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Title page

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Abstract

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

Keywords: circular economy, life cycle assessment, life cycle costing, global warming potential, agricultural plastic waste, recycling

Acronym list

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

1. Introduction

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

The intensive use of plastic in the agricultural sector creates a waste problem. Finland generates 12000 tons of agricultural plastic waste annually, dominated by bale wrap comprising around 7000 tons (Alenius, 2016). These plastic film wastes are disposed of in various ways such as landfilling (70%), open field burning (10%), energy recovery (10%), and recycling (10%) (Briassoulis et al., 2013; Erälinna and Järvenpää, 2018). The lack of proper management can be caused by difficulties in handling different types of plastic waste, impurities, and contaminants. Thickness is the main parameter for recycling plastic films, and multiple types of waste plastic films can result in mixed thickness that risks compositional uniformity (Briassoulis et al., 2013). Thus, source-separated collection or commingled collection with a proper sorting system becomes necessary. A thorough washing process is required to deal with impurities such as soil, organic material, dirt or metal (Briassoulis et al., 2013).

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

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

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

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

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

2. Literature overview

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

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

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

Besides addressing the collection issue, CLSC on plastics also focused on the environmental impact of recycling and the environmental benefit obtained from recycled material. Wäger & Hischier (2015) and Hou et al. (2018) evaluated the environmental performance of plastic recycling compared to other plastic waste treatments. Both studies showed that recycling was a better option than incineration or landfilling; moreover, recycling provided environmental savings due to avoiding virgin material production. The extent of environmental savings derived from recycling depends on the substitution factor. Simões, Xará, & Bernardo (2011) compared the environmental impact of anti-glare lamella (AGL) made of virgin high-density polyethylene (HDPE) and recycled HDPE generated from packaging recycling. It was found that AGL made of recycled material had more environmental advantage shown by a reduction in the fossil fuel impact category. Rajendran, Scelsi, Hodzic, Soutis, & Al-Maadeed (2012) conducted LCA to compare the environmental performance of composite made of virgin plastics and recycled plastics. The result showed that the environmental benefit of recycled material could be different depending on the product application. For non-automotive applications, recycled material provided environmental benefits, whilst virgin material performed better in an automotive application. Gu, Guo, Zhang, Summers, & Hall (2017) applied LCA to evaluate the environmental performance of different mechanical recycling routes for different types of plastics. They applied various substitution factors depending on plastic type, ranging from 10% to 50%, resulting in considerable environmental benefit.

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

A more comprehensive approach was found in studies that combined environmental and economic aspects through LCA and LCC. Martinez-Sanchez, Kromann, & Astrup (2015) provided a comprehensive model in performing an environmental and economic assessment of solid waste management using different types of LCC, namely conventional LCC (CLCC), environmental LCC (ELCC) and social LCC (SLCC). Each type of LCC has a different relationship with an LCA, and it consists of different types of cost items such as budget costs, transfer costs and externality costs (Edwards et al., 2018). Presenting the basic principles of cost analysis in harmony with LCA and examples of applying different types of LCC in waste management systems can provide comprehensive results to improve decision-making (De Menna et al., 2018). Accorsi et al. (2014) combined LCA and CLCC to evaluate the environmental and economic performance of food packaging throughout its entire lifecycle. Their results indicated that the best environmental performance depended on the EoL management. Simões, Pinto, & Bernardo (2012) integrated LCA and SLCC to assess anti-glare lamella made of virgin and recycled HDPE. The environmental and economic consequences of anti-glare lamella production favoured the use of recycled material. Faraca, Martinez-Sanchez, & Astrup (2019) performed ELCC of hard plastic recycling in Denmark. They combined the environmental and economic aspects to assess three different recycling systems for hard plastic collected at the recycling centre without considering the collection stage. The study showed the importance of integrated environmental and financial assessment as a key to improve decision-making.

