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

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A B S T R A C T

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

1. Introduction

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

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

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

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

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

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

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

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

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

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

2. Materials and methods

2.1. Wood polymer composites

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

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

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

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

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

2.1.2. Production technology description

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

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

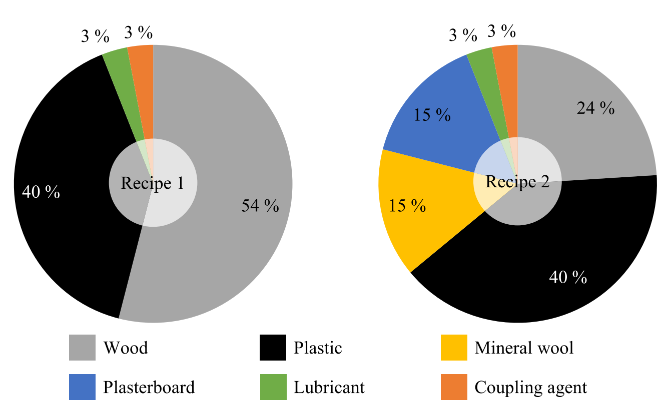

Fig. 1. WPC recipes assessed in the study (Keskisaari et al., 2016; Turku et al., 2017).

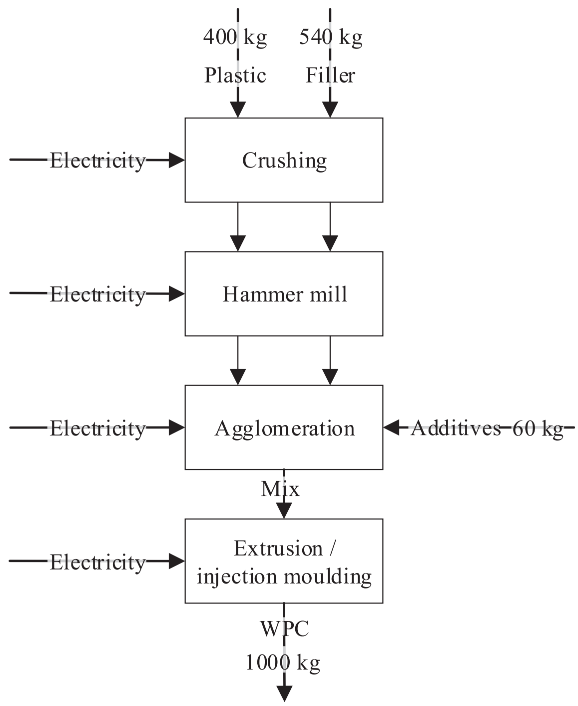

Fig. 2. WPC production process.

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

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

2.2. Life cycle assessment

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

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

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

2.2.1. Scenarios and calculation principles

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

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

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

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

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

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

2.2.2. System boundaries

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

3. Results and discussion

3.1. Contribution analysis

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

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

Table 1
LCI data of WPC production.

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

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

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

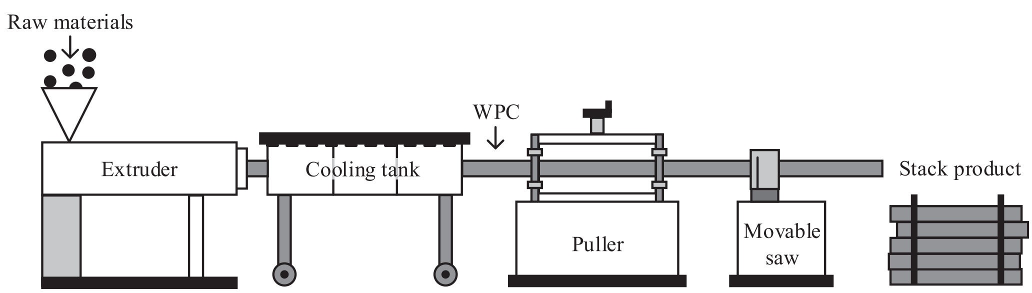

Fig. 3. WPC profile extrusion line (Wagner et al., 2014).

