In Scenario 4, the produced WPC substitutes an aluminium profile made of 75% recycled aluminium and 25% virgin aluminium. This ratio represents the standard aluminium production in Finland (Kuusakoski, 2018) (see Supplementary Material D for further information on the LCI data for Scenarios 2–4). The study scenarios are summarised in Table 2 and illustrated in Fig. 4.

In Scenarios 2–4, the produced WPC substitutes virgin material in a mass-based ratio of 1:1–1,000 kg of WPC substitutes to 1,000 kg of virgin material. This is due to the system boundaries of the study (see section 2.2.2 for further information). However, due to the different mechanical and physical properties of the materials, the substitution ratio can be lower than 1:1. This is particularly noteworthy and, therefore, will be further analysed in the sensitivity analysis.

2.2.2. System boundaries

The system boundaries include direct emissions generated during the transportation (Lipasto, 2017) of CDW to a waste incineration plant and material recovery facilities, landfill disposal of mineral wool and plasterboard, incineration of plastic and wood and WPC production (see Fig. 4). The system boundaries also consider the avoided emissions that would originate from substituted energy (i.e. electricity and district heat) and material production (i.e. plastic, wood and aluminium). The transportation of CDW to the waste treatment centre does not have an influence on the differences between the scenarios because all scenarios assume the same distance. This is therefore excluded from the system boundaries. In Scenarios 2–4, WPC is produced through extrusion (see Fig. 3). After extrusion, the produced WPC profile can be further manufactured through different post-production processes such as compression moulding (Toghyani et al., 2018). Therefore, the system boundaries end at WPC production and not the specific products made from it. The final use, purpose and function of a product determine whether WPC can replace virgin material and the extent of the substitution. Since no specific product is manufactured in the study, it is assumed that WPC substitutes virgin material in a mass-based ratio of 1:1. Therefore, for both WPC and virgin materials, the use and end-of-life phases lie outside the system boundaries. Regardless of this exclusion, the study provides valuable information, from a CDW management perspective, on the extent to which WPC production can decrease the environmental impacts of CDW management. Additionally, it provides information on the development of products made from WPC in terms of which conventional materials should be substituted by WPCs to reduce the environmental impacts.

3. Results and discussion

3.1. Contribution analysis

Fig. 5 illustrates the contribution of each scenario to climate change. The results of each scenario are shown in 2 bars, side by side, that represent the two investigated recipes for WPC production (see Fig. 1). The baseline scenario, Scenario 0, makes the highest contribution to climate change: 480 kg CO_2-eq. (for CDW fractions which would be used in WPC production based on Recipe 1) and 620 kg CO_2-eq. (Recipe 2). Scenario 1, with an advanced material recovery for plastics and plasterboard, had an impact on climate change of 180 kg CO_2-eq. in Recipe 1 and 316 kg CO_2-eq. in Recipe 2, representing respective reductions of 62% and 49% when compared to the baseline scenario.

A significantly reduced impact on climate change was achieved in Scenario 2, where the produced WPCs substitute different types of plastics. The emissions of plastic production avoided through material substitution, shown as negative emissions in Fig. 5, substantially outweigh the emissions generation during the

Table 1
LCI data of WPC production.

Parameter	Value	Unit	Reference
Diesel consumption of a wheel loader used in the process	0.5	$dm^3/t_{material}$	Expert estimation
Electricity consumption during the pneumatic moving	180	$kJ/kg_{material}$	Expert estimation
Electricity consumption during the crushing (mineral wool and plasterboard)	83	$kJ/kg_{material}$	Gao et al. (2001)
Electricity consumption during the crushing (plastic and wood)	62	$kJ/kg_{material}$	Gao et al. (2001)
Electricity consumption of the hammermill (mineral wool)	348	$kJ/kg_{material}$	Gao et al. (2001)
Electricity consumption of the hammermill (plastic and wood)	2500	$kJ/kg_{material}$	Väntsi and Kärki (2015)
Electricity consumption during the agglomeration	1440	$kJ/kg_{material}$	Expert estimation
Consumption of the coupling agent (maleated PP, MAPP)	3	% of total mass	Keskisaari et al. (2016); Turku et al. (2017b)
Consumption of the lubricant	3	% of total mass	Keskisaari et al. (2016); Turku et al. (2017b)
Electricity consumption during the extrusion	1800	$kJ/kg_{material}$	Väntsi and Kärki (2015)

Table 2
Study scenarios and their CDW mass flows (MW = mineral wool; PB = plasterboard).