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

3. Materials and method

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

3.1. Study area

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

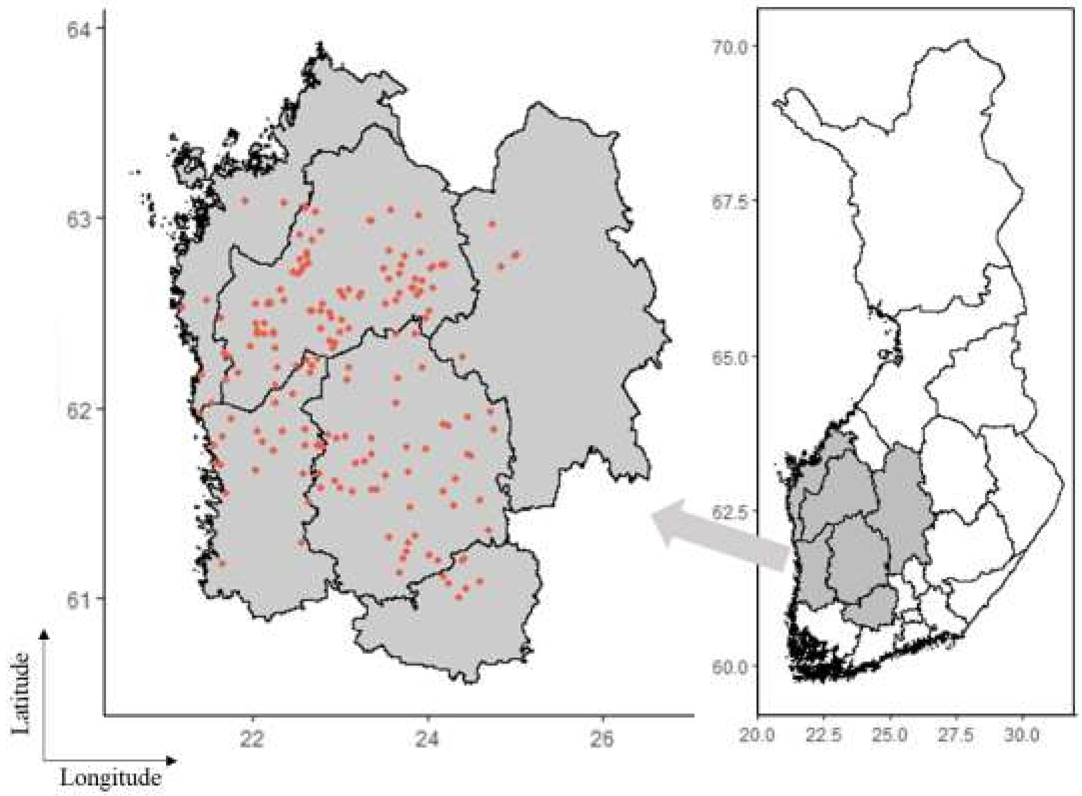

Fig. 1. Study area including the regions (grey color) and farm locations (coral dot).

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

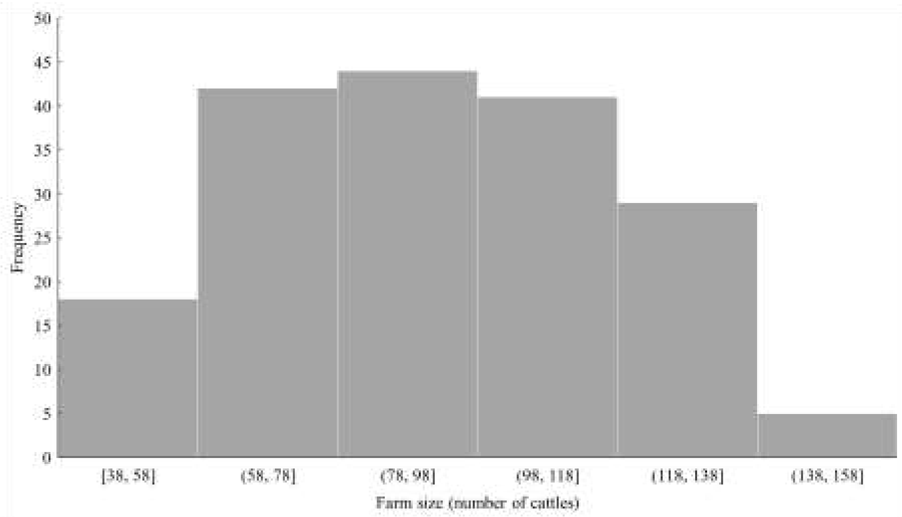

Fig. 2. Farm size distribution in study area.

