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An evaluation of thermoplastic composite fillers derived from construction and demolition waste based on their economic and environmental characteristics

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ABSTRACT

The use of waste is often justified by the economic and environmental benefits of their use. This study compares the use of waste materials derived from construction and demolition waste—namely wood waste, mineral wool waste, gypsum board waste, and stone cutting dust—as alternative fillers in the production of thermoplastic composites using recycled high-density polyethylene as a matrix material. In total, nine alternative composites were studied in terms of their production costs, as well as their climate change impacts in three distinct product applications. Compared with the plastic matrix, the wood fiber achieved a cheaper price of €0.8–1.2/kg and the best properties in relation to weight. The price of mineral-based fillers varied between €0.5–1.1/kg, but the effect of the higher density on the weight increased the total price of the products. The unfilled recycled plastic was the cheapest solution in the application where the covered volume was important. The impact of using recycled high-density polyethylene in composites production totals at −1.24 kg CO2-eq./kg, out of where 1.75 kg CO2-eq. is the avoided impact from avoided waste disposal and 0.51 kg CO2-eq. is induced impact from producing the composites. When also accounting for the avoided impact from the substitution of virgin high-density polyethylene with the recycled high-density polyethylene composites, the avoided impact further increases to −3.17 kg CO2-eq./kg. The mineral fillers with were preferable in the application where mass was important, however, had lower avoided impacts than unfilled polyethylene ranging between −2.06 kg CO2-eq. and −2.47 kg CO2-eq. Wood fiber filler was the preferred filler option in the application where the material properties were taken into account in the amount of required material, but resulted in the lowest cumulative avoided impacts ranging between −1.79 and −2.25 kg CO2-eq., with most of the avoided impact originating from the replacement of virgin high-density polyethylene.

1. Introduction

Recycling materials back into the economy embodies the strong potential for the reduction of manufacturing costs. It is also often seen as a means of mitigating climate change through the avoidance of using virgin raw materials. To prove that recycled materials are beneficial for the environment and for the producer, their carbon footprint and their costs should be lower than that of virgin materials.

The construction sector is one of the largest plastic consumers in Europe with a share of 19.8% of total converter demand (PlasticsEurope, 2019). The major plastic types in the sector are polyethylene, polypropylene, and polyvinyl chloride, all of which can be used in the production of thermoplastic composites. Furthermore, the construction sector is responsible for the largest waste-generating activity in the European Union, accounting for 36.4% of total waste generated in 2016 (Eurostat, 2016). Globally, the amount of construction and demolition waste constitutes

Abbreviations: LCA, life cycle assessment; LCI, life cycle inventory; LCIA, life cycle impact assessment; GWP, global warming potential; HDPE, high-density polyethylene; CDW, construction and demolition waste; r-PE, recycled high-density polyethylene; MW40, mineral wool waste composite 40% fill rate; MW60, mineral wool waste composite 60% fill rate; PB40, plasterboard waste composite 40% fill rate; PB60, plasterboard waste composite 60% fill rate; SC40, stone-cut waste composite 40% fill rate; SC60, stone-cut waste composite 60% fill rate; WF40, wood fiber waste composite 40% fill rate; WF60, wood fiber waste composite 60%.

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astonishing 3 billion tons annually (Akhtar and Sarmah, 2018). Plastic waste generated during construction and demolition activities could be combined with other construction waste—such as size-reduced wood (Sommerhuber et al., 2017), mineral wool (Väntsi and Kärki, 2014), and gypsum board (Sormunen and Kärki, 2019a) —acting as filler.

The cost of products manufactured from recycled materials is influenced by several factors, such as transportation costs, landfilling fees, recycling process, volumes, the quality of the material, gate fees, plant operating costs, and the taxation of waste (Coelho and de Brito, 2013; Duran et al., 2006). The identified measures for policymakers to influence the cost factors include green taxes, green public procurement and standardization of recycled materials (EEA, 2019). Typical applications for thermoplastic composites made with recycled material include wood-plastic composites, outdoor building, billets, profiles, and different types of cladding. The potential applications for recycled material composites are slowly expanding to include even more technically demanding applications, such as compression-molded door panels, molded with stiffening ribs, thickness variations, and molded-in holes (Gardiner, 2019).

In general, recent studies involving product or material cost calculations for alternative composite fillers are not widely available. According to Hueber et al. (2016) good cost prediction of is possible when the estimated component is part of an established family of products and sufficient historic manufacturing and cost knowledge is available. In the case of calculating the processing and material costs of waste derived fillers the historic data is not available. Bottom-up approach where the processing steps are calculated separately is usually the best option when limited information about new processing technology is available (Hueber et al., 2019). Åkermo and Åström (2000) modelled the component costs in the compression molding of thermoplastic-composite and sandwich components. Their study found that raw material cost dominated the component costs in the compression molding of composite components. They also found that compression-molded thermoplastic composites are cost-competitive for small and medium-sized components and for a small production series of large components. Bader (2002) highlighted the need for cost-performance evaluation in a comprehensive study of thermoset composites produced with a variety of manufacturing methods, but the study did not consider the use of recycled materials or thermoplastics. A study by Verrey et al. (2006) highlighted the high effect of mold-in times in thermoplastic-composite production, which can raise the total cost over thermoset molding due to longer cycle times. A modulus–cost ratio was used by Zampaloni et al. (2007), who compared compression-molded kenaf-polypropylene to kenaf-, sisal-, and coir-reinforced thermoplastics; their study did not calculate the price of the processing of materials. Zhou et al. (2019) compared the environmental and economic efficiency of compression-molded sludge cellulose plastic composite compared to traditional wood plastic composite; their study suggested that using cellulose sludge based filler was more eco-efficient than using recycled wood flour.