Scenario	Recipe	Landfill [kg]		Incineration [kg]		Material recovery [kg]		WPC production [kg]	Substituted material [kg]	
		MW	PB	Plastic	Wood	Plastic	PB	All CDW fractions		
S0	R1	-	-	400	540	-	-	-	-	-
	R2	150	150	400	240	-	-			
S1	R1	-	-	280	540	120	-	-	-	-
	R2	150	-	280	240	120	150			
S2.1	R1	-	-	-	-	-	-	940 CDW + 60 additives = 1,000 WPC	PP	1,000
	R2	-	-	-	-	-	-			
S2.2	R1	-	-	-	-	-	-		PVC	1,000
	R2	-	-	-	-	-	-			
S2.3	R1	-	-	-	-	-	-		HDPE	1,000
	R2	-	-	-	-	-	-			
S3.1	R1	-	-	-	-	-	-		Plywood	1,000
	R2	-	-	-	-	-	-			
S3.2	R1	-	-	-	-	-	-		Solid timber	1,000
	R2	-	-	-	-	-	-			
S3.3	R1	-	-	-	-	-	-		Laminated wood	1,000
	R2	-	-	-	-	-	-			
S3.4	R1	-	-	-	-	-	-		Particle board	1,000
	R2	-	-	-	-	-	-			
S4	R1	-	-	-	-	-	-		Aluminium profile	1,000
	R2	-	-	-	-	-	-			

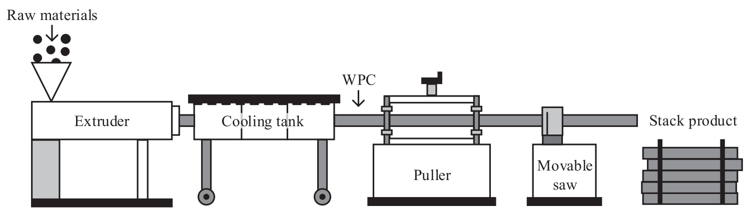

Fig. 3. WPC profile extrusion line (Wagner et al., 2014).

production of WPCs. As shown, the lowest contribution is achieved in Scenario 2.3, where the produced WPCs substitute PVC plastic. In that scenario, the impact on climate change is almost −1,800 kg CO_2-eq.

In Scenario 3, different types of wood materials were substituted with WPC. In this scenario, the contribution to climate change varies between 100 kg and 200 kg CO_2-eq. This positive contribution implies that the direct emissions generated in the production of WPCs are greater than the emissions avoided through wood substitution. Such an outcome can be expected since wood harvesting and processing do not make a significant contribution to climate change. With the assumption that wood is a carbon-neutral material, its substitution does not result in significant emission reductions. However, if the end-of-life phase for products made of WPCs were included in this study, the impact on climate change would change as the environmental impacts of incinerating wood rather than plastic would be significantly lower. In Scenario 4, substituting aluminium by the produced WPCs results in a significant negative contribution to climate change (−2,100 kg CO_2-eq.). This is due to the high energy intensity of aluminium profile production. These results indicate that, from a climate change perspective, WPC production using particular CDW fractions (i.e. wood, plastic, mineral wool and plasterboard) is a recommendable alternative to traditional waste treatment practices. Significant emission reductions are achieved through the substitution of energy intensive materials such as plastics and aluminium.

The difference between Recipe 1 and Recipe 2 (see Fig. 1) in terms of their contribution to climate change is more notable in Scenarios 0 and 1. In Scenarios 2–4, however, no such noteworthy difference can be detected. In Scenario 0, the emissions of Recipe 2 were 30% higher than those of Recipe 1. In Scenario 1, the difference is more significant: Recipe 2 emissions contributing to climate change were 76% higher than those of Recipe 1. This difference results from the lower amount of emissions avoided through energy substitution, since less wood is incinerated with energy recovery in Recipe 2. With Recipe 2, a lower material substitution rate is enough for WPC production to contribute less to climate change than conventional waste treatment activities (Scenarios 0 and 1). This indicates that, from a climate change perspective, Recipe 2, comprising a lower share of wood and also including plasterboard and mineral wool, is preferable to Recipe 1 as it results in higher emission reductions.