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

3.2. Collection route

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

3.3. Goal and scope definition

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

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

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

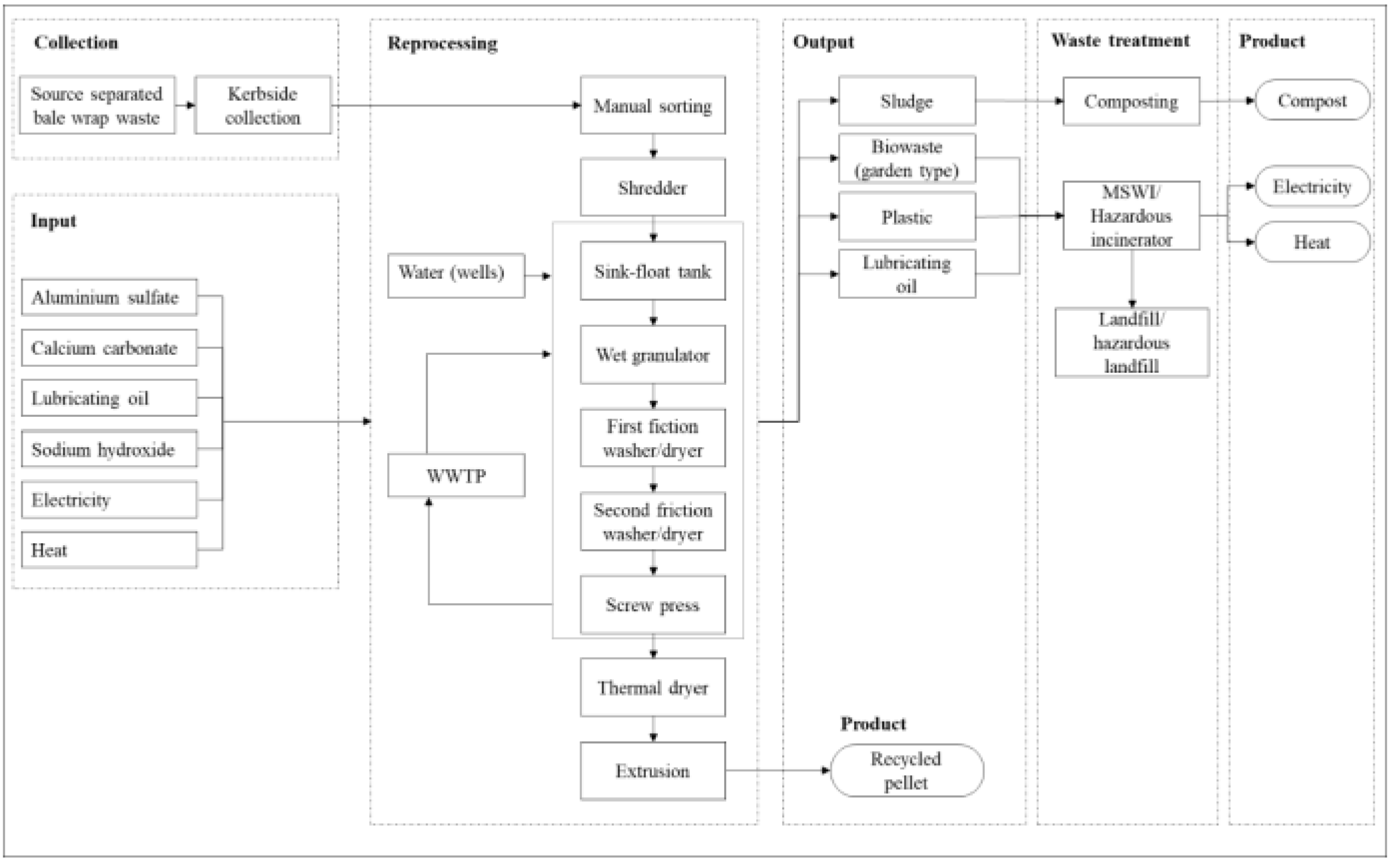

Fig. 3. System boundaries of bale wrap waste recycling.

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

3.4. Life cycle inventory

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

3.4.1. Operational data

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

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

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

Items	S1		S2		Note
	Value	Unit	Value	Unit	
• Electricity efficiency (incineration)	18	%	18	%	

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

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

3.4.2. Cost inventory

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

Budget costs were annualized to net present value based on plant capacity of 19000 ton/year (Ympäristö- ja terveyslautakunta, 2019) with 5% discount rate and 15 year discount period. They included capital cost, fixed operation and maintenance cost, variable cost related to operation and maintenance, and the sale of products generated from the treatment process (rPE, heat, electricity, compost). The equipment costs were normalized based on the usage rate (Martinez-Sanchez et al., 2015). The price of recycled LLDPE granulates ranged between 0.6-0.8 €/kg and was used as the basis to calculate the market substitution factor (Plasteurope, 2020a, 2020b). When the costs of equipment were known, but the capacity differed from the required one, an adjustment could be performed using equation (1) (Serna, 2018):

