Life cycle based approaches in assessing waste management options in the Finnish context

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Abstract

Amounts of waste are growing, non-renewable natural resources are becoming scarce and renewable resources are being over-consumed. To tackle these problems and to increase its economic competitiveness Europe has implemented waste strategies and the waste hierarchy to minimise waste generation and maximise waste recovery. Deviation from the hierarchy is acceptable if needed to reach the best environmental outcome. Life cycle (LC) based assessments are required to find environmentally and economically the most sustainable end-of-life and waste management (WM) solutions. In this study, Finnish waste flows and WM systems including waste or products from the forest sector were assessed with life cycle assessments (LCA), social life cycle costing (SLCC), environmental life cycle costing (ELCC) and material flow analyses (MFA). The aim was to evaluate whether results from LC-based environmental and economic assessments support the waste hierarchy order Finland and to find out whether the combination of approaches provides added value for decision-making. Normalisation as a tool to improve the understanding of life cycle impact assessment (LCIA) results was also studied. The results from the case-specific environmental and economic assessments did not clearly support the waste hierarchy order of WM options. Landfilling performed worse than recycling or incineration with energy recovery. In contrast, the use of recyclable packaging performed better than reusable packaging. Energy recovery of newspaper waste produced lower overall environmental impacts than recycling, the results varied in individual impact categories, however. In addition, the SLCC showed that due to its higher environmental costs, energy recovery was more expensive than recycling. The combined use of LC-based approaches revealed potentials for improving the performance of WM systems, e.g. the combination of MFA, LCA and ELCC identified the potential of different fractions for improving the overall performance of the WM system of construction and demolition waste (C&DW). Additionally, the joint assessment indicated that achieving the 70% recycling target for C&DW in 2020 requires major improvements in the system. The results indicated that regionally differing recycling targets may be required within Europe. The understanding of LCIA results was to some extent improved by including information on the LC approaches in the communication material for LCA non-practitioners. Also the use of different normalisation reference values increased had impact on the understandability. The impacts avoided due to the material or energy recovery of waste are one of the key factors in WM modelling, but changes in the operational environment and uncertainty about the future complicate this modelling and limit the applicability of the results. Changes in the energy sector and in the use of secondary materials might have conflicting effects on the environmental sustainability of WM and systemic assessments are required.

Keywords  life cycle assessment, life cycle costing, material flow analysis, normalisation, waste hierarchy, municipal solid waste, construction and demolition waste, wood
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Helsinki, 28 April 2018
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List of Abbreviations

CCB  Corrugated cardboard

C&DW  Construction and demolition waste (C&DW) arises from activities such as the construction and total or partial demolition of buildings and civil infrastructure, road planning and maintenance. Different definitions are applied throughout the EU. C&DW can consist of numerous materials, including concrete, bricks, gypsum, wood, glass, metals, plastic, solvents, asbestos and excavated soil. In this study rocks, gravel, soil or other earth-based materials were excluded from C&DW, since they are also excluded from the 70% recycling target given for C&DW in the Waste Framework Directive (2008/98/EC).

ELCC  Environmental life cycle costing (ELCC) summarises all costs associated with the life cycle of a product that are directly covered by one or more of the actors in that life cycle. These costs must relate to real money flows. Environmental impacts are not monetised, but ELCC is complemented with a life cycle assessment (LCA), using equivalent system boundaries and functional units. (Hunkeler et al., 2008).

FU  A functional unit (FU) provides a reference to which all the studied systems inputs and outputs are related. It gives a quantitative description of the primary function or service that the analysed system provides. (European Commission, 2011; ISO, 2006a; Laurent et al., 2014b).

GHG  Greenhouse gas

HDPE  High-density polyethylene

LCA  A life cycle assessment (LCA) is a quantitative tool that helps in implementing life cycle thinking (LCT). An LCA is used for analysing and assessing environmental impacts of a material, product or service during its entire life cycle. The full life cycle of a product is taken into account.

LCC  Conventional life cycle costing (LCC) is based on purely economic evaluation, considering various stages in the life cycle. However, LCC does not always consider the complete life cycle, for example end-of-life operations are not included. LCC generally includes conventional costs associated with a product that are borne directly by a given actor (Hunkeler et al., 2008).

LCI  A life cycle inventory (LCI) is the second component of a life cycle assessment in which an analysis is made of the environmental interventions associated with the processes required for that functional product unit.
LCIA  A life cycle impact assessment (LCIA) is the third component of a life cycle assessment in which the data gathered in the inventory analysis are interpreted and assessed in terms of their environmental impact potential.

LCT  Life cycle thinking (LCT) means a qualitative or quantitative assessment of upstream supply chains, downstream processes, and the use of products in terms of environmental, social, and economic considerations. It is applied to analysing the performance of, and improvement options for, both goods and services. An LCT accounts for the full life cycle of a product or service.

MFA  Material flow analysis (MFA) is a tool to analyse the transformation, transportation, or storage of materials within a defined system. (Brunner and Rechberger, 2004).

MSW  Municipal solid waste (MSW) refers to waste generated in households and waste comparable to household waste generated in production, especially in the service industries. The general common feature of municipal waste is that it is generated in the consumption of final products in communities and is covered by municipal waste management systems. (Statistics Finland, 2015)

SFA  Substance flow analysis (SFA) is a specific type of MFA tool used for tracing the flow of a selected chemical (or group of substances) through a defined system.

SLCC  Social life cycle costing (SLCC) looks at the costs associated with the entire life cycle of a product or a service from the viewpoint of the whole society. Various cost items can be distinguished. The most conventional costs are direct costs, i.e. the costs of investments, labour, energy, and so on. SLCC also includes all external costs associated with environmental impacts of the product’s or service’s life cycle. The stakeholders of SLCC include governments and other public bodies not directly concerned with the product system. Therefore the costs do not include transfer payments, such as subsidies and taxes, because they are internal to the system. (Hunkeler et al., 2008; Paper II).

SWMS  Solid waste management system

WM  Waste management

WtE  Waste to energy
This doctoral dissertation consists of the present summary and of the following papers which are referred to in the text by their numerals:


Author’s Contribution

**Publication 1:** H. Dahlbo and M. Melanen planned the study. H. Dahlbo was responsible for writing the paper. J. Laukka and T. Myllymaa performed the life cycle inventories with guidance from S. Koskela. The other authors commented on the paper.

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**Publication 3:** H. Dahlbo and S. Koskela initiated and planned the paper. H. Dahlbo produced the LCIA and analyses on normalisation factors. H. Pihkola contributed by providing support on communication issues. The other authors provided comments on the paper. The conclusions were written jointly by H. Dahlbo and S. Koskela.

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1. Introduction

1.1 Development of waste management

In early human history, people generated little waste and reused and repaired what they could. As city populations grew, waste management (WM) systems became necessary. The first known European waste statute was given in Athens in 500 BC. Regulations required waste to be deposited at least one and a half kilometres outside the city limits. (Barbalace, 2003; Nygård, 2016).

With growing cities, waste began piling up and threatening human health due to the presence of rats and other pests as well contaminated water supplies that led to the transmission of diseases. During the 14th century, orders and restrictions began to emerge for taking better care of wastes. The Black Death that killed approximately 25 million people in Europe in the mid-14th century, was one factor that affected European WM and sanitation. At least partly due to this, in 1388, the English parliament forbade waste dispersal in public waterways and ditches. In 1400 the garbage piles outside Paris gates were been reported to have been so high that they affected the city’s defences (Barbalace, 2003; Nygård, 2016).

The oldest street sanitation statutes in Finland were given in the 13th century in Sweden. They embodied orders on the property owners’ responsibility for keeping the streets clean and sanctions for dereliction of these duties. (Nygård, 2016).

More organised waste collection and disposal systems aiming to maintain public health were launched in the 18th century, when the Industrial Revolution gave rise to rapid growth of both production and amounts of waste (Barbalace, 2003; Nygård, 2016). Since then, WM has developed towards material and resource management, which takes into consideration environmental aspects in addition to health and safety aspects. Sophisticated technologies are now applied for the sorting, separating, processing, recovering and treating various types of waste and the valuable components it contains.

Due to growing the population and increasing consumption and production, the amount of waste is constantly increasing both in Europe and globally. Simultaneously, non-renewable natural resources are becoming scarce, while renewable resources are being over-consumed. Several attempts have been made and indicators developed to assess and predict the sustainability of the human use of global natural resources. For example, the Ecological Footprint measures the supply of nature (i.e., the planet’s biologically productive land areas including forests, pastures, cropland and fisheries), and the demand on nature (i.e., the productive area
required to provide the renewable resources humanity is using and to absorb its waste) (Global Footprint Network, 2016). According to the 2014 official Ecological Footprint assessments on the world and national level, humanity demanded the resources and services of 1.5 planets in 2010 (Mancini et al., 2016; WWF International et al., 2014).

In 2009 Rockström et al. proposed a new approach to global sustainability in which they defined planetary boundaries within which humanity could operate safely. They identified nine planetary boundaries and proposed quantifications for seven of them, namely climate change, ocean acidification, stratospheric ozone, the biogeochemical nitrogen (N) cycle and phosphorus (P) cycle, global freshwater use, land system change and the rate at which biological diversity is lost. The two additional planetary boundaries are chemical pollution and atmospheric aerosol loading. According to Rockström et al. (2009), transgressing one or more of the boundaries may be deleterious due to the risk of crossing thresholds that will trigger non-linear, abrupt environmental changes within continental- to planetary-scale systems. Rockström et al. (2009) estimated that humanity has already transgressed three planetary boundaries for: climate change, the rate of biodiversity loss, and changes to the global nitrogen cycle. In a more recent study, Steffen et al. (2015) concluded that anthropogenic perturbation levels of four of the Earth system processes (climate change, biosphere integrity (earlier biodiversity loss), biogeochemical flows, and land-system change) already now exceed the proposed planetary boundary.

To fight against the current, unacceptable development Europe has implemented waste strategies focusing on minimising waste generation and maximising the recovery of waste. In July 1975 the Council Directive on waste (75/442/EEC) addressed the need for the nine member states to support appropriate actions for waste prevention, recycling and processing of waste, the extraction of raw materials and possibly of energy from waste and any other process for the re-use of waste. The first Community Strategy for Waste Management (SEC (89) 934 Final 1989) was based on this directive and established the hierarchical system for WM, under which waste prevention and minimisation have the highest priority followed by recycling and disposal. In 1996 the Community Strategy for Waste Management was published (COM (96) 399).

In the Sixth Environment Action Programme of the European Union (the EU’s ten-year environmental policy programme, 2002-2012), the European Commission defined the sustainable use of natural resources and management of wastes as one of four priority areas. This led to the development of two thematic strategies on resources and waste management, namely 1) the Thematic strategy on the prevention and recycling of waste (COM (2003) 301 final, COM (2005) 666 final) (Salhofer et al., 2008), and 2) the Thematic strategy on the sustainable use of natural resources (COM (2005) 670 final). The strict five-level priority order for WM was introduced in EU legislation in 2008 with the Waste Framework Directive (WFD, 2008/98/EC), which gives waste prevention and re-use (or preparing for re-use) the highest priority, recycling taking the third place and thermal treatment with energy recovery and disposal representing the least favourable options (Fig. 1).
According to the WFD, when applying the waste hierarchy, member states must take measures to promote the options that deliver the best overall environmental outcome. This may require specific waste streams to depart from the waste hierarchy where this is justified for reasons of technical feasibility, economic viability and environmental protection, among other things. (WFD 2008/98/EC).

Recently, van Ewijk and Stegemann (2016) challenged the waste hierarchy by concluding that in its current form the hierarchy is an insufficient foundation for waste and resource policy to achieve absolute reductions in material throughput. Suggested improvements include the adoption of a value-based conception of waste and related collection practices, more stringent and targeted policies towards the least desirable options such as landfill, the specification of WM targets based on dematerialisation ambitions, and the use of the waste hierarchy within a resource productivity-oriented framework.

The concept of the Circular Economy highlighted especially by the Ellen McArthur Foundation (2012; 2013; 2014) has during recent years strongly emphasised the need for waste prevention through business models that do not base their value creation on producing materials but rather on fulfilling the needs of customers by providing services. Such development requires system level innovations. However, while developing these, we continuously generate waste that needs sustainable solutions. The circular management of resources and waste has been emphasized both in the bio-economy strategy of the European Commission (2012) and in the EU Circular Economy Package (European Commission, 2017a) as a means to increase resource efficiency. Especially for biomass and bio-based waste streams, cascading utilisation by recovering residues and waste as materials for as many rounds as possible and finally recovering them as energy, has been raised as way to improve sustainability and extend biomass availability in a system (Vis et al., 2016).
1.2 Life cycle assessment and waste management

1.2.1 Environmental life cycle assessment as a support to waste policy

In order to assess the most sustainable solutions for WM and end-of-life options for products, methods are needed which enable the consideration of various impacts and the benefits from reuse, recycling and recovery. According to the WFD, assessment methods based on life cycle thinking (LCT) should be used to justify decisions that are not in accordance with the waste hierarchy (WFD, 2008/98/EC). LCT aims to take into account overall environmental impacts by considering the range of impacts occurring throughout the life time of a product.