The environmental impacts of composite manufacturing using waste have been studied to some extent, with studies mainly focusing on a narrow range of materials with limited recipes. Liikanen et al. (2019) studied the impacts of manufacturing two composites: one recipe utilized 54% of wood waste and 40% of plastic waste while another recipe utilized 24% wood waste, 40% plastic waste, 15% mineral wool waste, and 15% plasterboard waste. Both scenarios of manufacturing composites resulted in avoided impacts on climate change and abiotic resource depletion compared with the baseline scenario of their incineration and landfilling. Other studies focusing on the environmentally-sound choice of the waste materials to be used in composite production include those of Sommerhuber et al. (2017) and Väntsi and Kärki (2015). Sommerhuber et al. (2017) compared the use of virgin and recycled wood and plastic whereas Väntsi and Kärki (2015) studied wood waste, glass fiber, mineral wool waste, recycled plastic, and virgin plastic. These studies did not analyze the economics of the raw material selection process.

Building on previously conducted research and the identified research gap, this study aims at comparing the use of several construction and demolition waste (CDW) materials as fillers in thermoplastic composites in terms of their economic and environmental performance. The specific objectives of this study were: to identify the production costs for the manufacturer, to assess the impacts on climate change from raw material provision till the production of composites, and to identify the trade-offs between the economic and environmental parameters for different groups of materials and applications.

2. Methods

This paper features a life cycle engineering study of thermoplastic composites made of waste-derived materials. Life cycle engineering represents a product development concept considering the technical, economic, and environmental characteristics of the specific product being assessed (Jeswiet, 2014). The study, however, omits the social impacts. The economic parameters were calculated using the accounting principle, considering the material provision cost and the cost of processing. The environmental impacts were calculated using the principles of life cycle assessment (LCA). Three different types of products were used to demonstrate the effect of design criteria on the best fitness of different filler materials. The volume related solution represents a typical commodity plastic application where it is used as a cover or a packaging material, the results of this example are presented in Fig. 7 for a product with 564 cm^3 of material. The mass related solution represents an application where the part is used as a counterweight, the results of this example are presented in Fig. 8 for a product with a required mass of 20 kg. The property related solution is a plate like application where the surface area is set to 0.4 m^2, but thickness can be optimized based on the material properties the results of this application are presented in Fig. 9.

2.1. The studied recipes

The composites comprised of a matrix of a filler, a coupling agent, a processing aid, and recycled high-density polyethylene (r-PE). Table 1 lists the nine recipes selected for this study, following the composition in a previous study by Sormunen and Kärki (2019). The variation in the rate of a specific filler highlights the effects of the particular filler on the results. The following assumptions regarding the recipes were made:

- High-density polyethylene is recycled post-consumer plastic, and it is used as a thermoplastic material in all composite recipes.
- Wood fiber is a size-reduced construction and demolition waste of A-grade (chemically untreated construction wood) or B-grade (chemically treated wood boards, plywood, etc. without hazardous contaminants), classified according to the study by Alakangas et al. (2015).
- Mineral wool size-reduced powder includes portions of both stone wool and glass fiber.
- Plasterboard waste is a size-reduced powder, consisting of gypsum and 3–6% of paper.

Table 1
The studied recipes of composites by mass.

Composite	Polyethylene	Wood fiber	Mineral wool	Plasterboard	Stone-cut waste	Coupling agent	Processing aid
r-PE	94%	–	–	–	–	3%	3%
WF40	54%	40%	–	–	–	3%	3%
WF60	34%	60%	–	–	–	3%	3%
MW40	54%	–	40%	–	–	3%	3%
MW60	34%	–	60%	–	–	3%	3%
PB40	54%	–	–	40%	–	3%	3%
PB60	34%	–	–	60%	–	3%	3%
SC40	54%	–	–	–	40%	3%	3%
SC60	34%	–	–	–	60%	3%	3%

- Stone-cut waste is assumed to be size-reduced particles of soft stone mining tail, such as soapstone.
- Coupling and processing aids were estimated to be consumed in equal amounts in all the compositions at the rate of 3% each.

The application of these fillers for potential applications has been studied in a review by Sormunen and Kärki (2019b).

2.2. Life cycle assessment of analyzed system

The environmental impact of the studied composites was assessed using the LCA methodology, only focusing on the climate change impacts. LCA is the most commonly used systems analysis tool in the field of waste management (Laurent et al., 2014; Pires et al., 2011) and is widely used for the assessment of the environmental impacts of products and services. The climate change impacts were assessed following (but not strictly complying with) the ISO 14040/44 standards (SFS-EN ISO 14040, 2006; SFS-EN ISO 14044, 2006), as well as the ISO 14067 standard (SFS-EN ISO 14067, 2018). The study was conducted using GaBi software (version 9.0.0.42, DP service pack 38) (thinkstep AG, 2019).

Each LCA study is typically initiated with definition of the goal and scope, a phase that determines the development of the study and defines the key research questions. Then, the studies are taken further through the collection of a life cycle inventory (LCI), the most laborious phase, required in order to collect the data for the studied product system in the form of either elementary flows or intermediate flows, supplemented with unit processes from secondary data. Once the data on the inputs and outputs of the product system are collected, the life cycle impact assessment (LCIA) phase takes place, the phase in which all the inventory data is classified into specific impact categories and characterized to be represented via a single pre-defined unit. LCIA could be expanded with normalization and weighting, which however were not included in this study. Finally, in life cycle interpretation, the results and their applicability to fulfilling the goal of the study are assessed. Each phase is described in detail in this section.

2.2.1. Goal and scope definition

The goal of this LCA study was to support a decision-making process for the selection of specific fillers and their amounts in the production of thermoplastic composites. The study is conducted for the recipes previously identified in Table 1. The function of the studied product system is to utilize waste-derived materials in the production of intermediate composite products without any intended specific application. Thus, the functional unit was set to 1000 kg of the manufactured composite.

The system boundaries are shown in Fig. 1. The impacts in this study are assessed, starting from the provision of waste running through to the composite production process (i.e., it adopts the so-called zero burden approach) (Ekvall et al., 2007). The waste separation facility receives different fractions of the construction and demolition waste from various operators. The impact from the collection of waste and its possible separation are not included since it is expected to occur in both scenarios that are compared. Furthermore, the possible impact from additional separation being required to produce composites is expected to have a negligible impact on the results overall.