Fig. 6 illustrates the results for the depletion of fossil fuels. The highest contribution within this impact category is detected in Scenario 3. This is due to the biological origin of wood and its neutral effect on this impact category. Therefore, the negative contribution caused by the avoided wood material production is minor compared to that in Scenarios 2 and 4. In Scenario 3, WPC production would require more fossil fuel consumption than it would prevent, resulting in a net positive impact of 160–190 kg oil-eq. All other scenarios result in net negative contributions to fossil fuel depletion. As expected, Scenario 2 achieves a significant amount of prevented fossil fuel consumption, with plastics, refined from crude oil, being substituted by WPCs. Depending on the substituted plastic type, the contribution varies between −1,000 and −1,400 kg oil-eq. In Scenario 4, the avoided impact is −500kg

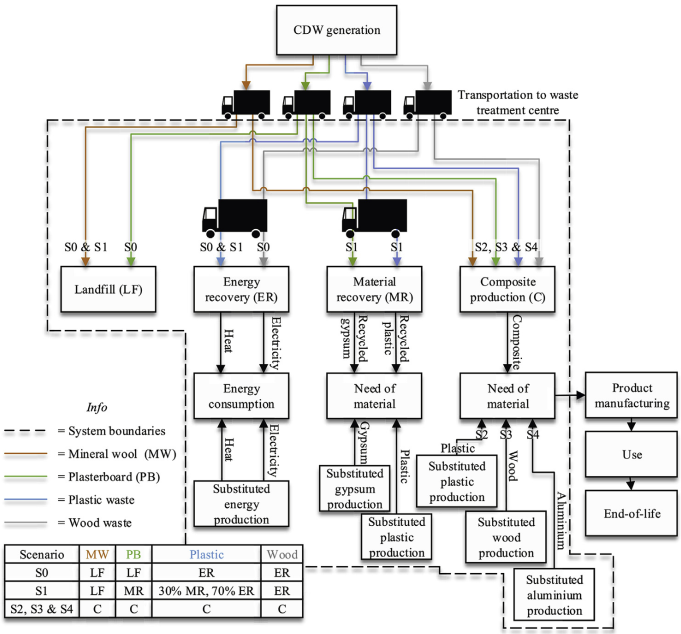

Fig. 4. System boundaries of the study.

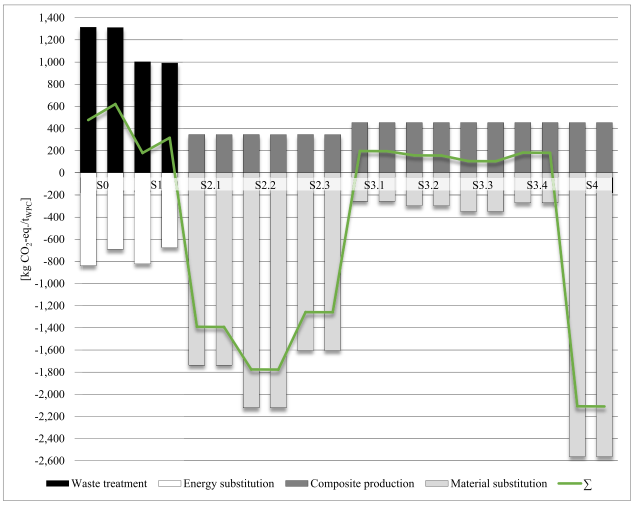

Fig. 5. Scenario contributions to climate change. The left bar in each scenario corresponds to Recipe 1, and the right bar to Recipe 2.

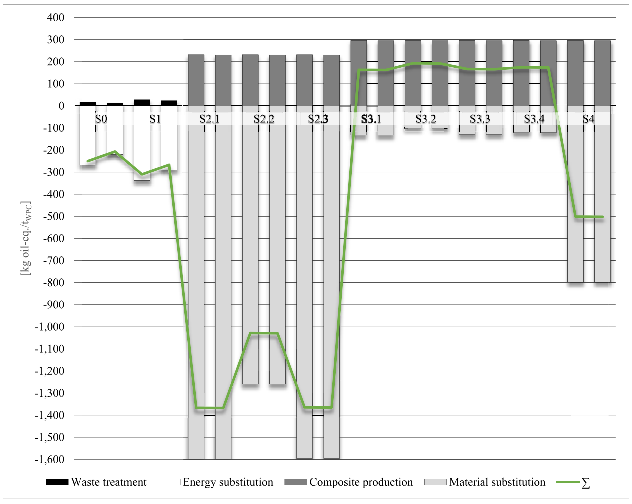

Fig. 6. Scenario contributions to fossil fuel depletion. The left bar in each scenario corresponds to Recipe 1, and the right bar to Recipe 2.