LCT is applied in a variety of approaches, most strictly, however, in life cycle assessment (LCA). LCA is a standardised tool for quantifying the life-cycle impacts of a product or service by assessing the emissions, resources consumed and pressures on health and the environment that can be attributed to the subject of assessment (ISO, 2006a; 2006b). It takes into account the entire life cycle – from the extraction of natural resources through material processing, manufacturing, distribution to use, and finally the re-use, recycling, thermal treatment with energy recovery and the disposal of the remaining waste at the end-of-life phase. It is important to note that reducing the environmental impact of one life cycle phase may lead to a greater environmental impact in another phase. An apparent benefit of a WM option can therefore be levelled out if not thoroughly evaluated. LCA provides a comprehensive view of the processes and impacts involved in the product system to be assessed and prevents problem-shifting between system compartments.

In the thematic strategies developed by the EU on resources and WM (see Chapter 1.1) the life cycle approach was emphasised as an important part of the work for prevention and recycling of waste and the sustainable use of resources. With the help and opportunities offered by LCAs, in the recent decades, the basis for EU policy on waste and resource use has changed from focussing on the amounts of waste and resources and shifted towards the potential environmental impacts they cause.

An LCA based impact assessment has recently been carried out (European Commission, 2014; 2015) to back the European Commission proposal for reviewing recycling and other waste-related legislation in the EU Waste Framework Directive (2008/98/EC), the Landfill Directive (1999/31/EC) and the Packaging and Packaging Waste Directive (94/62/EC). The proposal was first introduced by the European Commission in July 2015 with the aim of helping turn Europe into a circular economy, and to boost recycling, secure access to raw materials and create jobs and economic growth. This was done by setting ambitious targets and adding key provisions on the instruments required to achieve and to monitor them (European Commission, 2014). The proposal was included in the Circular Economy Package, which was adopted in December 2015, and is being implemented according to the Circular Economy action plan (European Commission, 2017a).
1.2.2 Life cycle assessment methodology for evaluating waste hierarchy

The LCA methodology for WM studies has been developed to take into consideration the specific issues concerning WM. LCA widens the scope of analysis from the WM itself (waste connected processes) to the links with other sectors such as manufacturing, energy production and agriculture (Björklund et al., 2010). Thus, the quantification of benefits from waste recycling and recovery becomes possible and relevant. Some of the earliest documentation of LCA methodology for WM was provided by Clift et al. (2000) and Ekvall and Finnveden (2000). Recent LCA guidelines have been developed by the Directorate General Environment and the Joint research Centre (JRC) to support environmentally sound decision-making in WM. These guidelines are tailored to the needs of different target audiences and partly focus on specific waste streams, e.g. biowaste and construction and demolition waste (C&DW) (European Commission, 2011; Manfredi et al., 2011). Additionally, Laurent et al. (2014a) have presented recommendations on implementing LCA for solid waste management systems (SWMS) based on their review and findings from over 200 LCA studies performed on these systems.

The overall framework for LCA is standardised (ISO, 2006a; 2006b) and well established as a four-step process, which is initiated with a goal and scope definition, followed by a life cycle inventory (LCI), life cycle impact assessment (LCIA) and interpretation (Fig. 2). The assessment proceeds iteratively, returning when necessary, from one step to the preceding one/ones to adjust the model.

**Goal and scope definition**

The goal definition describes the purpose of the study and the decision process it provides support for. In LCAs for WM, for example, the goal may be related to the determination of whether recycling or thermal treatment with energy recovery is environmentally preferable for a certain waste fraction, or whether one energy recovery technique is preferable over an-
other or whether one flue gas cleaning technology generates lower environmental impacts from waste incineration than another (Hauschild and Barlaz, 2010).

During the scope definition, the object of the LCA, i.e. the exact system to be analysed, is identified and defined in detail. One key aim is to define the functional unit (FU) of the system. FU provides the reference to which all the system inputs and outputs are related. It gives a quantitative description of the primary function or service that the analysed system provides (European Commission, 2011; ISO, 2006a; Laurent et al., 2014b). The same FU shall be used for all alternative systems compared in the LCA. For the LCA of a WM system, the FU could be defined as, for example, the quantity of waste to be managed during a certain period of time (e.g. tonnes/year), composition of the waste, duration of the WM service or emission limits (Hauschild and Barlaz, 2010).

In their review on LCAs for SWMS, Laurent et al. (2014b) showed that most LCA studies used the management or treatment of 1 tonne of waste as the FU. The authors considered this insufficient since specification of the type of waste considered should be included in order to give a precise picture of the function of the system (Laurent et al., 2014b). Composition of waste may vary in different regions and locations due to e.g. housing and consumption patterns, different practices and efficiencies in source separation of waste, differences in collection systems, availability of composting and several other aspects. The waste composition affects the performance of WM, hence the composition and the practical implementation of the SWMS studied should be defined and described in the scope definition (Laurent et al., 2014b).

Also other, geographically varying factors affect the environmental performance of WM processes and systems. These factors include the natural conditions, such as climate that can greatly affect the possibilities of e.g. arranging various collection systems. Institutional factors such as legislation, e.g. on the quality of waste-derived materials, provide requirements for the processing of waste and this way affect the performance of WM. Economic factors such as existence and location of potential markets for waste or recovered products from waste will determine logistical requirements of the system. Technical factors such as availability of waste treatment facilities and infrastructure for waste transports are crucial for WM strategies. All these issues should be considered when defining the goal and scope of the LCA study on WM. (European Commission, 2011).

Boundaries of the studied product or service system need to be defined on the basis of the goals of the assessment. An overall system of consumption, production and WM includes numerous elements, different products, processes and flows. Product specific LCAs include the whole life cycle of a product from resources extraction, production to consumption and end-of-life of the product. From the point of view of WM both the upstream (preceding the WM) and downstream (use of waste-derived materials or products) processes can be included in these assessments. The goal of a product LCA can be to identify the product life cycle phases that most influence the overall product performance, or whether recycling or thermal treatment with energy recovery of the final product can improve the overall performance, among others.
When focusing on a WM system, i.e. the end-of-life phase, the starting point for modelling is usually waste generation, the point where products are discarded as waste. Waste comes into the system with zero-burden, assuming that the same kind of waste will be generated no matter how the WM is organised. This assumption is in compliance with the LCA definition which allows users to disregard parts that are identical between compared systems. However, if one of the systems would include a waste minimisation strategy that affects waste generation, the production and use phase of the products would need to be included in the model as well, making the model much more complex. (Björklund et al., 2010; Ekvall et al., 2007).

The foreground of a WM system covers the actual WM processes, i.e. collection, separation, processing, recovery and treatment, whereas the background system covers processes outside WM (Clift et al., 2000). WM systems most commonly provide several functions, such as managing waste and producing recycled materials. Recycled materials can be considered by-products of waste treatment, and a share of the overall impacts of the system needs to be allocated to them. However, the ISO standard recommends avoiding allocation whenever possible, e.g., by expanding the product system to include additional functions related to the by-products (ISO, 2006b). Thus, all interventions can be attributed to the product system and credits can be given for the production of by-products, assuming that their production replaces other ways of producing similar products, such as using virgin materials. Nevertheless, it is not always easy to determine what product or production method the recycled products replace and there may even be several correct choices for a certain case (Björklund et al., 2010). The review by Laurent et al. (2014b) discovered that 75% of the LCA studies on SWMSs applied system expansion in order to credit waste material or energy recovery. However, the reporting of the type of material or energy substituted and the substitution rate used was found defective (Laurent et al., 2014b). LCA modelling of a WM system usually ends at the point of substitution, hence downstream processes are not included in the assessment (Björklund et al., 2010; Clift et al., 2000; European Commission, 2011).

The change-oriented, or consequential LCI modelling aims at identifying the consequences of a decision in the foreground system on other processes and systems of the economy and builds the to-be-analysed system around these consequences. These consequences include processes that are assumed to be affected due to a specific decision. In contrast, attributional modelling depicts the system as it can be observed, linking single processes to the flow of matter, energy, and services (i.e. the existing supply-chain). (European Commission, 2010a). WM modelling sometimes combines these two types of modelling.

**Life cycle inventory (LCI)**

The inventory phase focuses on data collection and system modelling. Inventory analysis is the most laborious and important phase of LCA. The compilation of inventory data relates the inputs and outputs of different processes to the FU of the study (European Commission, 2011; Hauschild and Barlaz, 2010). Two types of inventory data are usually required for an LCA. Process specific primary data on the input materials, energy, water, chemicals, waste, wastewater and gaseous emissions are needed for the foreground system, whereas non-specific, generic, secondary data is sufficient for the background system (Clift et al., 2000).
In some cases conclusions can already be drawn on the basis of the results of the inventory phase (Seppälä, 2003). However, an LCIA (Fig. 2) is often needed for interpreting the results of the inventory. The consideration of environmental effects as a consequence of environmental interventions provides additional information which is not covered by the inventory step (Seppälä, 2003).

**Life cycle impact assessment (LCIA)**

An LCIA is typically divided into five phases: 1) selection of impact categories, category indicators and characterisation models, 2) classification of LCI results, 3) calculation of category indicator results, i.e. characterisation, 4) normalisation and 5) weighting. The first three phases are mandatory and two last phases are optional according to the ISO standard (ISO, 2006a; 2006b).

The LCIA is still under development and apart from the global impact categories, global warming and stratospheric ozone depletion, no consensus has been reached on how to model the impacts (Hauschild and Barlaz, 2010). Land use impacts and their assessment is one of the developing topics. Baan et al. (2012) developed a methodology for assessing the impacts on biodiversity, focusing on occupational impacts, quantified as biodiversity damage potential (BDP). According to Baan et al. (2012) the reported characterisation factors for BDP can be used to approximate biodiversity impacts in LCA studies not focusing on land management practices. For studies focusing on land management practices, more detailed and site-dependent assessments are required (Baan et al., 2012). Land use impacts may not be relevant in all assessments of an SWMS, but could be important when focusing on biomass use (Laurent et al., 2014b). In recent years the need for and definition of reference situations for the assessment of land use impacts has been discussed by and Cao et al. (2017) and Soimakallio et al. (2015), among others.

Even on the global warming impacts various issues are still under discussion, such as the inclusion of methane oxidation impacts into the assessment, which has been challenged by Muñoz and Schmidt (2016).

Two approaches, midpoint and endpoint modelling, exist in LCIA modelling. In the most common midpoint approach impacts are modelled at a midpoint level in the environmental mechanism between emissions and damage, but as close as possible to the areas of protection in the causality web (Goedkoop and Spriensma, 2001; Hauschild and Barlaz, 2010; Seppälä, 1999; 2003). Commonly assessed impact categories include climate change, ozone depletion, human toxicity, particulate matter, ionizing radiation, photochemical ozone formation, acidification, eutrophication, ecotoxicity, land use and resource depletion (European Commission, 2010a). In endpoint modeling the impacts are modelled all the way down to the areas of protection, such as human health, ecosystem quality, and resource depletion, using the best available environmental models (Hauschild and Barlaz, 2010; Steen, 1999a; 1999b). Most recent LCIA methods, such as the ReCiPe Mid/Endpoint methodology (Goedkoop et al., 2009; Wegener Sleeswijk et al., 2008) combine the two approaches and give the user options to select which one to use.
In the classification phase of midpoint modelling, the emissions and resources derived from an LCI (i.e. interventions) are assigned to impact categories according to their ability to contribute to the environmental problem each category describes. In the characterisation phase interventions are converted into impact category indicators using characterisation factors that reflect the pressure that one unit of emission or resource consumed generates in the context of each impact category. The result of this is the characterised impact profile of the product, i.e. its LCIA result. (European Commission, 2010b).