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

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

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

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

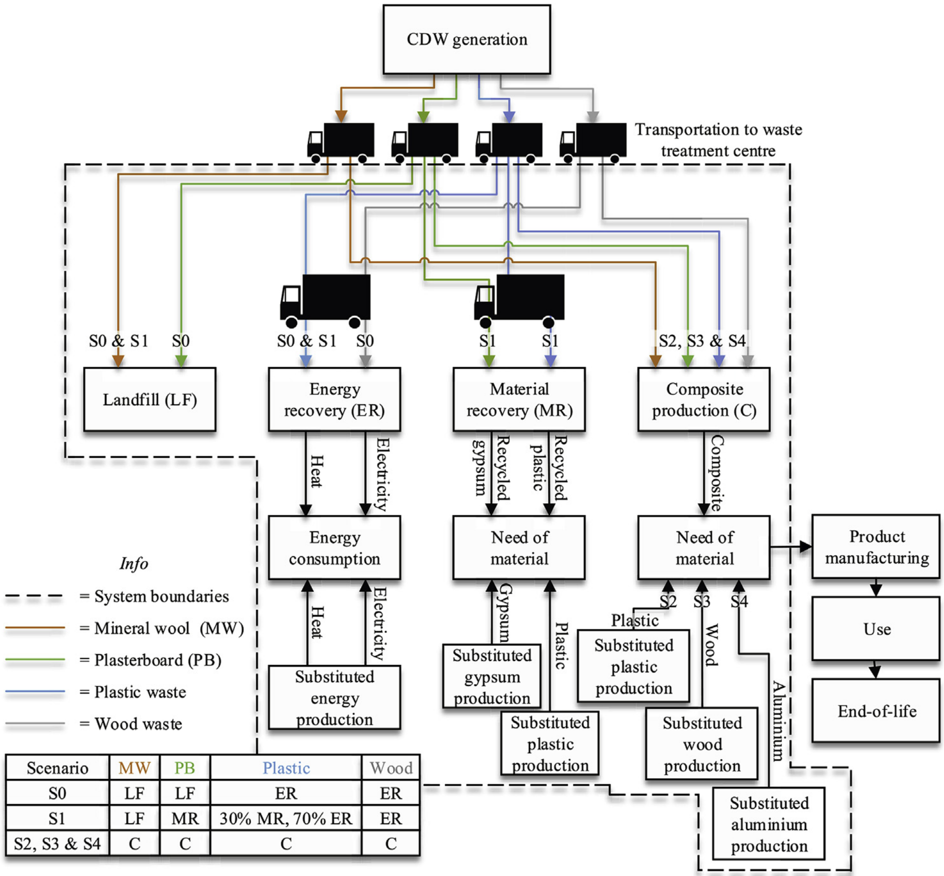

Fig. 4. System boundaries of the study.