Once the waste has been received and possibly separated, it is sent either for landfilling or incineration in the baseline scenario, which represents the business-as-usual case in Finland. Because the landfilling of waste containing more than 10% organic carbon is banned in Finland, only mineral wool, stone-cut, and plasterboard waste were modelled as being landfilled. For when plasterboard is landfilled, a 4% share of paper (Jiménez Rivero et al., 2016) was modelled using the process for the landfilling of paper that generates small amounts of landfill gas, thus giving it a credit through electricity substitution. The remaining waste fractions, namely polyethylene and wood, were modelled as being incinerated with energy recovery. The electricity and thermal energy generated were modelled to replace the average electricity and heat profiles of Finland through system expansion.

In the alternative product system, the waste was utilized in the production of composites. In this study, the composite production plant was located in the same place as the waste separation facility. Thus, there was no need for the transportation of waste to the place of production of the composite. The same machinery as that which loads the waste onto the trucks in the baseline scenario was used to load the waste into a crusher, so the impact from the loading operations was excluded. After crushing, the waste is milled and sent to agglomeration, where it is mixed with a coupling agent and a processing aid. Finally, the composite products are formed using the injection molding process. The intermediate composite products were modelled to replace the average molded products made of HDPE.

2.2.2. Life cycle inventory

The LCI in this study largely follows a previously conducted study by Liikanen et al. (2019). In the baseline scenario, the landfilling of inorganic waste was modelled using the process “EU-28: Inert matter (unspecific construction waste) on landfill ts.” The landfilling of paper from plasterboard was modelled via the process “EU-28: Paper waste on landfill ts <p-agg>.” The “p-agg” part of the name of the unit process indicates a partly terminated system. The incineration of wood waste and plastics were modelled using the processes “EU-28: Waste incineration of untreated wood (10.7% H2O content) ELCD/CEWEP <p-agg>” and “EU-28: Waste incineration of plastics (PE, PP, PS, PB) ELCD/CEWEP <p-agg>” respectively. The mass of waste landfilled or incinerated equaled 940 kg in each scenario, which is the mass of waste otherwise required to produce 1000 kg of composite products. The masses of waste disposed and energy generated in the baseline scenario are shown in Table 2.

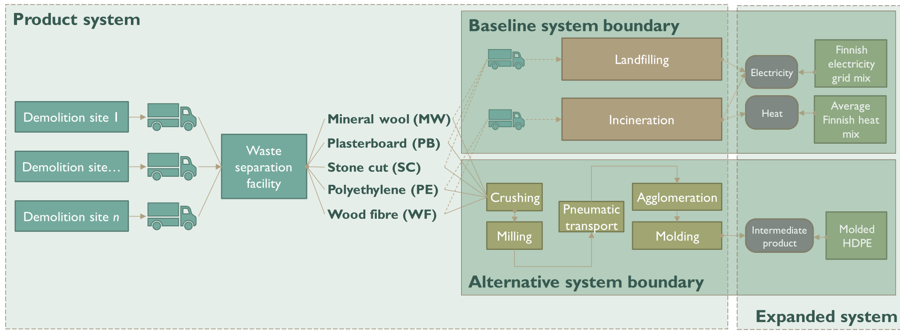

Fig. 1. The product system, expanded system, and the system boundaries of the baseline and alternative scenarios.

Substituted electricity was electricity supplied from the Finnish grid mix, modelled using the process “FI: Electricity grid mix ts.” The Finnish average heat supply was modelled using the statistics on the Finnish district heat generation described in the work of Liikanen et al. (2019).

The alternative scenario of composites production was modelled using the inventory of Liikanen et al. (2019) with the following several variations to the data used. First, the electricity consumption for crushing and milling soap stone was modelled in the same way as for plastic waste. Second, the composite production process was changed from extrusion to injection molding using the unit process “GLO: Plastic injection molding (parameterized) ts <u-so>.” The default energy consumption of injection molding is 4.5 MJ/kg injected product, which is higher than that of the extrusion process of 1.8 MJ/kg product used by Liikanen et al. (2019). Elduque et al. (2018) reported the electricity demand for the injection molding of plastic from Ecoinvent as 5.2 MJ/kg molded part and the value of 3.0 MJ/kg molded part, which were derived from laboratory tests. Third, the share of the coupling agent and the processing aid were increased from 0.5% to 3% each. Lastly, the work of Liikanen et al. (2019) was expanded by including the injection molding of HDPE using the same process as for molding of composites: “GLO: Plastic injection molding (parameterized) ts <u-so>.”

2.2.3. Life cycle impact assessment

Owing to the complementary nature of this LCA study, only the impacts on climate change were studied. The limitation of using only one impact category is addressed in the Discussion section. The climate change impacts were characterized using the characterization factors developed by the Intergovernmental Panel on Climate Change (IPCC, 2014) and implemented in GaBi software as “IPCC AR5 GWP 100, excl. biogenic carbon.” The results are calculated using carbon dioxide (CO_2) as an impact category indicator, thus giving the results in kilograms of carbon dioxide equivalent (kg CO_2-eq). The carbon uptake during the growth of the wood used in the process has not been accounted for as carbon sequestration, neither was its release during incineration considered to have a climate change impact.

Table 3
The density and estimated price range for material components.

Material	Density ρ (kg/m^3)	Unit cost (€/ton)
Mineral wool	2500–2700	−250 to −50
Gypsum waste	2400–2570	−100 to −50
Wood waste	900–1150	50–100
Soapstone, fine cutting waste	2800–3130	100–150
Polyethylene, recycled	900–970	500–700
Coupling agent	900–1000	4000–5000
Processing aid	900–1000	4000–5000

2.2.4. Sensitivity analysis

The sensitivity analysis was conducted by changing the parameters affecting both the composites production process and the substitution of the products. The impact of electricity on the impacts of the composites production is expected to be high, so the electricity source was changed from the Finnish electricity grid to electricity from natural gas, which is still a commonly used fuel in Finland for electricity generation (Suomen virallinen tilasto (SVT), 2019), and to solar electricity production to reflect a possible situation of green electricity procurement by a manufacturer. As per the impacts from the substitution, a substitution ratio was varied by 20% as to reflect for possible difference in properties of the produced composites versus products made of virgin plastic.