Characterisation is considered the last mandatory step of the LCIA due to its scientific character, but an LCIA can be continued with one or two optional steps, normalisation and weighting. Normalised LCIA results are generated to translate the impact category scores into more understandable and presentable figures and to place them in an adequate environmental context (e.g. Laurent et al., 2011; Pizzol et al., 2017; Wegener Sleeswijk et al., 2008). Normalisation is performed by dividing the LCIA results by the normalisation basis separately for each impact category. Regionally defined reference values are most commonly used as the basis for normalisation (Laurent et al., 2011; Lautier et al., 2010; Pizzol et al., 2017; Wegener Sleeswijk et al., 2008). However, the normalisation basis can be chosen based of various dimensions, such as system bases, spatial or temporal scaling or even regional consumption (Breedveld et al., 1999). The normalisation reference needs to be complete in respect to the analysed system (Pizzol et al., 2017). Biased results can be obtained if a substance which is part of the LCI of a product system and a main contributor to the characterised results of a given impact is not part of the LCI used to calculate the normalisation reference. This would mean the normalisation reference would be underestimated and the normalised results would be overestimated (Heijungs et al., 2007; Pizzol et al., 2017).

Normalised LCIA results of the different impact categories show to which impacts the analysed system mainly contributes. The relevance of the impact categories relative to each other cannot be judged on the basis of these results. To directly compare or sum up results across impact categories, weighting is needed. In weighting, the normalised LCIA results for the different impact categories are each multiplied with a weighting factor expressing the relative importance of the category. Weighting factors can be generated with several methods, which Huppes and van Oers (2011) classify into non-monetised methods, such as panel and distance to target method, and monetised methods, such as the damage prevention cost method. Pizzol et al. (2017) provide some recommendations on the generation of weighting factors, such as suggesting that panels comprising affected stakeholders should be preferred instead of expert panels unless the relevant stakeholders delegate the task to other experts or representatives. For monetary valuation methods, Pizzol et al. (2017) recommend using observed preferences (market prices) whenever possible. The normalised and weighted LCIA results can subsequently be added up across all impact categories or areas of protection to create one single score for the environmental impact of a product or scenario (European Commission, 2010b).

Interpretation

The interpretation of LCA results is performed against the goals of the assessment and within the restrictions arising from the scope of the study and the assumptions made during the
modelling process. The interpretation includes identifying significant issues that have the largest affect or relevance to the overall environmental impacts. Findings in the impact assessment and sensitivity and contribution analysis, which are performed as part of the interpretation, help to identify the key processes, elementary flows, impact categories, modelling choices and assumptions of the system. This way the reliability of the final results can be assessed. (European Commission, 2010a; European Commission, 2011; Hauschild and Barlaz, 2010).

Due to the aggregation of emissions over time and space in the inventory phase, the impacts calculated are the sum of contributions from emissions that may be released over a period of several years in different locations. Thus, it is difficult to interpret these impacts in terms of effects on the environment and instead they should be considered as environmental sustainability indicators that can be used for process optimisation (Hauschild and Barlaz, 2010).

In WM LCAs the assumptions made concerning processes which have been avoided often determine the overall results. In the interpretation phase it is therefore important to consider the effects of different alternatives on the processes avoided.

The final phase of the interpretation is drawing of conclusions and identification of limitations of the LCA, and development of recommendations by integrating the outcome of the elements of the interpretation phase, and summarising the main findings from the earlier phases of the LCA. This is done in accordance with the goal definition and the intended applications of the results. Typically, the conclusions state whether the questions that were posed in the formulation of the goal definition can be answered by the LCA, i.e. whether significant differences exist between alternatives and what role the issues of the sensitivity analysis play regarding such differences. The limitations of the study within the given goal and scope of the LCA study must be reported. These can include items such as the incompleteness of elementary flows with relevance to impact categories, the limited time-representativeness, or pre-selection of climate change impacts only for carbon footprint studies. (European Commission, 2010a; European Commission, 2011).

1.2.3 Life cycle assessment studies on waste management

LCAs have been applied for assessments of WM most commonly to do with municipal solid waste (MSW) management since from the early 1990s (Björklund et al., 2010; Laurent et al., 2014a). The different levels of waste hierarchy have been compared in numerous studies on various WM systems and parts of it, as well as on different waste fractions and recovery technologies. However, the first priority, waste prevention, has seldom been assessed in an LCA. Yet some examples can be mentioned, such as the study by Cleary (2014), who assessed the life cycle impacts of five types of waste prevention measures, and the study by Arushanyan et al. (2017), who included waste prevention in their overall model of the Swedish WM system.

LCAs of products with different end-of-life options have been performed to challenge the second priority level of the waste hierarchy, reuse (or preparation for reuse), and in product specific studies such as the UK Environment Agency (2008) study on nappies, Lighthart and Ansems (2007) study on drinking systems, Raugei et al.’s (2009) on drums for chemical transport, as well as Sørensen and Wenzel’s (2014) study on bedpans and Unger and Landis’
Numerous LCA studies have compared the three lowest levels of the waste hierarchy, namely recycling, thermal treatment with energy recovery and landfilling in different WM systems. Several meta-analyses have been performed to find out whether the results can be generalised for decision support. These meta-analyses include reviews by Bernstad and la Cour Jansen (2012) on food waste LCAs, Björklund and Finnveden (2005) on LCAs for various fractions of MSW, Cleary (2009) on mixed waste LCAs, Laurent et al. (2014a; 2014b) on all LCA studies on plastic, paper, organic and mixed waste fractions of MSW, Lazarevic et al. (2010) on plastic waste, Michaud et al. (2010) on LCAs for various fractions of MSW, Morris et al. (2013) on source separated organics and Villanueva and Wenzel (2007) on paper and cardboard waste. In addition, Bassi et al. (2017) compared the environmental performance of household WM in seven European countries. Most LCAs on SWMSs have been done in Europe, where the number of studies grew considerably after the acceptance of the EU Waste Framework Directive in 2008 (Laurent et al., 2014a). However, in recent years LCAs on WM issues in countries such as China have been published increasingly. These include studies by Havukainen et al. (2017) and Liu et al. (2017) on MSW management, among others.

Looking more specifically at paper WM, Villanueva and Wenzel (2007) conducted a review of nine paper and cardboard LCA studies with altogether 73 scenarios with different management options for waste paper. They found the outcome of individual LCA studies significantly depended on fifteen key assumptions including assumptions made regarding the use of wood saved by recycling, the efficiency of energy recovery technology modelled and the type of energy production substituted with waste based energy (Villanueva and Wenzel, 2007). Merrild et al. (2012; 2008) further assessed the impacts of different choices such as system boundary choices, and the significance of technical data when modelling the environmental impact of recycling and incineration of waste paper. Kim and Song (2014) applied the concept of temporary biomass carbon storage in their comparison of wood waste recycling in particle boards to energy recovery in combined heat and power production (CHP) plants.

Wood waste is currently mostly used for energy production. The environmental benefits of the cascading utilisation of wood compared to the primary use of wood and non-cascading uses of secondary wood have been assessed by Höglmeier et al. (2014; 2015). Additionally, the contribution of wood polymer composites (WPC) to the cascading utilisation of wood has been assessed by Teuber et al. (2016). The increased use of sawmill residues and other wood waste in WPCs instead of using the waste for energy purposes could increase biomass cascading and utilisation (Vis et al., 2016).

In recent years, concern has grown about the sustainable management of the increasing amounts of C&DW, and consequently the number of published LCA studies on C&DW has grown (Bovea and Powell, 2016; Butera et al., 2015; Diyamandoglu and Fortuna, 2015; Hossain et al., 2017; Martinez et al., 2013; Mercante et al., 2012; Ortiz et al., 2010). Hossain et al. (2017) applied an LCA to evaluate the performance of C&DW management systems which were equipped with different sorting technologies. Butera et al. (2015) studied mineral, source separated C&DW which was either utilised in road construction as a substitute for
natural aggregates, or landfilled. Bovea and Powell (2016) performed a review of studies applying LCA to C&DW management systems, but also included studies on the performance of recycling plants for different C&DW fractions and comparisons of natural materials versus materials manufactured from recycled C&DW. However, most studies analysed the whole life cycle of buildings or construction elements, including the end-of-life stage, and not the C&DW management as such (Bovea and Powell, 2016).

In Finland LCAs on WM have been carried out and increasingly published over the past 10 years (e.g., FCG, 2010; Kiviranta and Tanskanen, 2009; Manninen et al., 2016; Moliis et al., 2012; Myllymaa et al., 2008; Niskanen, 2012; Sundström et al., 2014). The increase is caused due to changes in the operational environment of WM. Before 2007, only one incineration plant for MSW existed in Finland, and most of the mixed MSW was landfilled. Source separation and material recovery options were available for biowaste, paper, cardboard, metal and glass. In some regions cardboard, paper and plastics were collected as energy waste to be co-combusted as REF (recovered fuel) together with other fuels. From 2007 onwards, eight new waste-to-energy power plants for mixed MSW have started operation and this has helped in diverting waste away from landfills. However, at the same time, the recycling rate for MSW has remained low, being now at the level of 41% (Laaksonen et al., 2017). The introduction of increased capacity and further possibilities to incinerate mixed MSW have led to questions concerning the benefits of material recovery of combustible fractions of MSW and have hence increased the need for LCA studies.

Several LCAs have focused on biowaste, and compared the environmental impacts of some or all of the following alternatives: composting, anaerobic digestion, thermal treatment with energy recovery and biofuel production (FCG, 2010; Manninen et al., 2016; Myllymaa et al., 2008; Sundström et al., 2014; Virtavuori, 2009). The majority of the Finnish LCAs on biowaste have only assessed climate change impacts (FCG, 2010; Virtavuori, 2009). However, Sundström et al. (2014) also addressed eutrophication impacts.

The separate collection and recovery of different packaging material waste flows has been assessed in Finland primarily from the point of view of the potential and environmental impacts of expanding the separate collection to small properties and sparsely populated areas (Kiviranta and Tanskanen, 2009; Moliis et al., 2012; Niskanen, 2017).

Finland has long traditions in recycling paper waste to be used as a resource for paper production. In contrast, waste wood has been considered preliminarily a source of energy. This arises from the fact that virgin wood resources in the country are vast and the need for recovering waste wood as a material has not been confronted in practice. In recent years recycling options for wood waste have been of interest due to the wide use of wood packaging and the high share of wood in, for example, C&DW in Finland. Manninen et al. (2015) compared the environmental impact of energy recovery of wood waste and the current applications for recycled wood waste, namely composites and particle boards. The use of wood residues or wastes in WPCs has also been studied by Judl et al. (2016) and Turku et al. (2017), for example.
In recent years scientific discussion concerning the role of forests and forestry on climate change has been vivid and is also connected to the management of wood-based products and waste. The issue of wood utilisation and the environmental benefits from it is complex, as was shown by Soimakallio et al. (2016), who recently analysed climate change mitigation challenges for the Finnish wood utilisation. However, thermal treatment with energy recovery was considered the end-of-life option for paper instead of recycling, based on the fact that paper fibres can only be recycled a limited number of times (Soimakallio et al., 2016).

1.3 Broadening life cycle assessment with cost and material assessments

An LCA can be used for decision-support both in industry, and by municipalities, policymakers and regulatory authorities. The information provided by carrying out an LCA must often be combined with other types of information, environmental and economic, for example, to reach a fully informed decision (Hauschild and Barlaz, 2010). To strengthen the LCA as a tool and eventually increase its usefulness in supporting sustainable decisions, it has been argued that there is a need to expand the ISO LCA framework through integration and linking it to other concepts and methods (Ekvall et al., 2007; Jeswani et al., 2010; Kloepffer, 2008).

Finnveden et al. (2007) performed an overview of methods and approaches that could be used for supporting WM decisions at different levels in society. The tools included approaches for both environmental and economic assessments such as the LCA, as well as life cycle costing (LCC), environmental impact assessments (EIA), strategic environmental assessments (SEA), cost-benefit analyses (CBA), risk assessments, material flow accounting or analyses (MFA), substance flow analyses (SFA) and environmental management systems (EMS).

In addition, Jeswani et al. (2010) performed an overview of some approaches and methods available for deepening and broadening the LCA to also cover other sustainability issues than environmental sustainability. The methods are classified according to their focus or level of assessment (micro, meso or macro) and according to the sustainability dimensions they are able to cover (Table 1). Jeswani et al. (2010) concluded that there is no “one size fits all” solution to integrating different life cycle related concepts for better sustainability assessments. Instead options for broadening the LCA depend on the field of application and the users, as well as their requirements and the goal and scope of an investigation.
Introduction

Table 1. Methods to assess environmental, economic and social sustainability (Jeswani et al., 2010).