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

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

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

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

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

3.5. Assessment method

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

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

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

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

3.6. Sensitivity and scenario analysis

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

3.6.1. Sensitivity analysis

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

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

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

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

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

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

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

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

3.6.2. Scenario analysis

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

4. Results

4.1. Vehicle routing problem

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

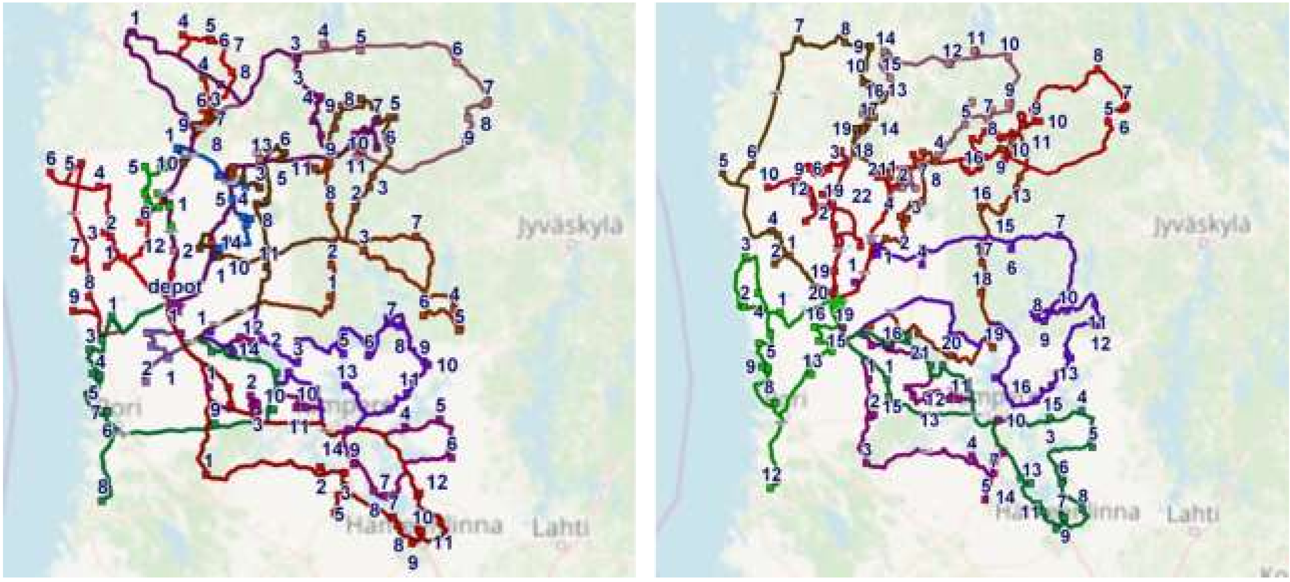

Fig. 4. Vehicle routing of bale wrap collection for S1 (left) and S2 (right).

4.2. Environmental assessment

Fig. 5 presents the environmental impacts of six impact categories based on the relative contribution of individual key processes, with the net result as the sum of various impacts and benefits. The impacts can be seen as benefits when represented in negative values. The environmental benefit from recycling was primarily acquired from avoided environmental impact due to the substitution of virgin plastic with recycled material. Moreover, the electricity and heat recovered from incineration as well as compost from composting the wastewater sludge also offered environmental benefits. On the other hand, impact on the environment occurred during collection, transportation, plastic reprocessing, incineration, and composting.