oil-eq., indicating that the avoided production of energy intensive aluminium saves more fossil resources than are used in WPC production.

In terms of the differences between the WPC recipes, the same phenomenon can be detected for fossil fuel depletion as was seen for climate change: Recipe 2 is preferable to Recipe 1 since it can achieve a larger reduction in fossil fuel use. Only a small difference was identified between Scenarios 0 and 1 in this impact category: the contribution in both is between −200 and −300kg oil-eq. Both scenarios save fossil resources due to the electricity and district heat production that is avoided in Finland. See Supplementary material E for further information on the results of the study.

3.2. Sensitivity analysis

This study assumes that the energy produced by the CDW wood and plastic fractions in Scenarios 0 and 1 would substitute the average electricity and district heat production in Finland. However, energy produced by wood and plastic waste can also substitute other types of energy production (marginal or local energy production). Therefore, it is relevant to examine how the climate change and fossil depletion results would change if the type of substituted energy production were to change. As the geographical location for the study is Finland, the Finnish electricity grid mix and district heat mix serve as the baseline energy productions in Scenarios 0 and 1. Fig. 7 presents the result if the substituted energy source were, instead, biomass, hard coal, peat or natural gas. All of these options are regionally-relevant energy sources in Finland. Regardless of the substituted energy source, the average recipe for a composite that substitutes HDPE (Scenario 2.3) and aluminium profile (Scenario 4) is always better than the baseline scenario (Scenario 0) or advanced waste treatment for plastics and plasterboard (Scenario 1). Fig. 7 only demonstrates this for HDPE, but the same principle applies to other plastics as well. The biggest reduction in emissions contributing to climate change occurs when energy produced with biomass is substituted. In this case, the scenario in which plywood is assumed to be the substituted material (S3.1) also results in emission reductions. In terms of fossil depletion, Scenario 3.1 always consumes more fossil resources, regardless of the substituted energy source in Scenarios 0 and 1.

As pointed out previously in this paper, due to the different mechanical and physical properties of WPCs and conventional materials, WPCs might not substitute conventional materials in a mass-based ratio of 1:1, as assumed in the results presented above. Therefore, the sensitivity analysis investigates the influence of the material substitution rate on the results. Fig. 8 illustrates the impact of the material substitution rate on climate change and fossil fuel depletion. In this figure, the results of Scenarios 2.3, 3.1 and 4 are presented with varying material substitution rates: starting from a 0% substitution rate (1,000 kg of WPC substitutes 0 kg of virgin material), and ending with a 100% substitution rate (1,000 kg of WPC substitutes 1,000 kg of virgin material). The results are presented aligned with the results of Scenarios 0 and 1 to identify the break-even points at which CDW fractions should be used in WPC production rather than being treated with conventional methods. The results are presented as averages of Recipes 1 and 2 since no major differences were identified between the two that would influence the main findings of the sensitivity analysis.

The results reveal that even if the produced WPCs substituted no virgin materials at all (0% substitution rate), the use of CDW fractions as raw materials for WPC production still decreases the impact on climate change when compared to the baseline scenario, Scenario 0. At the same time, in order to decrease the impact on climate change compared to Scenario 1, when producing WPCs from CDW some substitution of virgin materials must occur. For HDPE and aluminium (Scenarios 2.3 and 4), respective material substitution rates of at least 6% and 8% are needed to decrease the contribution to climate change compared to Scenario 1. For plywood, a significantly higher substitution rate of 80% is needed. When examining the fossil depletion results in Fig. 8, higher material substitution rates are needed for WPC production to compete with both Scenarios 0 and 1. HDPE (Scenario 2.3) requires the lowest material substitution rate (29–33%) depending on the scenario compared. For aluminium, a material substitution rate of