<table>
<thead>
<tr>
<th>Assessment method (assessment framework)</th>
<th>Focus/level</th>
<th>Sustainability dimension</th>
</tr>
</thead>
<tbody>
<tr>
<td>Procedural method</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Environmental Impact Assessment (EIA)</td>
<td>Project (micro)</td>
<td>Environmental and social</td>
</tr>
<tr>
<td>Strategic Environmental Assessment (SEA)</td>
<td>Policy (meso/macro)</td>
<td>Environmental and social</td>
</tr>
<tr>
<td>Sustainability Assessment (SA)</td>
<td>Policy (macro)</td>
<td>Environmental, economic and social</td>
</tr>
<tr>
<td>Multi-Criteria Decision Analysis (MCDA)</td>
<td>Policy/project (micro/meso/macro)</td>
<td>Decision-support tool which can include environmental, economic and social dimensions</td>
</tr>
<tr>
<td>Analytical method</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Material Flow Analysis (MFA)</td>
<td>Policy, plan (macro)</td>
<td>Environment (natural resources)</td>
</tr>
<tr>
<td>Substance Flow Analysis (SFA)</td>
<td>Specific substance (macro)</td>
<td>Environment (natural resources)</td>
</tr>
<tr>
<td>Energy/Exergy Analysis (EA)</td>
<td>Process, Product/service (micro)</td>
<td>Environment (natural resources)</td>
</tr>
<tr>
<td>Environmental Extended Input Output Analysis (EIOA)/Hybrid LCA</td>
<td>Policy, Product/service (meso/macro)</td>
<td>Environment</td>
</tr>
<tr>
<td>Risk Analysis (RA/ERA/HERA)</td>
<td>Chemicals/Projects (micro)</td>
<td>Environmental and health impacts</td>
</tr>
<tr>
<td>Life Cycle Costing (LCC)</td>
<td>Product/Service (micro)</td>
<td>Economics</td>
</tr>
<tr>
<td>Cost-Benefit Analysis (CBA)</td>
<td>Policy/Project (micro/meso/macro)</td>
<td>Economics (includes cost of environmental and social impacts)</td>
</tr>
<tr>
<td>Eco-Efficiency (EE) Analysis</td>
<td>Product/Service (micro)</td>
<td>Integration of environmental and economic</td>
</tr>
<tr>
<td>Social Life Cycle Assessment (SLCA)</td>
<td>Product (micro)</td>
<td>Social</td>
</tr>
</tbody>
</table>

LCC has gained support for its use as a complementary tool to LCA for sustainability assessments for example by Kloepffer (2008), who proposed to complement LCAs with LCC and a social life cycle assessment (SLCA) to generate a life cycle sustainability assessment (LCSA). The system boundaries of the three assessments need to be consistent and the best solution would be the use of one identical LCI for all three components. Economic issues often play a decisive role in decision-making, hence it is important to include economic aspects in assessments along with environmental aspects. In addition, a social impact assessment could be considered important, at least in less developed countries. According to Kloepffer (2008) it seems also that the societal LCI would be much more demanding in regard to regional resolution than LCAs or LCC.

For environmental life cycle costing (ELCC), a code of practice has been developed (Swarr, 2011). ELCC summarises all costs associated with the life cycle of a product that are directly covered by one or more of the actors in the life cycle (e.g., supplier, producer, user/consumer and those involved at the end-of-life). These costs must relate to real money flows (Hunkeler et al., 2008). ELCC is performed on a basis analogous to LCA, including the definition of an FU and similar system boundaries in both LCAs and ELCC. The costs are complemented with an LCA, using equivalent system boundaries and FUs. (Hunkeler et al., 2008).

Social life cycle costing (SLCC) focuses on the costs associated with the life cycle of a product or a service defined from the viewpoint of the whole society. Furthermore, the stakeholders of SLCC include governments and other public bodies not directly concerned with the product system. The most conventional costs are direct costs, such as the costs of investments, labour and energy. SLCC also includes all external costs associated with the environmental impacts throughout the product’s or service’s life cycle. The costs do not include subsidies or taxes, because they are internal to the system and are levelled out in the overall results. (Hunkeler et al., 2008; Paper II).
Developing a consistent data set for LCC can be challenging because the data comes from many different sources and may be business sensitive (Swarr et al., 2011). Recently, Martinez-Sanchez et al. (2015) developed a consistent cost model for LCC including cost items for all key technologies within modern WM systems. Cost items were classified as: (1) budget costs, (2) transfers and (3) externality costs. The cost model allows calculation of conventional LCC, ELCC and SLCC. (Martinez-Sanchez et al., 2015).

A number of studies have assessed both environmental and economic impacts of WM by combining LCAs with LCC (Carlsson Reich, 2005), SLCC, CBA (Ahamed et al., 2016) or other economic assessments such as input-output analyses (Ferrão et al., 2014). However, Martinez-Sanchez et al. (2015) found very few examples in the literature of really combining economic and environmental aspects in assessing WM systems.

In Eriksson et al. (2005) different combinations of incineration, materials recycling of separated plastic and cardboard containers, and biological treatment (anaerobic digestion and composting) of biodegradable waste, were studied and compared to landfilling. The evaluation covered the use of energy resources, the environmental impacts and the financial and environmental costs.

Cleary (2009) performed a review of LCA studies on mixed waste treatment and found that eight of the twenty LCAs incorporated financial LCCs focusing on waste treatment activities. Five of the twenty studies undertook an environmental LCC, attributing economic values to the environmental impacts of each waste treatment scenario (Cleary, 2009).

Packaging waste has been in focus of several studies combining environmental and economic assessments. In Finland, Moliis et al. (2012) studied the environmental impacts and costs of a separate collection system for packaging waste in sparsely populated Northern Finland. Ferreira et al. (2014a; 2014b; 2017a; 2017b) assessed the costs and benefits of recycling packaging waste from the perspective of local authorities in Portugal, Belgium and Italy. Pires et al. (2017) assessed the environmental impacts and costs to the WM company of packaging waste collection systems in Portugal. Larsen et al. (2010) assessed the environmental and economic impacts of an MSW management system with alternative collection systems for recyclables (paper, glass, metal and plastic packaging) in Aarhus. The economic assessment focused on the WM expenses for the municipality and included the costs of the collection equipment, as well as actual collection and treatment costs. (Larsen et al., 2010).

In recent years the use of material flow analyses (MFA) and substance flow analyses (SFA) as supporting tools for LCA have also emerged (Arena and Di Gregorio, 2014; Rochat et al., 2013; Sevigné-Itoiz et al., 2014; Wäger et al., 2011). MFA is a systematic assessment of the flows and stocks of materials (goods and substances) within a system defined in space and time. It connects the sources, the pathways and the intermediate and final sinks of a material. MFA is a descriptive approach that provides snapshots of parts of the physical economy (Reuter et al., 2005). SFA is a type of MFA that focuses on a specific substance within a system. MFA and SFA can be applied in different types of studies in different ways. For example, the internal flows, stocks and losses of a company can be calculated with MFA. Examples of
applying SFA to WM systems can be found in Döberl et al. (2002), and examples of SFA applied to specific treatment methods can be found in Morf et al. (2005). The combination of MFA and SFA with an environmental assessment method such as LCA appears to be an attractive tool-box for comparing alternative WM technologies and scenarios, and supporting WM decisions on both strategic and operating levels (Arena and Di Gregorio, 2014). These types of MFA and SFA are the focus in this study.

MFAs and SFAs have been combined with LCAs for determining the cause-effect chains made up of physical flows in order to properly assess the market links between raw materials and waste (Sevigne-Itoiz et al., 2015) or to transparently report waste flows, as well as their connections and the impacts of process changes on them in a WM system (Tunesi et al., 2016). Arena and Di Gregorio (2014) and Tunesi et al. (2016) integrated SFA and MFA combined with LCA results for planning an MSW system.

Turner et al. (2016) combined an MFA and LCA to assess the performance of an MSW management system of a local authority and to compare it to alternative systems to assess the potential effectiveness of different waste policy measures. The approach focused on greenhouse gas (GHG) emissions, and the most efficient ways to reduce them. Padeyanda et al. (2016) applied an MFA and LCA to assess the potential environmental impacts of food WM practices in different recycling facilities for Daejeon Metropolitan City in Korea in order to find sustainable solutions for food WM. Sevigne-Itoiz et al. (2015) applied a consequential LCA to GHG emissions and combined it with an MFA to assess the paper and cardboard recycling system in Spain.

1.4 Aims, focus and structure of the thesis

This thesis examines various Finnish waste flows and WM systems from the viewpoint of their environmental and in some cases also economic impacts.

The thesis comprises of case studies which were chosen based on the fact that they all considered wastes or products from the forest sector. Wood-based products and wood waste are crucial for Finland due to the vast forest resources and high economic importance of the forest industry. Issues connected to the forest sector and wood-based products are actively discussed and developed due to the need in Europe to move towards a sustainable bioeconomy to ensure sufficient supplies of raw materials, energy and industrial products under conditions of decreasing fossil carbon resources. The management of waste flows originating from forestry products, such as waste paper, cardboard and wood waste, and different options for them are therefore of interest.

One of the case studies focuses on construction and demolition waste (C&DW), one of the priority waste flows in the EU Commissions Circular Economy package (European Commission, 2017a). At the national, but also the European level, C&DW is a waste flow of major concern, due to the large and growing volumes and low recycling rates.

Different levels of waste hierarchy are assessed and compared in the case studies presented in the thesis papers. The case studies were analysed using life cycle based approaches.
The thesis aims to answer the following research questions (RQ) by focusing on specific case studies on newspaper, C&DW and delivery packaging:

RQ1: Do the results of the life cycle based approaches that generate information on the environmental and/or economic impacts support the waste hierarchy order of WM options in the Finnish context?

RQ2: Does the combination of life cycle based approaches provide added value for decision-making for Finnish WM systems?

RQ3: How can normalisation be used for improving the understanding of LCIA results?

The analysis of the waste hierarchy levels proceeds from the lowest hierarchy levels, i.e., landfilling, thermal treatment with energy recovery and recycling, towards the second highest level, i.e., reuse (or preparing for reuse) (Table 2, Fig. 1). The focus is on the WM phase, i.e. the options available when products or materials are about to be discarded. For this reason, product design is beyond the scope of this study and the first level of the waste hierarchy, i.e., waste prevention, is not addressed.

The materials and methods used in this thesis are presented in Chapter 2. In Chapter 3 the results from the papers constituting this thesis are presented and discussed from the perspective of the research questions defined above. Finally, conclusions are drawn and recommendations for future research are given in Chapter 4.
2. Materials and methods

2.1 Case studies

The papers comprising this thesis are based on results of the following four case studies with varying focus and objectives and using varying life cycle based approaches (Table 2):
1) WM options for discarded newspaper in the Helsinki Metropolitan Area (Papers I and II)
2) Communication of LCIA results concerning newspapers and digital books for non-LCA practitioners (Paper III)
3) Current level and future insights into the environmental impacts of C&DW management system (Paper IV)
4) Comparison of a reusable and recyclable bread transport packaging (Paper V)

The WM options included in the case studies were landfilling, thermal treatment with energy recovery and recycling (Papers I, II, III, IV). Paper V evaluated options at the higher levels of waste hierarchy, namely recycling and reuse (Table 2).

Methodologically the study began from the life cycle inventory (LCI) (Paper I), then broadened LCI to include a life cycle impact assessment (LCIA) (Papers II – V) and combined LCIA with social or environmental life cycle costing (SLCC, ELCC) (Papers II and IV). Furthermore, an LCA and ELCC assessment was complemented by an MFA (Paper IV). An LCA was used for the environmental impact assessment in all papers (Table 2).

Process specific primary data were used for the inventory of the foreground system when available. Generic data from databases were used for the background system (Table 2, Fig. 3).
Table 2. Overview of the papers constituting the thesis. LCI = life cycle inventory, LCIA = life cycle impact assessment, SLCC = social life cycle costing, LCA = life cycle assessment, ELCC = environmental life cycle costing, MFA = material flow analysis, NP = newspaper, WM = waste management, CCB = corrugated cardboard).

<table>
<thead>
<tr>
<th>Paper number</th>
<th>Research questions (RQ) dealt with</th>
<th>WM options included</th>
<th>Waste hierarchy levels evaluated</th>
<th>Approaches used</th>
<th>Description of data</th>
</tr>
</thead>
<tbody>
<tr>
<td>I</td>
<td>RQ1</td>
<td>Landfilling, energy recovery, recycling</td>
<td>5, 4, 3</td>
<td>LCI</td>
<td>Primary data: NP manufacturing, collection and sorting of discarded NP. Generic data: forestry, waste recovery and treatment</td>
</tr>
<tr>
<td>II</td>
<td>RQ1, RQ2</td>
<td>Landfilling, energy recovery, recycling</td>
<td>5, 4, 3</td>
<td>LCIA, SLCC</td>
<td>Primary data: NP manufacturing, collection and sorting of discarded NP. Generic data: forestry, waste recovery and treatment</td>
</tr>
<tr>
<td>III</td>
<td>RQ3</td>
<td>Landfilling, energy recovery, recycling</td>
<td>5, 3</td>
<td>LCA</td>
<td>Primary data: Printing. Generic data: Pulp and paper production, WM</td>
</tr>
<tr>
<td>IV</td>
<td>RQ1, RQ2</td>
<td>Landfilling, energy recovery, recycling</td>
<td>5, 4, 3</td>
<td>LCA, ELCC, MFA</td>
<td>Generic data: the whole system</td>
</tr>
<tr>
<td>V</td>
<td>RQ1</td>
<td>Recycling, reuse</td>
<td>3, 2</td>
<td>LCA</td>
<td>Primary data: CCB box manufacturing and recycling, delivery system. Generic data: plastic crate manufacturing, recycling and energy recovery</td>
</tr>
</tbody>
</table>

1) Research questions: RQ1. Do the results of the life cycle based approaches that generate information on the environmental and/or economic impacts support the waste hierarchy order of WM options in the Finnish context? RQ2. Does the combination of life cycle based approaches provide added value for decision-making for Finnish WM systems? RQ3. How can normalisation be used for improving the understanding of LCIA results?