2.3. Economic analysis

The costs are calculated for the production phase including preproduction activities: labor wages, and energy and material consumption. The expenditures related to capital costs—such as machinery and tooling costs, as well as the maintenance of equipment—were not calculated as they were estimated to be similar regardless of the filler used. The range of the costs was calculated for each composite based on the lower and upper boundaries of the expected material cost and density seen in Table 3. The negative cost values represent gate fees to the waste producer. The range of material unit cost includes the cost of transportation from the demolition site. The unit cost values for material and processing are based on the market prices in Finland. The following equations do not include the unit transformations between volume and mass.

Table 2
The mass of the waste disposed and energy generated in the baseline scenario, depending on the studied composite recipe.

	Unit	r-PE	WF40	WF60	MW40	MW60	PB40	PB60	SC40	SC60
Waste landfilled	kg	0	0	0	400	600	400	600	400	600
Waste incinerated	kg	940	940	940	540	340	540	340	540	340
Electricity generated and substituted	kWh	1380	997	806	791	498	793	501	791	498
Heat generated and substituted	MJ	16 300	11 700	9430	9370	5900	9370	5900	9370	5900

The estimations assume that the tooling is fully utilized, and the production is continuous.

The total cost (C_T) of processing was calculated as a sum of the material cost (C_M) and the processing costs (C_P). The material cost of a composite is estimated by using equation (1):

$$C_M = C_{Ma} + C_F + C_{CA} + C_{PA}, \tag{1}$$

where.

- C_{Ma} is the cost of the thermoplastic,
- C_F is the cost of the filler,
- C_{CA} is the cost of the coupling agent,
- C_{PA} is the cost of the processing aid.

The processing cost of a composite is estimated by using equation (2):

$$C_P = C_A + C_D + C_C + C_{CM}, \tag{2}$$

where.

- C_A is the cost of agglomeration,
- C_D is the cost of drying,
- C_C is the cost of size reduction and sieving,
- C_{CM} is the cost of compression molding.

The drying cost (C_D) was calculated for wood, gypsum, and mineral wool. The cost of size reduction and sieving (C_C) were calculated for all the waste fractions. The processing cost (C_P) for the coupling agent and processing aid were only calculated in the agglomeration phase. The calculation for phases C_A, C_D, and C_C were done according to equation (3). All the cost calculations employ the rule-of-mixture to calculate the minimum, average, and maximum density of a composite from Table 3. The processing cost values are presented in Table 4.

$$C_P = (w_i \cdot \rho_i \cdot C_i) / \rho_c, \tag{3}$$

where.

- w_i is the share of a material in a composite,
- ρ_i is the density of a material in a composite,
- ρ_c is the density of the composite,
- C_i is the unit cost of processing (see Table 4)

The calculation of the composite compression molding cost (C_{CM}) was done according to equation (4):

$$C_{CM} = \rho_c \cdot C_i, \tag{4}$$

where the density of the composite ρ_c was calculated according to equation (5):

Table 4
The processing unit costs used in the calculation.

Production	Cost (€/ton)
Crushing and milling	40–50
Drying	20–30
Agglomeration	100–150
Compression molding	450–550

$$\rho_c = 1 \Big/ \left(p_\% \big/ \rho_p + f_\% \big/ \rho_f + ca_\% \big/ \rho_{ca} + pa_\% \big/ \rho_{pa} \right), \tag{5}$$

where.

- ρ_p is the density of a polymer (see Table 3),
- $p_\%$ is the share of a polymer in a composite (see Table 1),
- ρ_f is the density of a filler (see Table 3),
- $f_\%$ is the share of a filler in a composite (see Table 1),
- ρ_{ca} is the density of a coupling agent (see Table 3),
- $ca_\%$ is the share of a coupling agent in a composite (see Table 1),
- ρ_a is the density of a processing aid (see Table 3),
- $pa_\%$ is the share of a processing aid in a composite (see Table 1).

The property index corrected total cost C_{TM} was calculated for the minimum and maximum value of C_M and C_P with equation (6):

$$C_{TM} = PU \cdot \rho_c \cdot (C_M + C_P) \cdot i_E, \tag{6}$$

where.

- PU is the volumetric production unit in m^3,
- is the index value based on the corresponding amount of material compared to the base value of r-PE calculated with equation (7).

The sensitivity analysis was conducted by calculating a range based on the variation of material weight and estimated price fluctuation. Life cycle costing usually also accounts for research and development, use, maintenance, and end-of-life costs, but these were left out of the scope for being dependent on the product application.

The technical analysis of filler composites is based on the results of a study by Sormunen and Kärki (2019) about the properties and densities of these materials when processed by compression molding. To evaluate to the reinforcement efficiency of fillers an index value i_E was calculated based on the corresponding amount of material compared to the base value of r-PE with the equation. The reinforcement efficiency i_E is calculated referring to r-PE due to its use as a material to be reinforced in the technical analysis by Sormunen and Kärki (2019). The application examples in this study are dimensioned according to Young's modulus (E) of the material. The calculated values for i_E are presented in Table 5. The smaller the value for i_E the greater the reinforcement property of the filler.

$$i_E = E_{HDPE} / E_{composite}. \tag{7}$$

3. Results

The results of this study are presented in their respective parts: environmental impacts and economic assessment.

3.1. Environmental impacts

The environmental impacts in this study are shown in a comparative way reflecting, first, the impacts of avoided disposal on the waste and provision of electricity and heat in the same amounts as derived from waste; second, the impacts of the production of composites from waste; and third, the impacts of the substitution of products on the market with composites.

3.1.1. Avoided impact from conventional disposal

Fig. 2 shows the results of the avoided disposal of 940 kg of

Table 5
The calculated indices for materials.