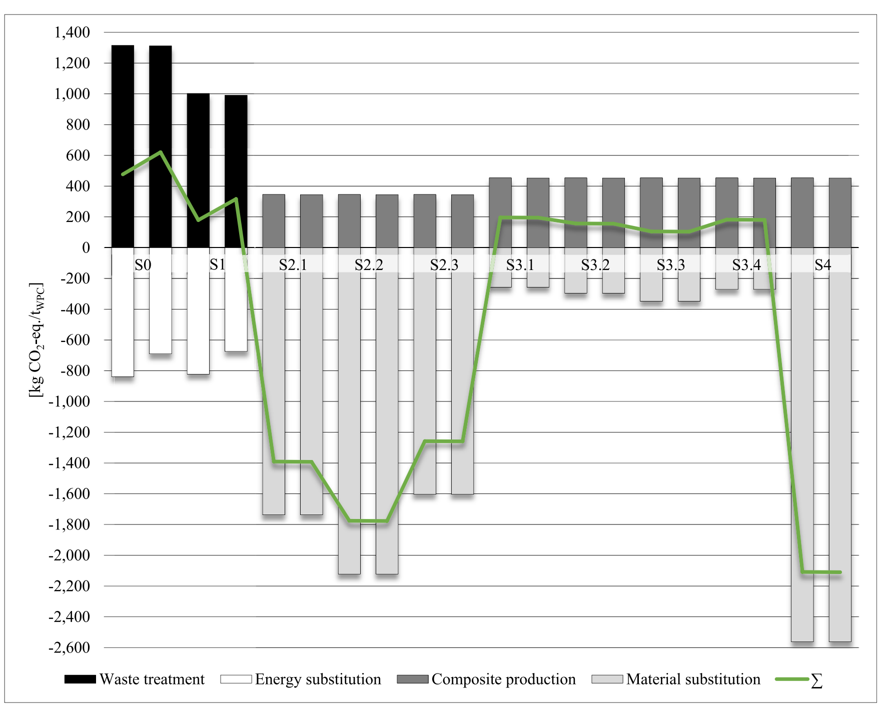

Fig. 5. Scenario contributions to climate change. The left bar in each scenario corresponds to Recipe 1, and the right bar to Recipe 2.

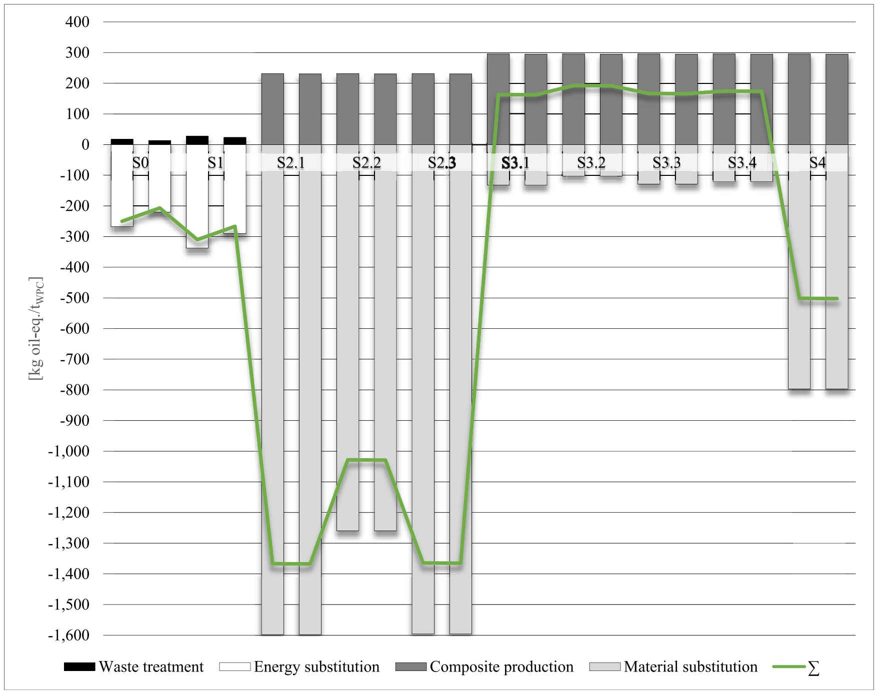

Fig. 6. Scenario contributions to fossil fuel depletion. The left bar in each scenario corresponds to Recipe 1, and the right bar to Recipe 2.