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

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

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

4.2.1. Global warming potential (GWP)

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

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

4.2.2. Fossil resource scarcity (FS)

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

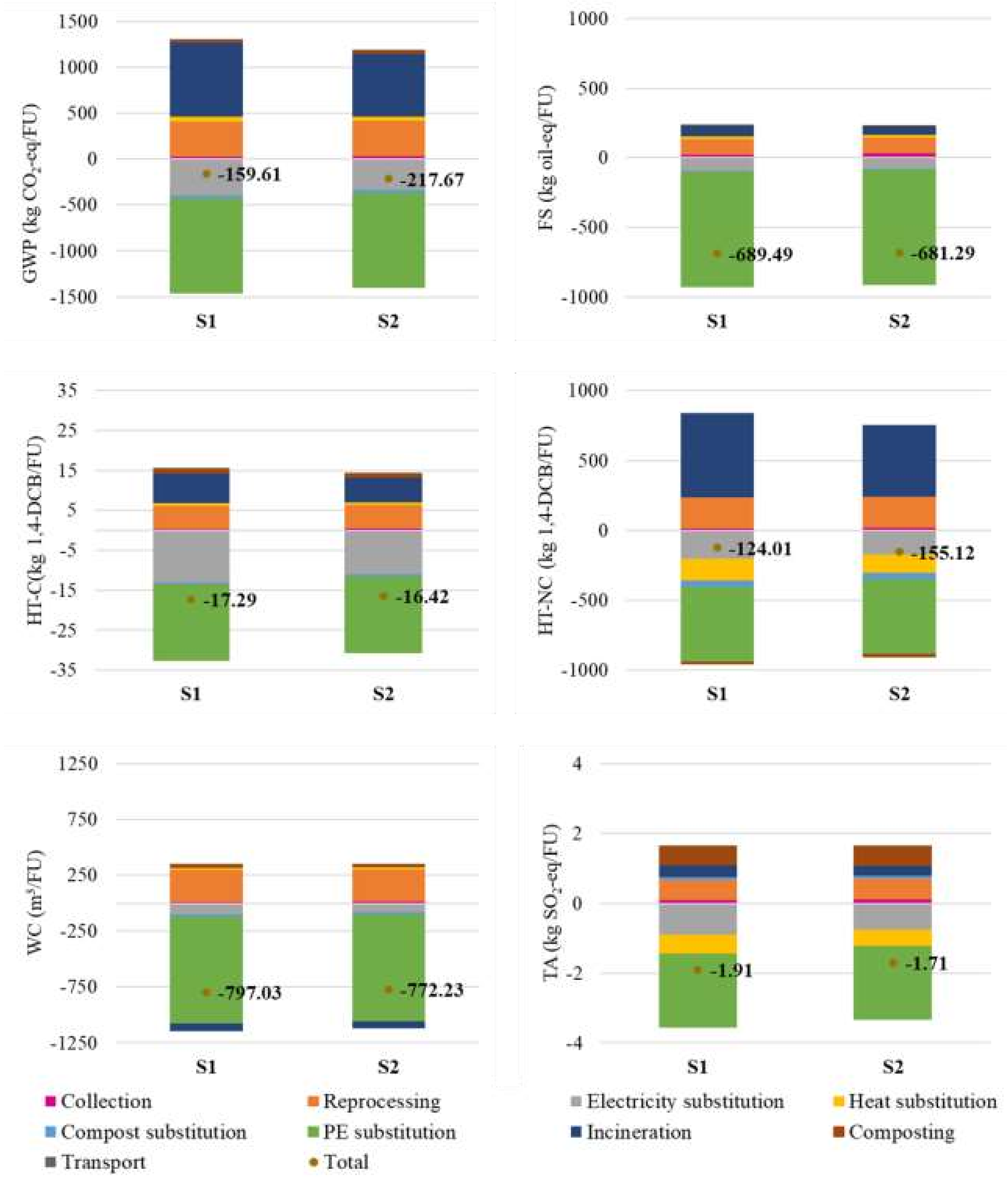

Fig. 5. Environmental impacts of bale wrap recycling based on its key-process for S1 and S2. GWP: global warming potential, FS: fossil resource scarcity, HT-C: human carcinogenic toxicity, HT-NC: human non-carcinogenic toxicity, TA: terrestrial acidification, WC: water consumption.

4.2.3. *Human carcinogenic toxicity (HT-C)*

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

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

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

4.2.5. *Water consumption (WC)*

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

4.2.6. Terrestrial acidification (TA)

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

4.3. Economic assessment

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

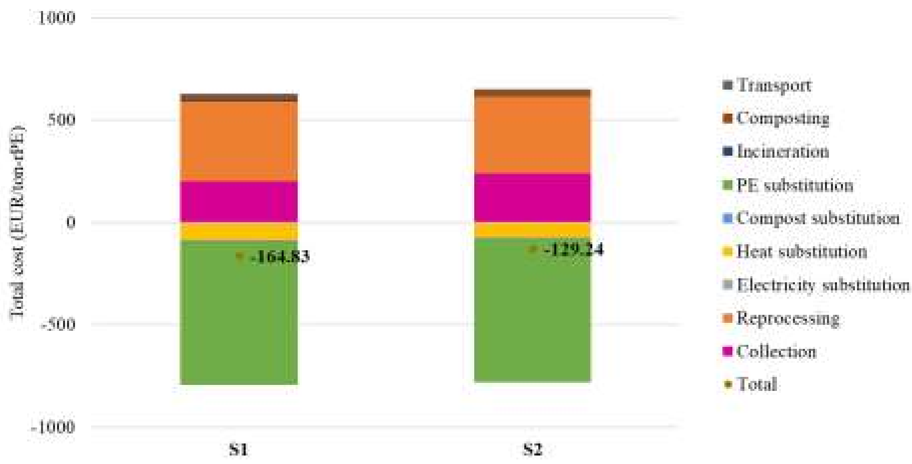

Fig. 6. Cost of bale wrap recycling based on its key processes for S1 and S2.

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

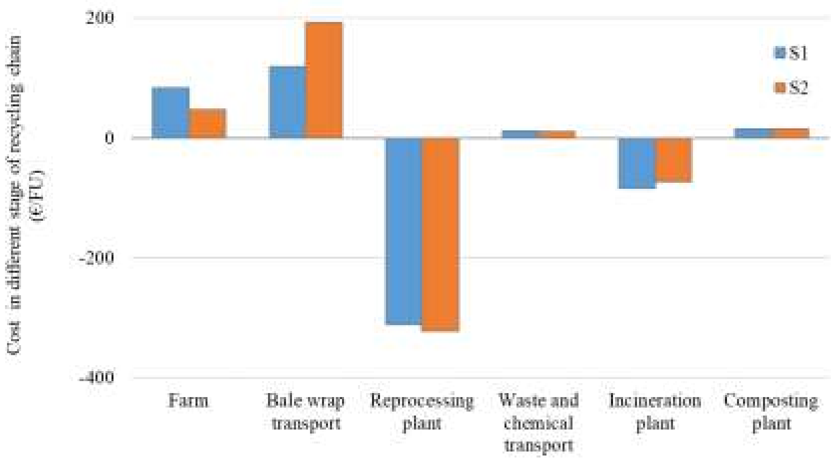

Fig. 7. Costs incurred in different stages of the recycling chain.