Index	r-PE	WF40	WF60	MW40	MW60	PB40	PB60	SC40	SC60
i_E	1.00000	0.62252	0.52809	0.81034	0.68613	1.05618	0.56970	0.80342	0.50538

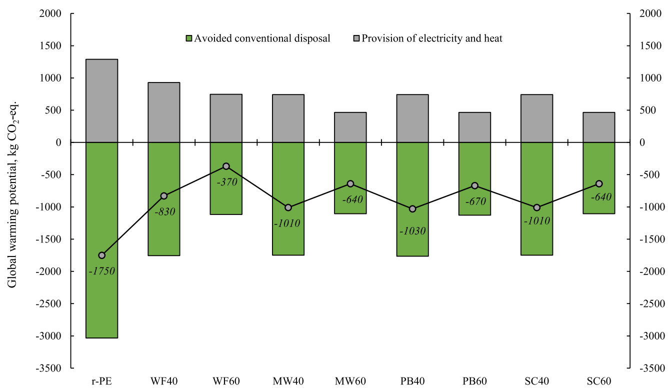

Fig. 2. The global warming potential of the avoided conventional disposal of 940 kg of waste and the provision of electricity and heat generated during conventional disposal.

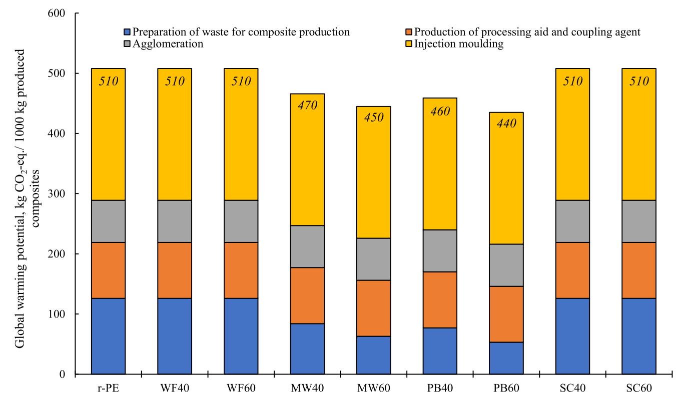

Fig. 3. The global warming potential of manufacturing 1000 kg of intermediate composite products.

waste and the impact originating from the provision of electricity and heat (see Table 2 for the amounts). The avoided incineration of polyethylene (unfilled recycled plastic [r-PE]) waste results in the largest avoided impacts (−1750 kg CO_2-eq) compared with the other scenarios. This is explained by the high emission factor of the incineration of plastics and the relatively low carbon intensity of the Finnish electricity grid and heat mixes that were used to model electricity provision if the plastic is not incinerated in order to maintain the same functionality in the study. When the mass of plastic incinerated decreases to 540 kg (i.e., scenarios WF40, MW40, PB40, and SC40), the avoided impact shrinks to −830 to −1030 kg CO_2-eq and further to −370 to −670 kg CO_2-eq when the mass of plastic incinerated decreases to 340 kg (i.e., scenarios WF60, MW60, PB60, and SC60).

Comparing the impact from different waste streams, avoiding the incineration of wood (WF60 and WF40) is seen as the option with the lowest avoided impact. This is because the impacts from wood combustion on climate change are lower than those of the average Finnish electricity grid and heat mix. There was no significant difference between the avoided impacts of soap stone, mineral wool, and plasterboard as all those waste streams are inorganic and are landfilled in the same way. The slight difference for the plasterboard is due to the low share of paper in it.

3.1.2. The impact from composite production

Fig. 3 breaks down the impacts associated with the production of composite materials (i.e., attributional cradle-to-gate impacts). The impact of the production of composites ranged from 440 to 510 kg CO_2-eq across all scenarios. No difference was expected between the impacts of producing composites from plastic, wood, and soap stone waste (scenarios r-PE, WF40, WF60, SC40, SC60). The impacts were the same because the expected energy consumption for their pre-treatment is the same. A slightly lower amount of electricity was required for the preparation of mineral wool and plasterboard, accounting for their lower strength compared to the other waste materials.

Considering the impacts of the separate phases of the production process, it can be seen that injection molding has the largest contribution to the results (43–50%). This is related to the high energy demand of the process required to melt the materials. However, there is a large variation in the energy consumption of injection molding based on the desired product. Therefore, the specific energy demand should be clarified for the specific product being manufactured. The production of the coupling agent and the processing aid, with the shares of 3% each from the total mass of composites produced, accounts for nearly a fifth of the impacts on climate change. However, the share of the additives can be as low as 0.5% (Liikanen et al., 2019).

3.1.3. Cradle-to-gate impacts

Fig. 4 aggregates the results of the impacts on climate change from the studied product systems, accounting for the changes in the waste management system, as well as the potential substitution of products on the market with waste-derived composites. Accounting for a constant avoided impact from plastic products made of HDPE of 1920 kg CO_2-eq, the largest avoided impact out of all the recipes studied is achieved when utilizing the largest amounts of plastic waste, which was −3170 kg CO_2-eq per 1000 kg of composite produced. Herein, more than half of the avoided impact originates from the replacement of plastic products made of virgin

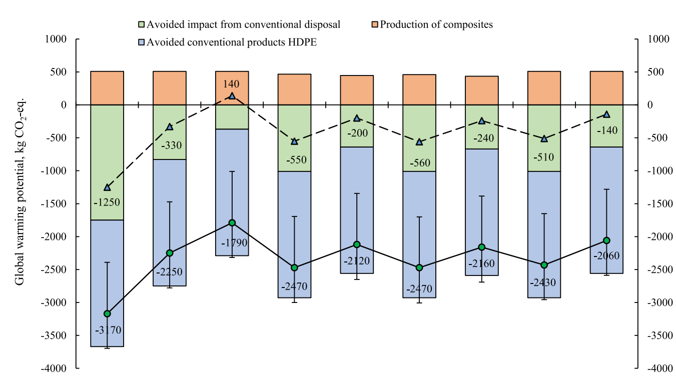

Fig. 4. The global warming potential of the product system utilizing waste for the production of 1000 kg of composite, substituting molded products made of virgin HDPE on a 1:1 mass basis. The dashed line sums up the impacts from avoided conventional disposal and only includes composites production whereas the solid line also accounts for the avoided impact from the substitution of HDPE products.