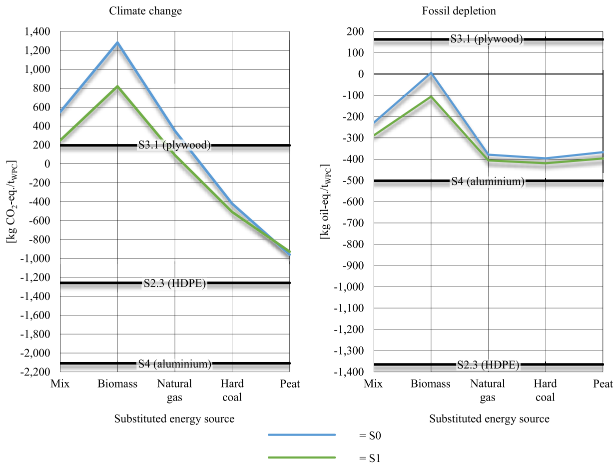

Fig. 7. The impact of the choice of substituted energy on climate change and fossil depletion results.

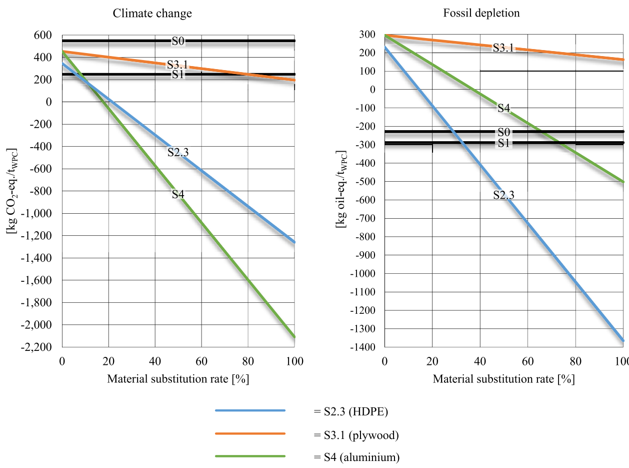

Fig. 8. The impact of the material substitution rate (0–100%) on climate change and fossil depletion.

66–73% is required. If plywood were to be substituted with WPC, it would not be reasonable in terms of fossil depletion, as, in practice, the requisite material substitution rate is unachievable.

3.3. Discussion

WPC production from CDW can be considered an intermediate step between landfill disposal or incineration and mono-material recovery. Therefore, waste materials should be primarily used as raw materials for WPCs if their mono-material recovery is not technically or economically feasible and they would otherwise be used as energy or sent to landfill. This study demonstrated the environmental impact reduction advantages that WPC production from CDW has over conventional waste treatment activities such as incineration and landfilling. The emissions reductions are highest when the produced WPC substitutes plastics or aluminium.

The findings of previous literature (Sommerhuber et al., 2017; Väntsi and Kärki, 2015) correspond with the findings of this study, confirming the hypothesis that, in terms of climate change and fossil depletion, waste-derived WPCs are more environmentally favourable than WPCs made of virgin materials. In this study, the environmental impacts of WPC production were 0.4–0.5 kg CO_2-eq./kg_{WPC} (climate change) and 0.2–0.3 kg oil/kg_{WPC} (fossil depletion). In the study by Sommerhuber et al. (2017), the environmental impacts of WPC production were approximately 0.8 kg CO_2-eq./kg_{WPC} and 0.2 kg oil/kg_{WPC} when recycled wood and plastic were used as raw materials. In the same study, the environmental impacts of WPC production were notably higher when virgin wood and plastic were used as raw materials: 1.7–2.2kgCO_2-eq./t_{WPC} and 0.9–1.3 kg oil/kg_{WPC}. When comparing the environmental impacts of CDW-derived WPCs and virgin materials such as plastic, this study demonstrated that WPCs are more environmentally favourable than plastic and aluminium, but less favourable than wood. Also taking into account the avoided emissions of CDW management, for example incineration and landfilling, it is environmentally favourable to produce WPCs from CDW even if the produced WPCs substitute wood, from the perspective of climate change.