2) Primary data = measured/reported data from the processes involved in the system, generic data = data from literature and generic databases.
Materials and methods

Figure 3. General overview of elements identified in a combined production, consumption and waste management system of several products. The case study systems were defined to include parts of the overall system case specifically.

2.2 System descriptions and modelling assumptions

The papers included in this study considered products or fractions of MSW and C&DW streams (Table 3). The studies focused on one or several materials within these waste streams. Hazardous wastes were excluded.

The objectives of the studies varied (Table 3), from very general aims such as generating information on the impacts of different WM options for the development of waste policy on paper waste (Papers I and II) to very specific goals such as comparing the environmental impacts of a delivery system using either reusable high-density polyethylene (HDPE) plastic crates or recyclable corrugated cardboard (CCB) boxes for product transportation (Paper V). The FUs for the assessments were defined to meet the objectives of the studies (Table 3).

The most significant assumptions made for the assessments are related to the calculation of the benefits of material or energy recovery through avoided processes, i.e., processes that could be compensated for by waste recovery (Table 3). These processes were determined case specifically, based on aspects such as the level of assessment (regional, national), or the characteristics of the waste-based material or fuel, knowledge about current practices and foreseeable alternatives for recovery. Different methods for calculating the benefits of recycling and their consequences were tested using open-loop allocation (ISO, 2000), monetary allocation and the system expansion approach (Paper V). The results did not differ significantly from each other (less than 1%). Hence, the system expansion approach applying avoided processes recommended by ISO standard (ISO, 2006a) was chosen.
Table 3. Basic features and modelling assumptions for the systems studied. CCB = corrugated cardboard, C&DW = construction and demolition waste, FU = functional unit. LCIA = life cycle impact assessment, MSW = municipal solid waste, SLCC = social life cycle costing, SRF = solid recovered fuel, WM = waste management.

<table>
<thead>
<tr>
<th>Paper</th>
<th>Waste flows / products studied and FU used</th>
<th>Objectives</th>
<th>Scenarios</th>
<th>Assumptions on processes avoided by material or energy recovery of waste</th>
</tr>
</thead>
</table>
| I     | Discarded newspapers collected separately or within mixed MSW; FU: One tonne of newspapers delivered to consumers (76% to separate collection, 21% discarded into mixed waste, 3% to small-scale recycling) | To provide information based on inventory results on the emissions of different paper WM options. | 5 scenarios including closed-loop recycling\(^1\) or energy recovery (co-combustion or incineration) for newspaper, and landfilling or energy recovery for mixed waste; Additionally 4 different rates for separate collection (0, 45, 82, 97%) | Material recovery: use of virgin wood in newsprint manufacturing  
Energy recovery: the average Finnish energy production + use of coal and natural gas as fuels |
| II    | As for Paper I | To provide information based on LCIA and SLCC on different WM options for the development of waste policy on paper waste. | As for Paper I | As for Paper I |
| III   | Newspaper, digital photo-book; FU: One tonne of the product | To assess the differences between Finnish and European reference values for normalisation, and the impacts these have on the interpretation of normalised results. | None | Material recovery of newspaper: use of virgin wood in newsprint manufacturing  
Energy recovery: average Finnish energy production  
No avoided processes for the photobooks |
| IV    | C&DW\(^2\); FU: One tonne of C&DW generated | To provide a general view of the environmental and economic impacts of C&DW management and the key issues influencing these.  
To assess the Finnish C&DW management against the EU recycling target 70% by 2020. | 3 scenarios including recycling of metals and minerals, and energy recovery of wood and SRF\(^3\) produced from miscellaneous fraction | Material recovery of metals: use of virgin ore in iron and aluminium production;  
Material recovery of concrete and minerals: use of gravel;  
Energy recovery of wood: heavy fuel oil in a heat boiler;  
Energy recovery of SRF: natural gas in a co-combustion plant |
| V     | Plastic crate, CCB box; FU: Delivery of 8 loaves of bread in one crate/box (One delivery requires 1/700 of a plastic crate or 1 CCB box) | To compare the environmental impacts of a delivery system using either reusable plastic crates or recyclable CCB boxes for product transportation. | 2 scenarios with different delivery packaging including recycling (20%) and energy recovery (80%) for the plastic crate, and recycling for the CCB box | Material recovery of plastic crate: production of impregnated wood;  
Material recovery of CCB box: use of virgin fibres (fluting) for core-board manufacturing;  
Energy recovery of plastic crate: average separate heat production |

\(^1\) Closed-loop recycling of separately collected newspapers for newsprint manufacturing  
\(^2\) C&DW constituted of five main fractions: metals, concrete & mineral, wood, miscellaneous and an unsorted mixed fraction  
\(^3\) SRF consisted of 80% bio-based materials (e.g. cardboard and wood) and 20% plastics.
Materials and methods

The system boundaries were defined for an LCA case-specifically to include only the processes and flows relevant for the assessment (Fig. 4 and 5).

Paper I evaluated emissions and Papers II, III and V evaluated the impacts of various end-of-life options for a product life cycle performance, where the processes of the whole product life cycle were included (Fig. 4). Two types of approaches were distinguished: 1) one product with one or several end-of-life options (Papers I, II and III), and 2) several products performing the same function, but having different end-of-life options (Paper V).

The study on C&DW in Paper IV represents an assessment of a WM system (Fig. 5). The common Finnish C&DW management system began with the waste generated and sorted at construction, demolition and renovation sites and ended at the point where outputs such as material or waste based fuel were recovered into production. The basic flows and description of the system produced by implementing the MFA was expanded for an LCA and ELCC by adding flows (mainly fuels and energy) needed to maintain the system and processes that could be avoided by recovering the outputs.
2.3 Data sources and methods for life cycle assessment

The inventory data used in the case studies were compiled from several sources depending on the goal and scope of the study and the system assessed (Table 4).

The approaches used for the LCAs differed in the studies (Table 5). In Paper I only emission profiles based on LCIs were assessed, whereas an LCIA was carried out to the normalisation phase in Papers III and V. In Paper II an LCIA was carried out even further and was extended to weighting. Different methods and approaches were used for assessing the life cycle environmental impacts (Table 5). Paper II compared the LCIA results produced by three different methods, whereas in Papers III and V, and LCIA was performed using one method.

The impact categories included in the LCIA were selected based on the objectives of the studies, availability of data and the relevance of the impact categories to the studied product systems. Several impact categories were assessed in Papers II, III and V, whereas in Paper IV only climate change impacts were assessed (Table 5). In Paper IV climate change impacts were considered sufficient for providing a general picture of the environmental and economic impacts of C&DW management and the key issues influencing these.
Table 4. Data sources for the case studies included in the thesis. If not separately indicated, the cost data originated from the same source as the environmental data. CCB = corrugated cardboard, C&DW = construction and demolition waste, NP = newspaper.

<table>
<thead>
<tr>
<th>Paper</th>
<th>Processes / life cycle phases</th>
<th>Data sources</th>
</tr>
</thead>
<tbody>
<tr>
<td>Paper I</td>
<td>• Forestry operations</td>
<td>• KCL-ECO Database</td>
</tr>
<tr>
<td></td>
<td>• Pulp and paper mill: NP manufacturing, recycling, energy production</td>
<td>• Process data from the pulp and paper mill (Paper I)</td>
</tr>
<tr>
<td></td>
<td>• Separate collection and pretreatment of NP</td>
<td>• Data from operators (Paper I)</td>
</tr>
<tr>
<td></td>
<td>• Landfilling of NP</td>
<td>• Generic data from literature (Paper I)</td>
</tr>
<tr>
<td></td>
<td>• Thermal treatment with energy recovery of NP</td>
<td>• Planning data for co-combustion; Literature data for incineration (Paper I)</td>
</tr>
<tr>
<td></td>
<td>• Energy production avoided</td>
<td>• Generic data from literature (Paper I)</td>
</tr>
<tr>
<td>Paper II</td>
<td>As for Paper I</td>
<td>As for Paper I</td>
</tr>
<tr>
<td>Paper III</td>
<td>• Pulp and paper production</td>
<td>• KCL-ECO Database</td>
</tr>
<tr>
<td></td>
<td>• Printing</td>
<td>• Data from operators (Paper III)</td>
</tr>
<tr>
<td></td>
<td>• End-of-life scenarios for NP, including material and energy production avoided</td>
<td>• Generic data from literature (Paper III)</td>
</tr>
<tr>
<td>Paper IV</td>
<td>• Quality and quantity of waste generated</td>
<td>For all processes/phases:</td>
</tr>
<tr>
<td></td>
<td>• Distribution of waste fractions</td>
<td>• Expert evaluations (Paper IV)</td>
</tr>
<tr>
<td></td>
<td>• Quality of outputs from C&amp;DW management</td>
<td>• Generic data from literature (Paper IV)</td>
</tr>
<tr>
<td></td>
<td>• Energy production avoided</td>
<td>• Cost data from operator (Paper IV)</td>
</tr>
<tr>
<td></td>
<td>• Material production avoided</td>
<td></td>
</tr>
<tr>
<td>Paper V</td>
<td>• Plastic crate manufacturing</td>
<td>• Ecoinvent v. 2.2 Database (Paper V)</td>
</tr>
<tr>
<td></td>
<td>• Plastic crate delivery</td>
<td>• Data from operators (Paper V)</td>
</tr>
<tr>
<td></td>
<td>• Plastic crate end-of-life, including material and energy production avoided</td>
<td>• Generic data from literature (Paper V)</td>
</tr>
<tr>
<td></td>
<td>• CCB manufacturing</td>
<td>• Data from operators (Paper V)</td>
</tr>
<tr>
<td></td>
<td>• CCB delivery</td>
<td>• Data from operators on the current system (Paper V)</td>
</tr>
<tr>
<td></td>
<td>• CCB end-of-life, including material production avoided</td>
<td>• Data from operators, based on assumptions on the recycling process (Paper V)</td>
</tr>
</tbody>
</table>

Table 5. Comparison of the methodologies used for the life cycle assessments (LCAs) in the papers. LIC = life cycle inventory, LCIA = life cycle impact assessment, DAIA = Decision Analysis Impact Assessment, EPS = Environmental priority system.

<table>
<thead>
<tr>
<th>Paper</th>
<th>Final phase of the LCA</th>
<th>LCIA methodology used</th>
<th>Modelling approach of the LCIA methodology</th>
<th>Number of impact categories assessed</th>
<th>Reference values for normalisation</th>
<th>Weighting method used in the LCIA model</th>
</tr>
</thead>
<tbody>
<tr>
<td>I</td>
<td>LCI</td>
<td>None</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>II</td>
<td>LCIA (weighting)</td>
<td>DAIA</td>
<td>midpoint</td>
<td>5</td>
<td>Finnish panel</td>
<td>Panel</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Eco-indicator</td>
<td>endpoint</td>
<td>11</td>
<td>European panel</td>
<td>Panel</td>
</tr>
<tr>
<td></td>
<td></td>
<td>EPS 2000</td>
<td>endpoint</td>
<td>16</td>
<td>None</td>
<td>Monetary</td>
</tr>
<tr>
<td>III</td>
<td>LCIA (normalisation)</td>
<td>ReCiPe</td>
<td>midpoint</td>
<td>7</td>
<td>European, Finnish consumption, Finnish production</td>
<td></td>
</tr>
<tr>
<td>IV</td>
<td>LCIA (characterisation)</td>
<td>ReciPe</td>
<td>midpoint</td>
<td>1</td>
<td>None</td>
<td></td>
</tr>
<tr>
<td>V</td>
<td>LCIA (normalisation)</td>
<td>ReCiPe</td>
<td>midpoint</td>
<td>6</td>
<td>European</td>
<td></td>
</tr>
</tbody>
</table>

\(^1\) see figure 2, chapter 1.3; \(^2\) Seppälä, 1999; 2003; \(^3\) Goedkoop and Spriensma, 2001; \(^4\) Steen, 1999a; 1999b; \(^5\) version 1.08, December 2012; Goedkoop et al., 2009
2.4 Life cycle costing and material flow analysis

To strengthen the LCA, an economic assessment was combined with it in Paper II (Table 2). In Paper IV the LCA was combined with both an MFA and an economic assessment. The cost calculations and environmental impact assessments were performed separately, but they were both applied to the system defined for the LCA as the basis for calculations. Some exceptions (explained below), were made case specifically to the boundaries. MFA results on the treatment lines of different waste fractions were used as the basis for the LCA system definition in Paper IV.