HDPE. The lowe

st avoided impact of −1790 kg CO_2-eq originates from the scenario utilizing wood the most (i.e., WF60). If no plastic were replaced, then the cumulative impact in scenario WF60 would be 140 kg CO_2-eq, the only scenario with induced impact if the substitution of plastic products is not considered.

The sensitivity analysis showed that the electricity and substitution of virgin plastic have a high impact on the results. However, the cumulative impact across all scenarios studied did not exceed the impact from the baseline scenario: the maximum reduction of the climate change impacts achieved in the scenarios is reduced by 25–44% in the worst case when the electricity is supplied from natural gas and only 0.8:1 substitution rate is achieved. When supplying the electricity from solar energy and substituting plastic on a 1.2:1 substitution rate, the increase in the environmental impact is in the range 17–29%.

If the electricity supply to the composites production was from natural gas and there were no substitution of plastic at all, then the scenarios WF40, WF60, MW60, PB60, and SC60 would have a higher impact from the composite production compared to the avoided impact from conventional disposal. The minimal break-even substitution rate of plastic in those scenarios is 4% (WF40), 34% (WF60), 12% (MW60), 10% (PB60), and 17% (SC60).

3.2. Economics

The total cost (C_T) and impact on climate change for composite manufacturing is presented in Fig. 5. More than half of the avoided impact originates from the replacement of plastic products made of virgin HDPE. The use of recycled filler material in a plastic matrix decreases the kilo price of the composite. On average, the use of wood filler decreases the cost of the r-PE plastic, by 13% in WF40 and by 19% in WF60. The results indicate that the recipe with the highest costs, namely r-PE, has the highest avoided impacts of −3.17 kg CO_2-eq per kilogram of composite. The decrease in cost compared to r-PE is 21% in MW40, 20% in PB40, and 16% in SC40. The drop in cost compared to r-PE further increases with higher filler rates to 31% for MW60, 29% for PB60, and to 25% for SC60. The mineral-based composites of equal filler rate have similar total cost despite the differences in material cost. The avoided climate change impacts do not vary significantly between mineral composites of equal filler rate.

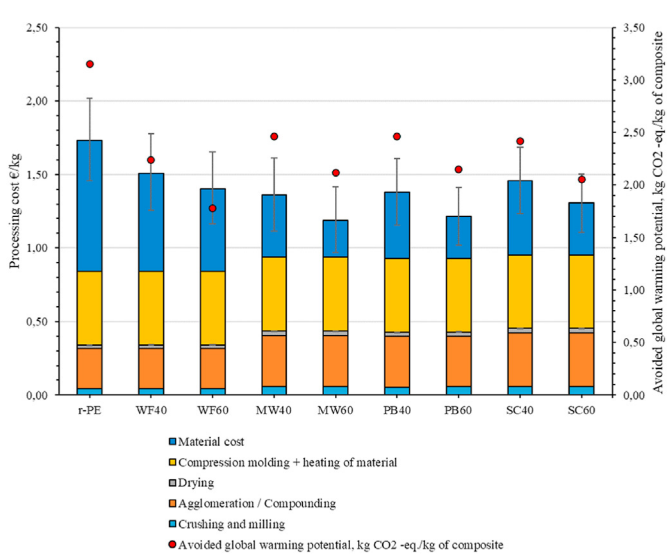

Fig. 5. The total cost (€/kg) of compression-molded composite material and the division of processing costs. The tolerance bar shows the variance in total cost.

Fig. 6 shows the price per volumetric product unit (in m^3) for different composites. When i_E is taken into consideration, the total cost of a product unit decreases in relation to property improvement. The result suggests that when possible the property improvement should always be taken into consideration in the product dimensioning to gain a cost advantage of waste filler materials. If the replacement of material is done on a one-to-one basis, there is no expected cost advantage in mineral composites due to expensive pre-processing activities.

The volume-based application has a set of space that the application needs to fill. In this case, the lower total cost (C_T) per kilogram of mineral waste–based composites offer no benefit as the material properties cannot be taken into consideration. In the volume-based application, the total avoided impacts on climate change per product are higher in the mineral waste–filled composites; this is due to a higher mass of the product with mineral waste–based composites. The comparison of composites in the volume-based product example is shown in Fig. 7. The calculation was made for 564 cm^3 of material. The cheapest solution is achieved with filler combination WF60, which decrease the total cost of the composite while having a low avoided impact see Figs. 4 and 5.

The mass-related solution presents an application where the product needs to have a certain weight. The example was calculated for 20 kg of material, which could represent a product application such as counterweight. Lower C_T favors the mineral fillers over the r-PE, due to their high density and low cost per kilogram. From the climate change point of view, the product design favors the unfilled plastic because of the high avoided impact per kilogram of r-PE as larger volume of HDPE is replaced with recycled material. The comparison of composites in the mass-based product example is shown in Fig. 8.

In the last example, a property-optimized profile has been calculated. Here the property index i_E is used to take the properties of the composite into consideration in material usage. The height of the profile can be optimized while keeping the needed surface area 0.4 m^2. The wood fiber–filled composite WF60 is the cheapest solution for the studied materials for this product. The superiority of wood fiber filler compared to other alternative fillers in reinforcement can be seen in quite clearly from the results in Fig. 9. The wood fiber filler does not significantly increase the mass of the composite but affects the modulus more effectively than the mineral-based fillers such as minerals or gypsum. Less raw material can be used produce the profile with the lighter WF40 and WF60,

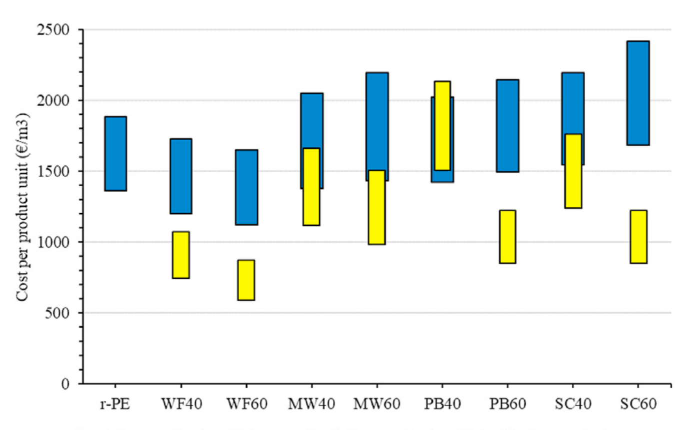

Fig. 6. The cost per product unit of compression-molded material with property index (i_E) values.