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

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

3.2. Sensitivity analysis

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

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

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

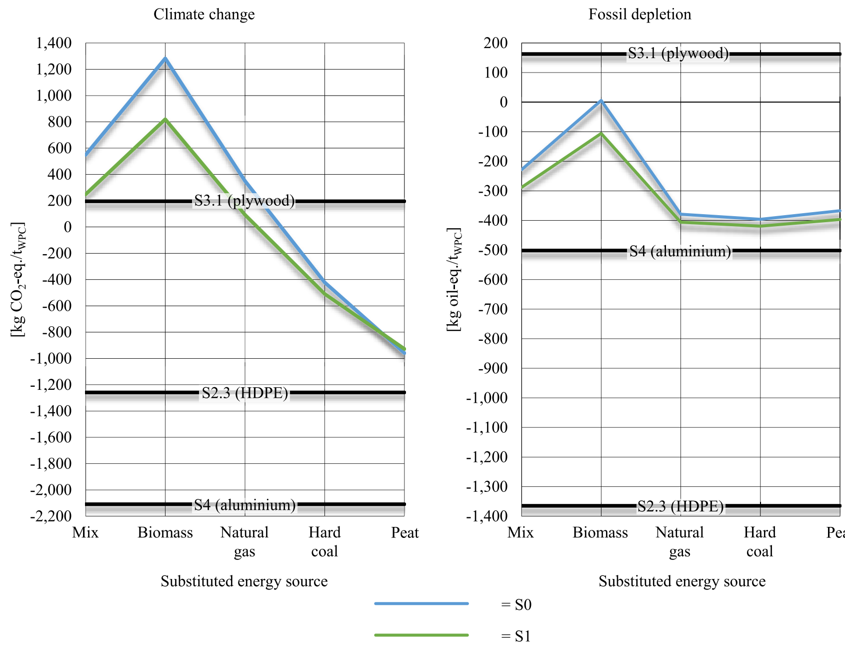

Fig. 7. The impact of the choice of substituted energy on climate change and fossil depletion results.

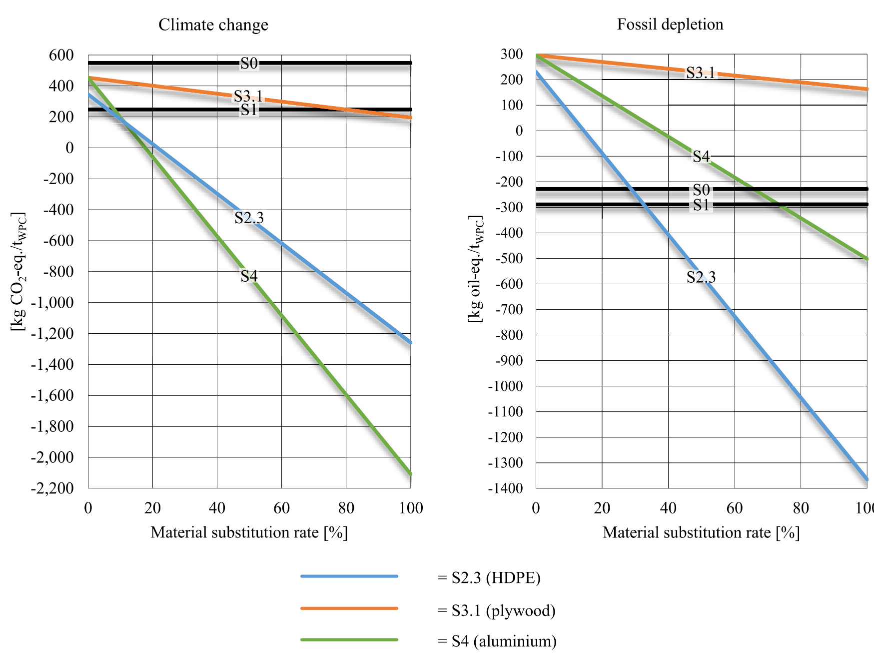

Fig. 8. The impact of the material substitution rate (0–100%) on climate change and fossil depletion.

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

3.3. Discussion

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

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

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

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

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

4. Conclusions

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

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

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

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Appendix A. Supplementary data

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

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