The economic assessment was performed using SLCC in Paper II and ELCC in was used in Paper IV. SLCC looks at the costs from the point of view of society, hence SLCC was preferred for analysing the costs of waste policy alternatives for paper waste, for which the responsibility of management is shared by society (i.e. municipalities) and industry (Paper II). The regulations on producer responsibility (Government decree on separate collection and recycling of wastepaper, 528/2013) have obliged the forest industry to organise the collection and recovery of wastepaper but left the municipalities with the responsibility for non-recovered wastepaper. Here the analysis must combine the costs and benefits accruing to multiple actors, firms and consumers. In contrast, in the case of C&DW (Paper IV) the municipalities are only responsible for the C&DW generated by households, and the majority of C&DW is taken care of by companies. Here the viewpoint of a single enterprise can be used in the cost calculations, as is done in ELCC, since the company both bears the costs and receives the credits from the treatment and recovery of C&DW.

The SLCC (Paper II) covered the costs associated with the life cycle of a newspaper within the boundaries of the LCA: e.g., direct costs of investments, labour and energy (Paper II). One exception was made compared to the LCA: the costs of increased communication efforts were included, although the environmental impacts were not included in the LCA, since they were assumed to be low compared to the overall impacts. The ranking for newspaper waste management options from the LCIA was compared to that produced by SLCC. Additionally, the external costs associated with environmental impacts related to climate change and depletion of fossil fuels were added and compared to the LCIA results of the corresponding impact categories.

The ELCC assessment (Paper IV) was implemented in a C&DW management company. The timescale of the analysis was one year of C&DW management system operations and the assessment included direct internal costs and capital, following the guidelines of ELCC (Swarr et al., 2011).

A material flow analysis (MFA, Paper IV) was applied to produce a description of the waste flows within the system and to quantify the inputs and outputs of the C&DW system. Material and energy recovery rates were calculated from the outputs of sorting and separation, i.e. the inputs to the recovery processes. The MFA was performed using the STAN 2.0 software (Cencic and Rechberger, 2008).
3. Results and discussion

3.1 Environmental and economic impacts on different waste hierarchy levels

Different levels of waste hierarchy and their emissions or environmental and/or economic impacts were assessed especially for two systems, namely for newspaper production and WM (Papers I and II) and for a bread delivery system focusing on packaging (Paper V). The results showed that moving up in the waste hierarchy does not necessarily improve the environmental or economic performance of the system studied.

Of the two studied cases, only the one concerning the WM of discarded newspaper (Papers I and II) focused on comparing the hierarchy levels for the identical waste fraction. The evaluation was performed in three phases: 1) using emission profiles from the LCI (Paper I), 2) using LCIA results obtained via three LCIA methods (Paper II) and 3) using additionally economic results from SLCC (Paper II). The LCI emissions profile results supported the waste hierarchy by showing lower emissions for recycling and thermal treatment with energy recovery than for landfilling. Different emission parameters slightly changed the preference order, nevertheless landfilling clearly showed the highest emissions. However, inventory data refers to inputs and outputs of the system, not environmental impacts, hence the environmental performance of different options cannot yet be interpreted on the basis of the LCI results (ISO, 2006b).

From the LCI phase, the study was continued with an LCIA (Paper II). The LCIA results showed that increasing the energy recovery of newspaper waste in a co-combustion plant resulted in a better performance compared to recycling if fossil fuels were compensated for. However, different LCIA models and different impact category results gave slightly different results. Especially, the impacts related to land occupation and conversion, which are included in the Eco-Indicator 99 LCIA methodology (in the damage to ecosystem quality category), showed that recycling outperformed energy recovery. Nevertheless, none of the models adequately assessed the biodiversity effects of forestry. Hence, a highly important impact of wood-based products was underestimated in the study. In contrast to the LCIA results, the SLCC (without environmental costs) showed a preference, i.e. lowest costs, of landfilling followed by recycling (combined with incineration).

Several LCA studies on paper WM (e.g. Merrild et al., 2008; 2012; Villanueva and Wenzel, 2007) have found recycling to perform environmentally better than energy recovery. Additionally, based on their review of LCAs for SWMSs Laurent et al. (2014a) concluded that with the exception of landfilling, there was no definite agreement on the best performing waste
treatment technology, but most studies tended to favour recycling over landﬁlling and thermal treatment for paper. However, they also concluded, that the strong dependence of each SWMS on its context or local conditions prevents generalisation of LCA results, which is currently symbolised by the waste hierarchy (Laurent et al., 2014a). Contradicting the studies by Merrild et al. (2008; 2012) and Villanueva and Wenzel (2007) this study indicated that recycling of newspaper generated fewer environmental beneﬁts than its energy recovery. Common to Finland, the mill studied is an integrated pulp and paper mill, where the interconnections between processes and ﬂows are complex and may affect the product system in unexpected ways when changing WM options. Recycling could have been improved by making changes in the paper mill phase, such as using grid electricity with low emissions for power generation instead of fossil fuels.

The case study on delivery packaging (Paper V) compared the impacts of reusable or recyclable packaging on the performance of the delivery system. The waste hierarchy speciﬁes reuse on a higher level of priority than recycling and several studies comparing packaging systems have concluded that reusable plastic containers outperform single-use packaging (e.g. Levi et al., 2011; Raugei et al., 2009; Singh et al., 2006). This assessment showed that with the speciﬁc delivery system studied, the recyclable packaging performed better than the reusable packaging. A prerequisite for this result was that the recycled product had an efﬁcient recycling system, where virgin material was compensated with recycled material. In Raugei et al. (2009) a comparison of transport of a batch of chemicals by means of single-use disposable fibre drums versus reusable steel drums resulted in the reuse of drums 200 times with lower environmental impacts than the single-use ﬁbre drums. However, unlike the recyclable delivery boxes in Paper V, the ﬁbre drums were landﬁlled.

The results of the delivery packaging study are very case speciﬁc. Transportation was shown to dominate in all impact categories and changes in the weights of the crates/boxes and the modes of transport, for example, may change the results. The results underline that material properties such as reusability or recyclability cannot solely be used as the basis for choosing materials for a speciﬁc use. In contrast, a broader view needs to be obtained using a systemic assessment. This conclusion is also supported by Sørensen and Wenzel (2014), who found bedpans made of disposable plastics to perform better than reusable steel products.

The assessment of the beneﬁts gained by waste recovery is crucial in LCAs for WM. For the material recovery options in both the newspaper study and the bread delivery packaging study, the impacts of the wood saved by recycling were assessed conventionally, i.e., without considering the potential increase of carbon sequestration in forests or an alternative use for the saved wood. The impacts of wood harvesting on the forest carbon balance have been studied and discussed, for example by Soimakallio et al. (2016). However, scientiﬁc consensus is yet to be found on how to incorporate CO2 removal by forests into product speciﬁc assessments.

Assumptions on the alternative use of wood saved by recycling have been identiﬁed as one of the key factors for the environmental impacts concerning the recycling of paper in comparison to its thermal treatment with energy recovery by Villanueva and Wenzel (2007), for example, and Merrild et al. (2012). Energy use of the saved wood to compensate fossil fuels has
been assumed when modelling paper recycling by Merrild et al. (2012). However, the energy use of saved wood was not included in the LCAs of this study. The reasoning for this was that the wood for newsprint manufacturing is obtained mainly from forest management operations, i.e. from thinning. If the demand for wood for paper manufacturing decreases, the outcome may be fewer forest management activities leaving the thin trees (i.e. the pulp wood) in the forest. Hence, no alternative use would exist. In the current situation, however, energy use would be a relevant assumption, since the use of forest-based bioenergy is constantly growing. Wood-based energy is primarily (80% in 2014) produced using residues from the forest industry and from silvicultural and harvesting operations (Koponen et al., 2015).

Additionally, new products are being developed from wood fibres to compensate for the decreasing use of wood in paper manufacturing. A recent example of such product development are wood based textiles (Michud and Rissanen, 2014). Such textiles could in the long run replace the production and use of cotton and this production could in the future be assumed as an alternative for the wood saved by paper recycling. The European Commission’s Circular Economy Package also calls for new wood-based products and for the cascading use of renewable resources (European Commission, 2017a). Biorefineries can offer practical solutions for generating a wide range of products for food, material and energy applications (Metsä Group, 2017). The cascading use of wood requires that the biomass is converted into products, which can be recycled or used for energy production after their first utilisation stage (Vis et al., 2016). The consideration of cascading would probably have changed the results of this study, but to verify this, the outcome would need to be assessed.

The benefits of the waste to energy (WtE) options studied in Paper II were assessed assuming that the energy produced from waste based fuels would compensate fossil fuels, which substantially affected and improved the performances of the WtE options. In the comparison of MSW management for seven countries, Bassi et al. (2017) concluded that WtE plants can lead to environmental loads or savings, depending on the energy efficiency of the plant and on the composition of the energy being substituted. However, on the one hand, the use of renewable resources for energy production is increasing (Bioeconomy, 2017), and with this development the benefits gained from WtE will decrease and probably improve the environmental performance of recycling in the future. On the other hand, in the long run, in the circular economy the use of secondary materials in production should become a default and this would probably reduce the benefits gained by recycling. Hence, it is challenging to predict the effects of these changes on the environmental performance of different WM options without systemic assessments. These conclusions are supported by the results by Arushanyan et al. (2017), who assessed several future scenarios for the Swedish WM system. They concluded that by recycling and recovering materials and energy the WM system would continue to contribute to the production of materials, energy carriers and other products replacing virgin production. However, this would lead to lower environmental benefits in the more sustainable scenarios, where the surrounding energy and transportation systems are less harmful to the environment and less waste is produced (Arushanyan et al., 2017).
3.2 Added value for decision-making of combining life cycle based approaches

All the case studies presented here were research oriented, and did not directly aim at producing results for specific decision-making situations. However, in general it can be claimed that in the contemporary society, decision-making is still heavily based on costs. Environmental aspects are gaining more weight, but it is clear that very seldom do environmental values alone determine decision-making. Adding an economic analysis to the environmental assessment, as was done in Papers II and IV, increases the usability of the LCA results.

The joint assessment of environmental and economic impacts of the newspaper WM options (Paper II) showed that including both dimensions in the assessment is crucial for providing a basis for making sustainable decisions. The differences in the environmental impacts between material and energy recovery were relatively small. Concentrating solely on the economic aspects would lead to the environmentally worst alternative, whereas the environmentally best solution would result in the highest costs. When adding environmental costs to the SLCC calculations, the results did not change crucially, but showed that both options that included higher recycling (Cases 2a and 3a) had the second lowest costs (Table 6). This way SLCC provided an important additional aspect for the decision-maker by showing that when environmental costs are considered in the assessment, the thermal treatment with energy recovery of separately collected waste paper is more expensive than recycling (Table 6). Larsen et al. (2010) assessed the environmental and economic impacts of an MSW management system with alternative collection systems for recyclables (paper, glass, metal and plastic packaging) in Aarhus and obtained similar results on the economic impacts but contradictory results on the environmental impacts. The results by Larsen et al. (2010) showed that enhancing recycling instead of incineration was recommendable due to improved performance in several environmental impact categories. In addition, the costs for the municipality from waste collection and treatment were reduced by increasing recycling, mainly because the high cost of incineration was avoided. However, the results were considered to be sensitive to assumptions made about the quality and potential utilisation of the recyclables and the energy substituted by the WtE. (Larsen et al., 2010).

Table 6. The ranking of the newspaper (NP) waste management alternatives with the life cycle impact assessment (LCIA) results produced by DAIA (Decision Analysis Impact Assessment), Eco-indicator 99 and EPS 2000 (Environmental priority system) (I = lowest environmental impacts, V = highest environmental impacts), with the social life cycle costing (SLCC) results without environmental costs, and with the SLCC results including environmental costs (related to CO2 emissions and resource scarcity) (I = lowest costs, V = highest costs) (modified from Paper II).