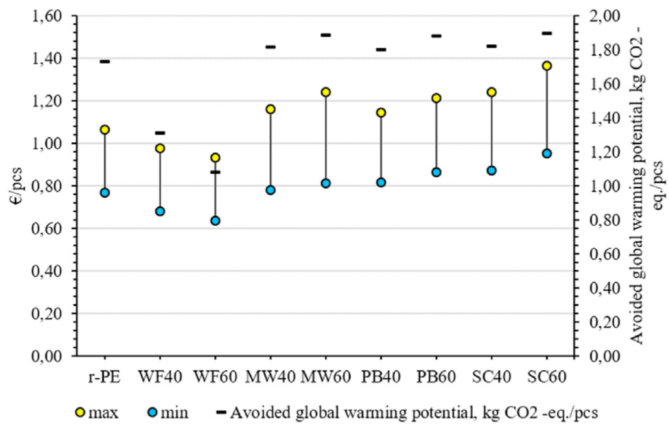

Fig. 7. Example of a volumetric application's (564 cm^3) unit costs.

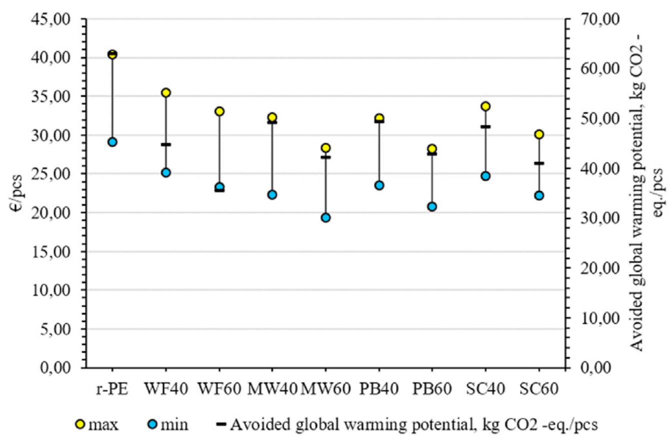

Fig. 8. An example of a mass-based (20 kg) product unit costs.

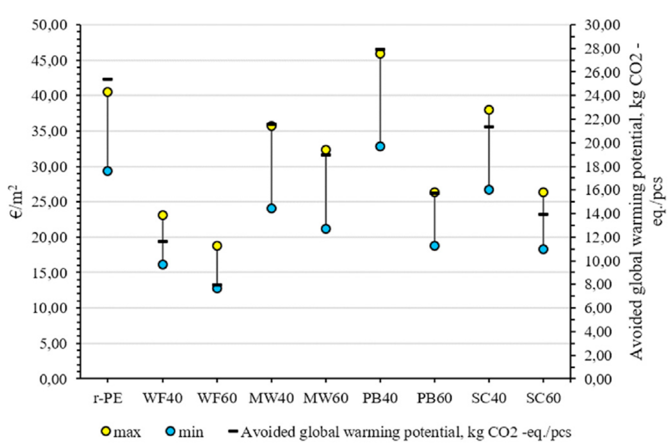

Fig. 9. An example of the property-optimized profile 0.4 m^2 costs per m^2.

therefore the avoided impact from HDPE substitution is also smaller.

4. Discussion

The climate change impacts of producing 1 kg of composites in this study ranged 0.44–0.51 kg CO_2-eq which is line with the results of Liikanen et al. (2019) indicating the impact of 0.4–0.5 kg CO_2-eq. Somewhat higher impact of 0.76–0.78 kg CO_2-eq was reported by Sommerhuber et al. (2017) where waste plastic and waste wood were utilized. In the study by Väntsi and Kärki (2015), however, the impact from the production phase cannot be distinguished from the cumulative impacts presented in the study, thus eliminating cross-comparison. When virgin plastic was used in the production of composites, the impact from the raw materials provision stage was roughly 1.5 kg CO_2-eq per 0.68 kg of plastic used (Sommerhuber et al., 2017) or 2.2 kg CO_2-eq per kg of plastic used. However, in this paper, the impact from virgin plastic production was modelled in this paper as avoided impact at the level of 1.9 kg CO_2-eq per kg of replaced HDPE. If no plastic were substituted, the cumulative impact from composites production was higher than the avoided impact from conventional disposal in the scenario WF60. According to the study by Liikanen et al. (2019), the minimum substitution rate of 6% was required to achieve lower impact from composite production than the production of the substituted material.

The potential environmental benefits were evaluated using only one impact category of climate change omitting a range of other areas of environmental concern. However, Liikanen et al. (2019) assessed a wide range of impact categories and showed that the environmental impact was reduced across the majority of impact categories when wood plastic composite replaced virgin plastic, namely for climate change; fossil depletion; freshwater and marine ecotoxicities; freshwater, marine, and terrestrial eutrophication; human toxicity; and stratospheric ozone depletion potentials. In most cases, the impact reduction was dominated by the avoided impacts of virgin plastic production. Another limitation of this study relates to the narrowed system boundary, which starts with the provision of the waste for the production process and ends at the gate of the composite production facility. The zero-burden approach that was utilized in this study implies avoidance of the debit associated with the use of waste in the process while allocating all the impacts from the previous life cycles to the products that resulted in the generation of waste. Such an approach gives substantial benefits to waste materials compared to virgin ones, yet this approach is seen as obsolete in the era of the circular economy where waste should be phased out gradually and products should be designed for recycling (Djuric Ilic et al., 2018). Also, when limiting the system boundaries to the gate of the factory, the impact from the end of life, from disposing of the composites and alternative materials, remain uncertain. Owing to the varying share of plastic in composites, the impacts from its disposal are expected to change significantly, especially if incinerated.