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

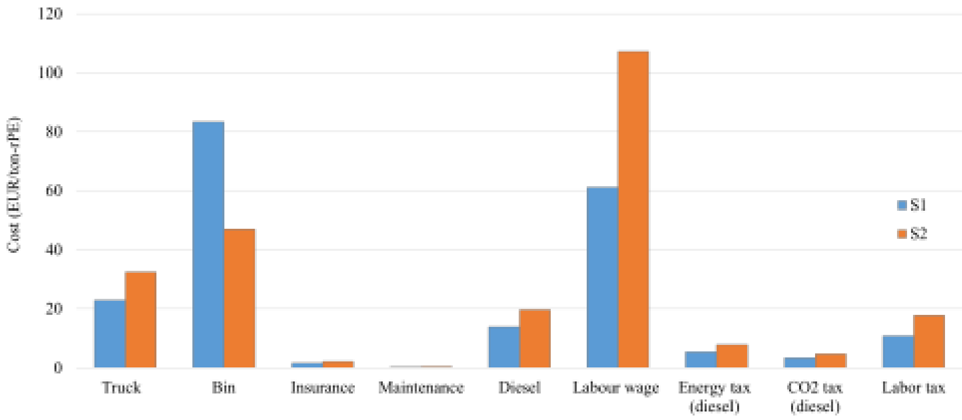

Fig. 8. The cost breakdown in the collection stage for S1 and S2

4.4. Sensitivity and scenario analysis

4.4.1. Sensitivity analysis

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

Fig. **9** shows the SR results for GWP and economic assessment in both scenarios. All parameters for environmental assessment were presented, whilst the 10 most sensitive parameters in the economic assessment that were overlapping in both scenarios were shown (see supplementary material in Table 4 and Fig. 1 for SR in all impact categories).

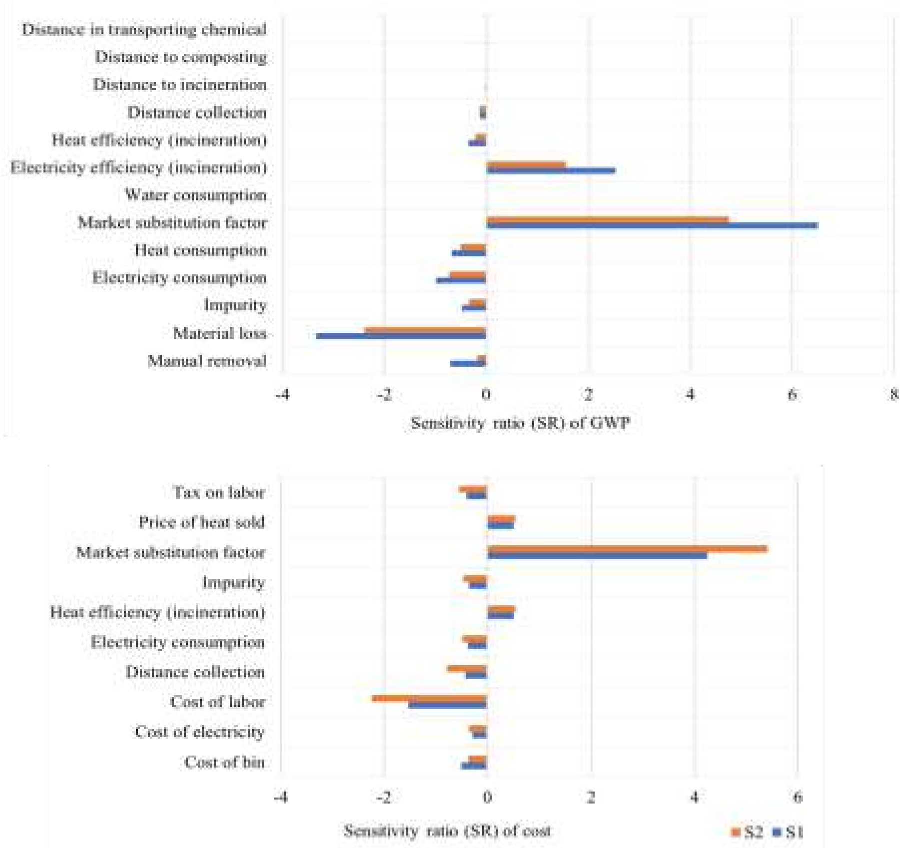

Fig. 9. Sensitivity ratio (SR) of global warming potential (GWP) and costs for S1 and S2.