<table>
<thead>
<tr>
<th>Waste management alternative</th>
<th>Ranking with LCIA results</th>
<th>Ranking with SLCC results</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>DAIA, impact value / t of NP</td>
<td>Eco-indicator 99, ecopoints / t of NP</td>
</tr>
<tr>
<td>Case 1</td>
<td>7.31 (V)</td>
<td>36.1 (V)</td>
</tr>
<tr>
<td>Case 2a</td>
<td>6.20 (III)</td>
<td>33.8 (III)</td>
</tr>
<tr>
<td>Case 2b</td>
<td>3.73 (I)</td>
<td>28.0 (I)</td>
</tr>
<tr>
<td>Case 3a</td>
<td>6.74 (IV)</td>
<td>35.4 (IV)</td>
</tr>
<tr>
<td>Case 3b</td>
<td>5.32 (II)</td>
<td>33.1 (II)</td>
</tr>
</tbody>
</table>

\(^{(1)}\) ELUs = environmental load units

In the C&DW management system assessment (Paper IV) an MFA and ELCC were combined with an LCA to provide a better understanding of the studied system. The MFA provided the
Results and discussion

basis for an LCA and ELCC. Similarly, Sevigne-Itoiz et al. (2015) combined an MFA with a consequential LCA of GHG emissions to assess the paper and cardboard recycling system in Spain. They concluded that it is necessary to have methodologies that properly map all material flows as a first step in determining the potential environmental impacts to facilitate the accurate accounting of GHG for decision making (Sevigne-Itoiz et al., 2015).

The overall assessment of the C&DW management system showed that both environmental and economic benefits can be gained with the current system. Together the different methodologies gave a diversified view of the potentials that different waste fractions have in improving the C&DW management (Table 7). The rate of material or energy recovery calculated using the MFA shows the current recovery situation for the specific waste fraction, while the climate change impacts from an LCA indicate the environmental performance and the economic profits from ELCC describe the economic performance of treatment of the specific fractions. The volume of each fraction reflects the potential that the fraction provides for improving the performance of the overall system (Table 7).

Table 7. Material and energy recovery rates (material flow analysis, MFA, results), climate change impacts (LCA results) and economic profits (environmental life cycle costing, ELCC, results) of the treatment of different waste fractions in the current construction and demolition waste (C&DW) management system. The results are comparable only within one methodology. (Modified from Paper IV).

<table>
<thead>
<tr>
<th>Source separated waste fraction</th>
<th>MFA(^1)</th>
<th>LCA(^2)</th>
<th>ELCC(^3)</th>
<th>Volume of the fraction</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Material recovery</td>
<td>Energy recovery</td>
<td>Climate change impacts</td>
<td>Profits</td>
</tr>
<tr>
<td>Metal</td>
<td>++</td>
<td>-</td>
<td>+</td>
<td>++ small</td>
</tr>
<tr>
<td>Concrete and mineral</td>
<td>++</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Wood</td>
<td>-</td>
<td>++</td>
<td>++</td>
<td>+ large</td>
</tr>
<tr>
<td>Miscellaneous waste</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+ medium</td>
</tr>
<tr>
<td>Mixed waste</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>- medium</td>
</tr>
</tbody>
</table>

Scaling (based on the results of MFA, LCA and ELCC):

\(^1\) MFA: ++ = maximal material/energy recovery, + = medium material/energy recovery, - = low material/energy recovery.

\(^2\) LCA: ++ = minimum climate change impact, + = medium climate change impact, - = high climate change impact.

\(^3\) ELCC: ++ = maximal profits, + = medium profits, - = low profits.

Combining the different approaches indicated ways to achieve the recycling target of 70% set by the EU for 2020. At the time of the study, the recycling rate for C&DW was way beyond the target. Assessment of the effects of assumed changes in the waste composition on the recycling rate (Table 8) showed that an increase in the concrete and mineral fraction (Scenario 2) would improve the material recovery rate but would generate a reduction in the overall climate benefits and economic profits of the system. In contrast, an increase in the wood fraction (Scenario 1) would reduce the material recovery rate but increase the climate benefits and economic profits (Table 8). Hence, changes in waste composition will not be adequate for achieving the recycling target for C&DW.

According to these results, the 70% recycling target could not be attained unless the share of wood in the C&DW can be reduced or recycling technologies for low-quality wood can be developed. According to a recent study on C&DW, the share of wood had decreased to 15% in 2013 due to recession and a reduction in building wooden one-family houses (Salmenperä et al., 2016). In a better economic situation, an increase is again foreseeable. Since the share of wood is exceptionally high in the Finnish C&DW (36%) compared to other EU countries (2-
4%, Monier et al., 2011), the results indicate that regional differences in waste composition may support arguments for differing recycling targets in different European regions or alternatively focusing the targets on materials, not the overall waste flow.

Similarly, Turner et al. (2016) combined LCA and MFA to compare the performance of existing MSW management system and alternative systems formed using different waste policy measures. The recycling rates calculated from the MFA results suggested that national recycling targets were unlikely to be met even if the assessed policies were optimally implemented. Turner et al. (2016) considered it likely that for the targets to be met, the investigated policies would need to be combined with additional policies that aim at reducing waste generation.

Table 8. The overall recovery rate, material and energy recovery rates, rate of utilisation in landfills, disposal rate and the climate change impact reductions and profits for the typical status, and scenarios 1 and 2 for construction and demolition waste (C&DW) management (modified from Paper IV).

<table>
<thead>
<tr>
<th></th>
<th>Typical status</th>
<th>Scenario 1</th>
<th>Scenario 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Overall recovery rate</td>
<td>73</td>
<td>70</td>
<td>66</td>
</tr>
<tr>
<td>Material recovery, %</td>
<td>38</td>
<td>27</td>
<td>44</td>
</tr>
<tr>
<td>Energy recovery, %</td>
<td>35</td>
<td>43</td>
<td>22</td>
</tr>
<tr>
<td>Utilisation at landfills, %</td>
<td>6</td>
<td>5</td>
<td>7</td>
</tr>
<tr>
<td>Disposal, %</td>
<td>21</td>
<td>25</td>
<td>26</td>
</tr>
<tr>
<td>Climate change impact reduction, kg CO₂-eq/t of C&amp;DW</td>
<td>360</td>
<td>390</td>
<td>210</td>
</tr>
<tr>
<td>Profit, €/t of C&amp;DW</td>
<td>81</td>
<td>79</td>
<td>48</td>
</tr>
</tbody>
</table>

*material and energy recovery, excluding utilisation at landfills

As a whole, the overall performance of the C&DW management system could be significantly improved by enhancing the sorting of the mixed and miscellaneous waste fractions and developing the current recovery practices especially for the wood waste and mineral waste fractions. This finding is supported by Manninen et al. (2015) who concluded that environmentally, thermal treatment with energy recovery outperformed the currently most common recycling concepts for wood waste, namely the manufacture of composites and particle boards. As for the mineral fraction Butera et al. (2015) also concluded that the current recycling in road construction is preferable to landfilling, but does not generate environmental benefits. Hence, for both wood and mineral waste there is clearly a need for developing environmentally more beneficial recycling concepts.

Research and development (e.g. ongoing Horizon 2020 project HISER; HISER, 2017) on the possibilities to increase recycling of the various materials, e.g. wood, in C&DW may lead to new solutions. Nevertheless, before implementation, new solutions need to be assessed for their environmental and economic impacts. Considering several rounds of recovery for wood waste, as is the case in cascading use of wood waste, benefits from both material and energy recovery could be gained. According to Vis et al. (2016), the construction and demolition sector is a questionable source of wood for cascading primarily due to the growing material complexity and use of different wood materials. There is more potential in the construction sector, for which research is ongoing to increase wood recyclability and recycled products (Vis et al., 2016).

Although the environmental and economic assessments produced somewhat controversial results in the case studies, their joint use deepened the understanding of the systems (Papers
II and IV) and showed that striving for the maximum environmental improvement and minimum cost may produce a good compromise between ecological and economic aspects (Paper II). Using different methodologies and analysing their results in different combinations can reveal various potentials for improving the environmental or economic performance of the system overall. Although the application of several methodologies increases the amount of information generated, this needs not be confusing for the decision-making. The results can be broken down and combined in various ways in order to ease their interpretation and to highlight different aspects.

A common problem for all the methodologies especially in the C&DW case (Paper IV) was the lack of proper data on the treated waste, the quality and quantity of outputs and the technology and costs of the processes. The results show that in the future, basic information needs to be produced on the characteristics and management of C&DW in order to support the generation of reliable and accurate information on the possibilities and benefits of increasing waste recycling.

3.3 Using normalisation to improve the understanding of LCIA results

The understandability of LCIA results is crucial to their usability and value for decision-making. Normalisation was used to show the contribution of the studied system to the impacts of a reference system and hereby improve the understandability of the LCIA results (Papers III and V). The use of different normalisation reference values and their impact on the understanding of LCA results by LCA non-practitioners were assessed for newspapers and digital photo books (Paper III).

The use of different reference values for normalisation (Paper III) produced slightly different LCIA results for the studied products. LCA non-practitioners considered that the consumption-based national reference values produced somewhat more easily understandable LCIA results than the EU-based reference values. In their survey on the state of the art of normalisation and weighting approaches in LCA, Pizzol et al. (2017) concluded that consumption-based normalisation references were preferable to production-based ones because of better consistency with the geographical scope of the LCA studies. However, they highlighted that consumption-based normalisation references are in practice almost entirely unavailable today (Pizzol et al., 2017).

The fact that the harmfulness of the impacts in one category relative to impacts in another category cannot be judged on the basis of the normalised results was difficult for LCA non-practitioners to comprehend (Paper III). Generally, even after normalisation, the LCIA results were considered difficult to understand. The complexity of the processes and systems they describe is one reason behind this. Descriptions of the methods and assumptions used were considered important information for understanding the methods and their restrictions and to support the interpretation. Furthermore, Pizzol et al. (2017) recommended communicating normalised results clearly by reporting units and explaining their meaning, as these may not be easily understandable to audiences beyond LCA experts.
The study showed that normalised LCIA results can be feasible in business to business communication since they provide information on the contribution of the studied system to the overall environmental impacts of the reference system (Paper III). This helps to focus the system development and technical improvements to mitigate the relevant environmental problems. Descriptions of the methods and assumptions used are important for supporting the interpretation. However, it is challenging to keep the information simple and at the same time scientifically valid.

These results highlight the complexity of LCIA results and the challenges in their interpretation. The results support the approach of ISO standards in that characterisation should be the last mandatory step of an LCIA due to the fact that normalisation brings subjectivity into the results and their interpretation. However, subjectivity can be entailed in the assessment also in the earlier phases of LCA, for example in the LCI phase via the selection of input data and assumptions for modelling the system as well as in the LCIA phase via the selection of impact categories. Therefore, transparency is a key requirement for reporting LCA and other assessments.

The challenge of presenting results from LC-based assessment in a comprehensive and understandable way has been recognised by Finkbeiner et al. (2010), for example, who suggested more sophisticated communication tools such as the “Life Cycle Sustainability Dashboard” and the “Life Cycle Sustainability Triangle” for communicating to both experts and non-expert stakeholders. To improve the communication of LCIA results to experts, decision-makers and the public, the European Commission DG Environment and European Commission’s Joint Research Centre (JRC IES) have worked on developing a harmonised methodology for the calculation of a Product Environmental Footprint (European Commission, 2016). Communication to business partners, consumers, stakeholders, e.g., public administration, NGOs and/or investors was tested in the Product Environmental Footprint Pilots (European Commission, 2017b). Different ways to communicate on environmental impacts to target audiences, such as performance (improvement) labels, barcodes, information available in shops, instruction manuals, product declarations, consumer receipts, websites, applications, campaigns and sustainability rankings were used during the testing (European Commission, 2017b).

3.4 Uncertainties of the assessments

The assessments included in the study contain a number of uncertainties related to the data used, the assumptions concerning, e.g., avoided impacts and the environmental impact categories.

The data sources used in the environmental assessments are summarised in Table 4 (Chapter 2.3). In general, the environmental data used for assessing C&DW management included the highest uncertainty, starting from the volume and quality of C&DW, distribution of the waste into various fractions, processing of the fractions, the quality and recyclability of the outputs and the recycling processes. Due to these uncertainties the climate change impacts can be over- or underestimated. Sensitivity checks were not performed since the range of parameter values could not be estimated. Hence, the results of this assessment are rough approximates.
and can only be utilised for generating a better understanding of the overall C&DW management chain and the critical aspects affecting the performance of the chain.

In contrast to the C&DW study, parts of the data for the other cases were specific to the processes and process chains under study. For Papers I and II the data concerning newsprint manufacturing, recycling and energy production at the pulp and paper mill were obtained from the mill, and hence can be considered reliable. Additionally, the data on separate collection and pre-treatment of paper waste prior to recycling consisted of process specific data from operators. In contrast, data on the WtE options were very uncertain, due to the fact that the plants were not yet operating at the time of the study. This together with the assumptions on fuels and energy production compensated for by the waste-based energy generates high uncertainties in the results.