The cost calculation in this study present extreme cases where the processed material weight and the material price were either at their highest or lowest resulting in a range. To evaluate the cost of preprocessing the densities in Table 3 represented waste in its processed state. The low-density materials such as mineral wool (70 kg/m^3) use the same machine capacity as the high-density materials but produce lower amount of processed filler material in kilograms. Using the density in processed form takes this fact into consideration by calculating the processing cost evaluated in €/kg. Clear consensus is missing on how waste processing cost should be calculated, but most used method seems to be €/ton, which was also used in this study. A more accurate product specific cost calculation can be done by taking into consideration the geometrical complexity of a part for which production time and tooling costs for a given quantity of products can be estimated (Hagnell and Åkermo, 2015). The manufacturing cost also depend on the used material heating method isothermal (heating the tool and the surroundings) or non-isothermal (heating the material directly) (Pantelakis et al., 2009). However, this study shows a general direction of costs because the material plays a major part in thermoplastic product molding costs. The application of novel materials also requires practice and small series production is more likely to suffer from a high portion of rejects and disruptions in processing before acceptable quality is achieved. It is also probable that the processing cost variable (C_{CM}) would variate between different waste fillers in its subprocesses like heating, tool closing time, cooling time, inspection, dimensional stability control, etc. The composite agglomerate from recycled materials price ranges were calculated to be: €1.1–1.5/kg for r-PE, €0.9–1.2/kg for WF40, €0.8–1.1/kg for WF60, €0.7–1.1/kg for MW40, €0.5–0.90/kg for MW60, €0.8–1.1/kg for PB40, €0.6–0.9/kg for PB60, €0.8–1.1/kg for SC40, and €0.7–1.0/kg for SC60. The price of WF40 and WF60 suggest that reprocessing recycled CDW wood has the potential for providing cheaper fiber source as commercial wood plastic granulates cost €1.00–4.00/kg (Dammer et al., 2013). The price of composite agglomerates were not far from the virgin polyethylene prices €0.9–1.2/kg. The waste derived composites should have lower cost than the virgin HDPE for the replacement potential to turn into reality. The economic benefits of recycling are further influenced by the oil-prices that at the moment of writing have taken a drastic drop as low as 20$ per barrel due oil price war between Russia and Saudi Arabia. The cost calculation for composites made with recycled materials are quite rare, although lower costs are almost always mentioned as incentives for their use. Gu et al. (2016) used performance -material cost analysis to evaluate the use of injection molded recycled polypropylene composites and found them having a cost of 50% lower than the ones with virgin polypropylene. In a paper by Keskisaari and Kärki (2018) the composite manufacturing cost with waste fillers consisted approximately 50% from material and 50% of processing. Their paper did not evaluate the influence of material density to the processing, but the range of total cost was similar to the one in this study €0,9–1,2/kg. The processing cost was higher in this study compared to the studies of Gu et al. (2016) & Keskisaari and Kärki (2018), which is explained by more efficient processing methods extrusion and injection molding.

To benefit fully of filler materials requires the optimization of the product volume taking into consideration the reinforcement ability of the filler. This is not often possible as products usually have volumetric requirements. The example in Fig. 7 shows that heavy waste derived fillers does not lower the cost of a volumetric product due to the influence of density in volume cost. Fig. 8 shows a product type that favors waste derived mineral fillers this could be a traffic sign holder or similar where the part needs to have mass. The maximum volume of the part and cheaper material alternatives are limiting this type of application for recycled material. The structure optimization that was shown in Fig. 9 is limited by the filler particle size, and minimum and maximum wall strength by the processing method (with the studied waste fillers this would be 2–10 mm compression molding).

5. Conclusions

Recycled mineral fillers offer the opportunity to stiffen recycled polyethylene in non-structural applications. Wood filler has the best strengthening effect of the studied types when properties are compared relative to the increase in weight. Plasterboard and soapstone seem to be more cost-efficient than mineral wool in equal weight-based filling percentages. In volume-based applications, recycled wood filler is the most efficient filler material of the studied group. It also decreases the cost of the product in comparison with unfilled recycled polyethylene. In weight-based applications, the composites with 60% mineral filling (by weight) offer the most cost-efficient solution of the studied group; however, the drop in cost is relatively small in comparison to wood filling, therefore, using wood filler is more risk free for the manufacturer as the wood plastic composite solutions have a long use and production history. In property-optimized profiles, wood fiber fillers are clearly the best option from the studied group due to their reinforcement capabilities. PB60 and SC60 could prove to be better than WF40 and WF60 if other technical properties, such as moisture resistance, are taken into consideration. The impacts of producing thermoplastic composites from CDW are significantly lower than the avoided impacts from their conventional disposal (i.e., incineration and landfilling), especially for plastic waste. The production of composites has a high potential for the mitigation of climate change when accounting for the substitution of conventional products made of virgin plastic. The use of wood waste in the production of composites has lower benefits compared with that of plastic waste due to lower avoided impacts from their conventional disposal. The limitations of the research relate to its focus on the direct costs of processing in a northern European operating environment. The investment costs related starting the activities such as tooling, machinery, surrounding infrastructure, and local differences were out of the scope of this study. The results should be considered when looking for new open-cycle applications for recycled materials. The reuse of the material does not necessarily create a cost advantage or the desired reduction in CO2 emissions due to the required pre-processing or material properties. Further research could address product-specific costing for applications with CDW filler composites, in which case the tooling costs, investment costs, and the supply chain could be considered in the cost and environmental impacts modelling. Also, calculations of the climate change mitigation possibilities from the production of composites on a nation- or region-wide scale accounting for a variety of waste feedstock could be calculated to indicate a scale of the proposed solution in the climate change mitigation targets.

CRediT authorship contribution statement

Petri Sormunen: Conceptualization, Methodology, Formal analysis. **Ivan Deviatkin:** Conceptualization, Methodology, Formal analysis.

Declaration of competing interest

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

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