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

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

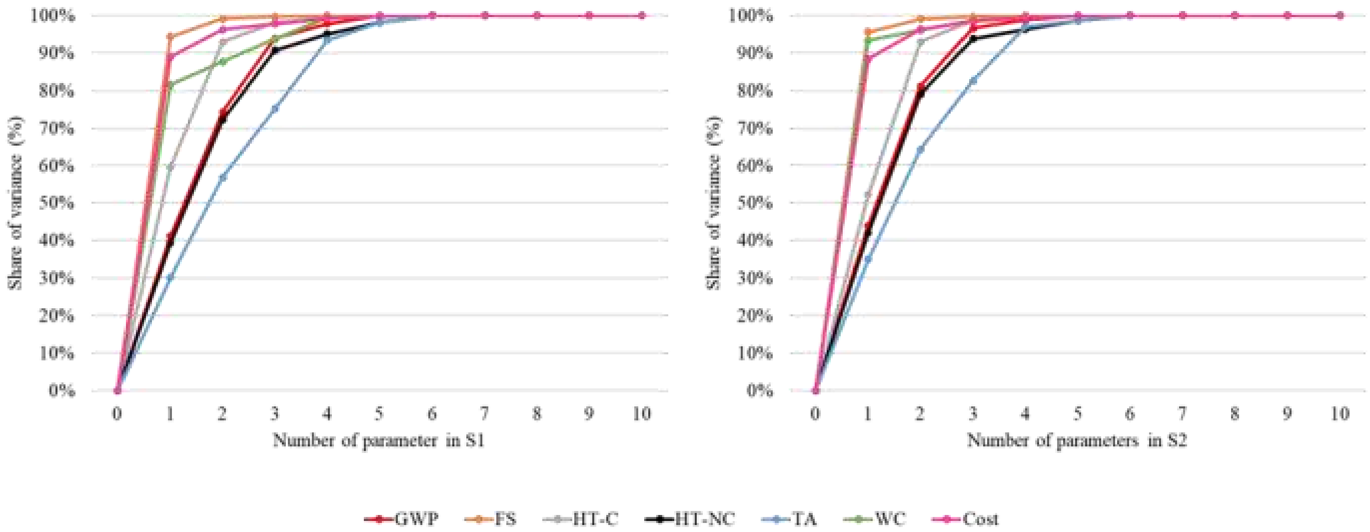

Fig. 10. Share of uncertainty contribution analysis for S1 and S2.

Fig. 11 summarizes the three most crucial parameters and their associated contribution to the overall variance. The value indicates the share of variance covered by the related parameter. A similar pattern was found in S1 and S2, except for TA. Electricity efficiency in incineration, which was one of the highest contributors to the total variance of TA in S1, was not found in S2. Meanwhile, material loss was found only in S2.

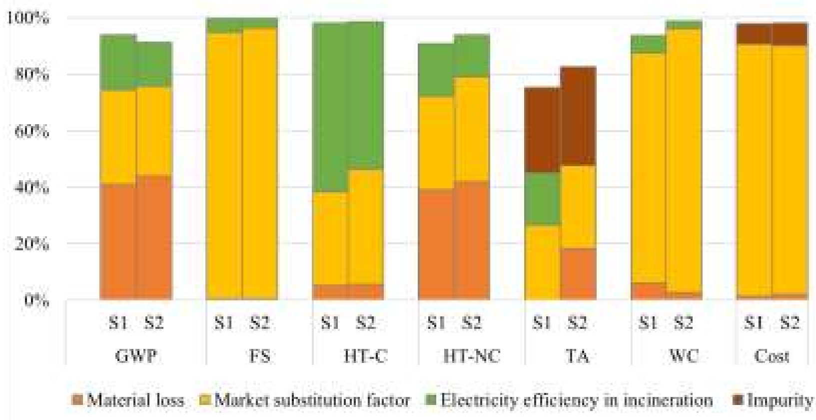

Fig. 11. Ranking of the three most important parameters associated with their percentages for S1 and S2.