This study examined the environmental impacts of using CDW in WPC production. As such, some considerations are beyond the scope of the work; these include the quality (e.g. possible contaminants) and availability of raw materials, demand for the produced WPC, the use and end-of-life phases for WPC-derived products, the physical and mechanical properties of different WPC types, comparisons between WPCs and other materials (e.g. wood and plastic) in terms of mechanical and physical properties, and the optimisation of the manufacturing process. This leaves room for further research. The end-of-life phase for WPCs is particularly interesting. WPCs made of CDW are recyclable, but only in the same manufacturing process. Since the plants are not yet common, material recovery for end-of-life WPCs is limited. Therefore, one has to wonder whether this technology will only allow us to lengthen the life cycle of these materials by one cycle or whether, with a sophisticated takeback mechanism, it could provide a method for moving towards a circular economy.

How can WPC production assist in reaching the 70% material recovery rate for CDW in Finland? According to Dahlbo et al. (2015), it is unlikely that the material recovery target will be reached by 2020 and this would require major changes in the sorting, separation and recovery processes within the CDW management system. Wood has been identified as a critical CDW fraction for increasing material recovery because, in Finland, it composes a high proportion of CDW and is currently incinerated in most cases. Wood comprises 720,000 of the two million tonnes of non-hazardous CDW generated annually in Finland (Dahlbo et al., 2015). Material recovery techniques for wood are thus a key requirement and WPC production has been proposed as a possible solution. The annual capacity of a large-scale WPC production plant would be approximately 20,000 tonnes (Grand View Research, 2018). If such a production plant existed in Finland, the CDW material recovery rate would increase by 1%-unit with the assumption that all raw materials were CDW. Therefore, approximately 10 WPC production plants would be needed to increase the material recovery rate from the current 60% to 70%. In light of these numbers, it is evident that WPC production cannot be regarded as a sole solution for meeting the material recovery target, rather as a single solution among other material recovery techniques and methods. It is important to remember that, due to the aforementioned limitations for end-of-life WPCs, mono-material recovery should be prioritised over WPC production.

This study broadens the literature on the environmental impacts of WPCs: it has assessed WPC production in terms of its environmental impacts, as a material recovery method for CDW and as part of the entire CDW waste management system. Previously published literature (Bolin and Smith, 2011; Feifel et al., 2015; Sommerhuber et al., 2017; Väntsi and Kärki, 2015) assessed the environmental impacts of WPCs from a product perspective without considering WPC production as a part of a CDW management system. It can be concluded that, in terms of climate change and fossil depletion, WPC production is an advisable treatment method for CDW when the produced WPC substitutes plastic or aluminium in the final application of the material.

4. Conclusions

The European Commission's ambitious material recovery target for CDW (requiring a 70% material recovery rate by 2020) has placed pressure on member states, including Finland, to increase their CDW material recovery. This study assessed the environmental impacts of WPC production, a novel and emerging material recovery option for CDW. The study examined the Finnish context and focused on the environmental impacts of using CDW in WPC production rather than treating CDW fractions with conventional waste treatment methods such as landfill disposal and incineration.

The results demonstrated that utilising CDW in WPC production can decrease the environmental impacts of CDW management. Significant environmental benefits can be achieved when the produced WPC substitutes virgin material whose production consumes fossil resources and contributes to climate change (i.e. plastic and aluminium). Conversely, it is not environmentally favourable to substitute wood with WPCs because the production of wood materials has lower environmental impacts than WPC production. Since the physical and mechanical properties of WPCs are different to those of plastic and aluminium, WPCs cannot necessarily substitute them in a mass-based ratio of 1:1. Therefore, the study determined the minimum substitution rates required to reach the break-even point for environmental impacts. For instance, the climate change impact of WPC production is lower than that of the advanced waste management scenario when the mass-based substitution rate is at least 6% for plastic and 8% for aluminium.

In this study, and in general, WPC production is regarded as an intermediate step between mono-material recovery (recycling CDW fractions individually) and conventional waste treatment methods (landfill disposal and incineration). Therefore, the scenarios did not directly compare WPC production with mono-material recovery. Rather, WPC production offers a material recovery option for CDW fractions that are currently landfilled or recovered as energy. This study provides a foundation for further research into the environmental impacts of waste-derived composites on a product level, also taking into account the use and end-of-life phases of WPCs. In addition to environmental impacts, it is important to investigate the economic and social impacts of the WPC product across its whole life cycle and, thus, to gain better insight into its overall sustainability.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2019.03.348.

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