For Paper III, database data and generic data from the literature were mostly used. For the printing phase, data were compiled from operators. The data on environmental releases and extractions behind the normalisation reference values used in the study were of different accuracy and extensiveness. The Finnish reference values were generated from the environmentally extended input-output (EE-IO) model of the Finnish economy, where the emissions were compiled for each industry from the national emission inventories or calculated on the basis of activity information (Seppälä et al., 2009). For the ReCiPe reference values for Europe, on the other hand, the emissions of several substances were based on estimations and calculations (Wegener Sleeswijk et al., 2008). These differences would weaken the comparability of the normalised LCIA results produced with the different reference values. However, they would not affect the stakeholders’ perception of the results, since here the focus was not on comparing, but on understanding the results obtained with different normalisation references.

In the bread delivery study (Paper V) data on the current delivery system using plastic crates and on CCB manufacturing were obtained from the operators. However, databases were used for plastic crate manufacturing and its end-of-life options. Since the use of CCB in the system was an option for the future, assumptions were made regarding its recycling. However, the data for the recycling process were obtained from the operator, but it is uncertain, whether the CCB would in practice be recycled in the modelled process.

Economic data is always difficult to obtain, but in the newspaper case study (Paper II) reliable and comprehensive data on costs were obtained from operators similarly to the data on emissions and use of resources. For the options including thermal treatment with energy recovery the cost data were more uncertain, since the facilities were not yet operating. In contrast, in the C&DW management system study (Paper IV), cost data was received from just one company. Yet the data gained was comprehensive for the treatment chain of this company and therefore helped in improving the understanding of the C&DW management chain.

The assessment of environmental impacts was not comprehensive and sufficient in all case studies. In the newspaper case studies (Papers II and III), the biodiversity impacts of paper recycling were not included in any of the LCIA models used, although they are of great importance in connection with wood based products. Additionally, only a restricted number of
impact categories were analysed in the other case studies (Papers III, IV and V), which were selected based on data availability and the relevance to the studied product systems. For the newspaper and digital photobook LCA (Paper III) the impact categories related to human toxicity or ecotoxicity were left out primarily due to the incompleteness of inventory data. Inventory data from generic databases included toxic emissions, but the process-specific data for the core processes did not. For the C&DW management system (Paper IV) only climate change impacts were assessed due to the lack of data on other than energy use within the treatment chains. In comparing the reusable and recyclable delivery boxes (Paper V), one important category missing from the selected impact categories was land use, for which no data were obtained and hence it was not considered. Restricted impact categories may limit the possibility to interpret the environmental impacts of the system from the LCA results alone. Hence, care should be taken in drawing any conclusions.

When assessing waste containing wood-based products, the inclusion of the positive impacts on biodiversity or carbon sequestration arising from saving wood by recycling would probably affect the results of this study. However, the models used in this study did not consider these elements. The impacts of forestry on biodiversity may result from complicated and wide-ranging chains of consequences, such as increased recycling reducing the demand and price of fibre wood, possibly lowering the value of forest land, and consequently giving rise to interest among forest owners to protect their forest (if there are incentives to promote protection). The biodiversity impacts from such a chain could be more extensive than could be evaluated on the basis of the use of wood alone. Carbon sequestration of forests is an additional important part of the life cycle of fibre-based products. However, at the time of conducting the case study, no scientific consensus existed on how to incorporate CO₂ removal by forests into product specific LCAs. Therefore a conventional approach was applied and no benefit from carbon sequestration was allocated to wood-based products.

Landfills as carbon sinks are an additional issue concerning wood-based products, such as newspaper and cardboard. The calculations in Papers I, II, III and IV included only emissions generated from the degradation of waste in landfills. However, studies have shown that the degradation of paper can be extremely slow and environmental conditions in landfills and the presence of lignin can limit the decomposition (Barlaz, 2006; Ximenes et al., 2008). It has been suggested that the role of landfills as a sink of biotic CO₂ in connection with paper waste, for example, should be included in LCAs as benefits for the waste remaining non-degraded in the landfill (Björklund et al., 2010; Christensen et al., 2009). Some of the LCA-models designed for assessing WM options include carbon sequestration in landfills, but some do not (Gentil et al., 2010). In practice, the option of landfilling organic waste is becoming non-available due to the legislative restrictions taking place in many countries in Europe. Theoretically, however, considering benefits for landfilled paper waste acting as a CO₂ sink would improve the environmental impacts of landfilling.

Most of the assessments in this study included assumptions made concerning the possibilities for waste based energy or materials to compensate for other types of energy and materials. However, uncertainty about the future complicates this type of modelling and therefore, the applicability of the results is limited. The operational environments are constantly changing, and currently, several strong drivers such as the Circular Economy Package (European
Commission, 2017a) and the Finnish Bioeconomy Strategy: Sustainable growth from the bio-
economy (Bioeconomy, 2017), are driving towards major changes in the European and Finn-
ish economies. For example, in the future, fuels used for energy production will increasingly
be bio-based fuels or other fuels with low emissions. Hence, compensating energy production
with waste-based energy will in the future be less beneficial to the environment. Since the
study on discarded newspapers (Papers I and II), the operational environment of WtE sys-
tems has changed significantly in Finland. Since 2007 eight incineration plants have started
operating, which are more efficient in comparison to the modelled incineration plant, pro-
ducing between 20% and 40% electricity, the rest being heat. In comparison, the incineration
plant used in Papers I and II generated 3.5% electricity and 96.5% heat. Hence, the evalua-
tion performed in the case study does not reflect the situation today concerning the waste
incineration option.

Finally, the comparisons of environmental impacts of reusable and recyclable products (e.g.
Raugei et al., 2009; Sørensen and Wenzel, 2014) are in general limited by the fact that they
do not consider the production avoided by the reuse of the product. The same limitation ap-
plies to the LCA in Paper V. Here, the reusable packaging (plastic crate) is reused 700 times
and the modelling allocates the environmental burden of the plastic crate production equally
for each 700 uses. Hence, one delivery (the FU used) only receives 1/700 of the burden of the
crate production. Nevertheless, with each reuse, the production of one new packaging could
additionally be avoided. The modelling of this avoided production and the implications of
adding it to the assessment faces challenges, since the avoided product would not be identical
to the one reused, since the durability would presumably be lower. Additionally, the modell-
ing should extend to include the end-of-life phase of the avoided product, whether it would
be recycled, or used for energy recovery or landfilling. The impacts of such additions cannot
be predicted but need to be studied.
4. Conclusions and recommendations for future research

The reviews of LCA studies on MSW or fractions of it have indicated that the performance of different WM options varies due to differences in the waste composition, system boundaries used in the modelling, technologies within the system and substituted energies among other things, but also very much due to the systems context and local specificities. Similar constraints can be recognised for C&DW management, since building materials and solutions and practices for the demolition of buildings and treatment of waste fractions vary in different regions. Hence, although several studies already exist, there is still need for national, regional and municipal level assessments of the environmental performance of WM options representing various levels of the waste hierarchy.

In this study, various Finnish WM flows and systems were assessed. The primary objectives of the assessments varied. Common to all the assessments was that they included wastes or products from the forest sector, which is one of the most important economic sectors in Finland. The case studies were analysed using life cycle based approaches such as LCA, SLCC, ELCC and MFAs. The research aimed at evaluating whether the environmental and economic results of the life cycle based approaches supported the waste hierarchy order of WM options in the Finnish context (RQ1). Moreover the aim was to assess whether the combination of life cycle based approaches provides added value for decision-making concerning Finnish WM systems (RQ2). Finally, the use of normalisation was analysed to improve the understanding of LCA results (RQ3).

The environmental or economic impacts assessment results of the case studies included in this thesis did not give clear support to the waste hierarchy order of WM options (RQ1). In case studies where landfilling was one of the evaluated options, the environmental performance of this lowest hierarchy level was worse than that of recycling or thermal treatment with energy recovery. Otherwise, the results were less unambiguous. Although reuse is on a higher level in the waste hierarchy than recycling, the delivery system using a recyclable delivery packaging (cardboard) and utilising an efficient recycling system generated lower environmental impacts than the one using a reusable one (plastic crate). Hence decisions on which material to choose for a specific use cannot be made on the basis of the material itself, but need to be supported by system level assessments. For the newspaper waste, the WtE option showed lower overall environmental impacts than recycling, but the results varied in individual impact categories. The lower costs favoured recycling of newspaper over thermal treatment with energy recovery.
In the modelling of the case studies, not all aspects related to wood and forestry were satisfactorily covered. The benefits of wood saved by recycling through carbon sequestration in forests or alternative use of the saved wood were not included in the assessments due to the lack of scientific consensus on methodologies, among other reasons. The current national and international policies that strive towards a bioeconomy and circular economy, aim at reducing the use of non-renewable resources and using renewable resources more efficiently, e.g. by cascading the use of biomass. If cascading were considered the results would probably be different. In order to verify the results, the outcome would need to be systematically assessed. Modelling biogenic processes to understand the consequences of the increasing and more efficient use of renewable resources is highly important. In Finland, and other regions with ample forest resources, methods are needed for combining WM systems for wood-based waste with forestry, especially the balances of carbon in forests and the use of wood.

All the case studies were research oriented, and did not directly aim at producing results for specific decision-making situations. However, in general, LCA results can be used in decision-making. Environmental aspects are gaining more weight, but very seldom do environmental values alone constitute a basis for decision-making. This research showed that the combined use of different life cycle based approaches can increase the usability of the results by deepening the understanding of the systems and of the critical aspects for improving the overall performance (RQ2). For the newspaper WM system the SLCC showed that when environmental costs are considered in the assessment, the thermal treatment with energy recovery of separately collected waste paper is definitely more expensive than recycling. For C&DW management the combination of MFA, LCA and ELCC identified the potential of different waste fractions for improving the overall performance of the WM system. Various fractions contribute to the performance differently, for example material recovery of the mineral fraction generated few or no environmental benefits, whereas energy recovery of wood generated high environmental benefits. Environmentally and economically more beneficial recycling concepts should be developed for both fractions to increase the recycling rate of C&DW. However, systemic environmental and economic assessments should be connected to the development of recycling solutions in order to ensure improvements in both impacts. Additionally, the joint assessment of recovery rates (by MFA), climate impacts (by LCA) and costs (by ELCC) of scenarios with varying waste compositions enabled the evaluation of the ability to achieve the 70% recycling target for C&DW in 2020. The results indicate that regional differences in waste composition may support arguments for differing recycling targets in different European regions or alternatively focusing the targets on materials, not the overall waste flow.

The understandability of LCIA results is essential to their usability for decision-making. The use of different normalisation reference values was shown to have some impact on the understandability of the LCIA results, yet overall the results remained complex to understand for an LCA non-expert (RQ3). The inclusion of documentation on the methods and assumptions used were considered important for understanding the methods and their restrictions and to support the interpretation. The challenge of communicating LCA results has been recognised broadly and to tackle it, the European Commission together with industry is developing a harmonised Product Environmental Footprint methodology. This methodology could be a
way to improve and ease the communication of LCIA results for non LCA-practitioners by using unified terminology, consistent calculations and clear output.

All the applied approaches suffered from inadequacy of data. Databases are available for process data for environmental assessments on a general level, but region specific public data are also needed. The results of this study show that basic information needs to be generated especially on the amounts, quality, processing and management of C&DW in order to support comprehensive assessment of the possibilities for increasing waste recycling and maximising the benefits. Data for economic assessments is in general more challenging to obtain than environmental data. Recently a cost model including cost data on several WM processes has been developed. However, the applicability of this data in Finland is still to be assessed.

Shortcomings in all the adopted approaches were recognised and hence the outcomes of the case studies are not unambiguous. LCAs and other life cycle based studies are always simplifications of reality and include many assumptions, which can be decisive for the outcome. In the modelling of WM systems the assumptions made on avoided impacts potentially obtained through the material and energy recovery of waste are key factors, but uncertainty about the future complicates making the assumptions. The energy sector especially is facing constant changes resulting from the increasing use of bio-based fuels and other cleaner forms of energy production. Hence, in the long run the benefits gained from compensating for fossil fuel based energy with energy recovered from waste will decrease and the performance of WtE options will decline. Simultaneously, in the circular economy the use of secondary materials in production is increasing and causing a reduction in the benefits of recycling obtained primarily from compensating the use of virgin materials. Hence, predicting the effects of such changes on the environmental performance of various WM systems is challenging and calls for comprehensive systemic assessments.
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