4.4.2. Scenario analysis

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

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

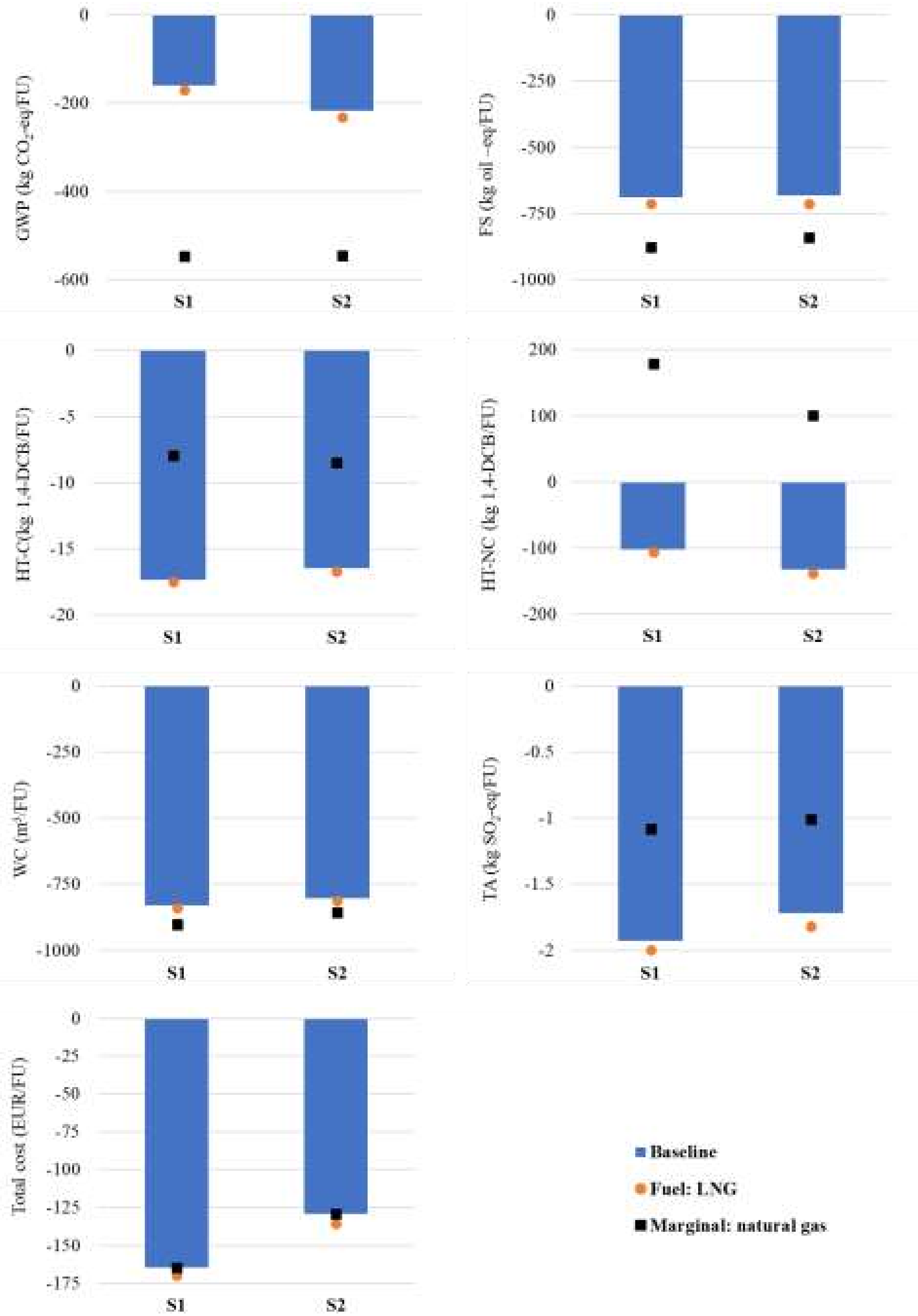

Fig. 12. Results of scenario analysis by changing fuel type and marginal energy for S1 and S2.

5. Discussion

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

5.1. Overall result

Total environmental savings for GWP in both scenarios ranged from around -160 kg CO_{2}-eq to -217 kg CO_{2}-eq per ton rPE, indicating much less savings compared with studies on plastic recycling as performed by Faraca et al. (2019), Rigamonti et al. (2014) and Shonfield (2008). Even when the FU in this research was adjusted from ton rPE to ton plastic waste to match these studies, the environmental saving was still 50% lower. Previous studies showed that recycling PP, PE, PET, PS, and PVC would avoid GWP to around 500-700 kg CO_{2}-eq per ton of plastic waste. The difference was heavily affected by operational data. Faraca et al. (2019) and Rigamonti et al. (2014) used a higher market substitution value, resulting in higher avoided virgin material production. Specifically for PE in an advanced mechanical recycling scenario, Faraca et al. (2019) applied a value of 91% for market substitution, which contributed to higher environmental saving compared with the value of 54.5% in this study. The difference of operational data across various studies was caused not only by the types of plastic but also by the source of plastic waste and its impurity. Recycled material derived from a single polymer with little organic impurity can replace virgin material with almost a 1:1 ratio (Lazarevic et al., 2010). Even though agricultural plastic waste is a good input for recycling due to its limited resin type, it is commonly impure due to organic contaminants (Briassoulis et al., 2012). This study applied a 54.5% market substitution factor based on the selling price of recycled LLDPE in the market. LLDPE is the material used in agricultural applications, and the value chosen in this study showed similarity with Gu et al. (2017), who used 50% as the substitution factor for recycled material derived from agricultural plastic.

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

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

5.2. Influence of process parameter and assessment methods

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

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

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

5.3. Shortcomings

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

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

5.4. Managerial implications and policy recommendations

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

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

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

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

6. Conclusions

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

- The quantification of environmental and economic performance in S1 and S2 show a trade-off between GWP and cost. The trade-off indicates that it is not possible to maximize both environmental savings and economic benefits. The scenario that offers more economic benefits will provide fewer environmental benefits when compared with the other scenario. In this case, S1 provides 27% more economic savings with 36% less GWP savings compared with S2. Hence, decision-makers must prioritize using weighting criteria to achieve the balance between economic and environmental goals.
- The collection contributes little to the environmental impact; however, it is one of the key processes for economic performance. It covers around 32-36% of the total cost for both scenarios, with S2 incurring 18% higher cost than S1. This cost is borne by the farmers, whose willingness to participate will determine the success of bale wrap recycling.
- Material substitution is the primary key process for economic and environmental saving by avoiding virgin material production, whilst the incineration of waste generated in reprocessing

causes the highest impact on GWP. It implies the importance of efficient reprocessing, where material loss should be minimized.

- The market substitution factor is the most sensitive parameter for both GWP and financial assessment. It results from the price of recycled material, which is affected by its supply and demand, quality, and acceptability. The highest uncertainty in GWP is generated from material loss, and in financial assessment it is derived from the market substitution factor. The results of sensitivity analysis are particularly important for the actors involved in CLSC and for decision-makers. When actors or decision-makers decide to adjust the recycling process or impose a certain policy, even a small change can significantly affect the output if the action affects the sensitivity parameter.

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

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