Sustainable Production and Consumption 25 (2021) 77–90

Contents lists available at ScienceDirect

Sustainable Production and Consumption

journal homepage: www.elsevier.com/locate/spc

Research article

Paper vs leaf: Carbon footprint of single-use plates made from renewable materials

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ARTICLE INFO

Article history:
Received 16 June 2020
Revised 13 August 2020
Accepted 18 August 2020
Available online 20 August 2020

Editor: Prof. Adisa Azapagic

Keywords:
Sustainable Development
Life cycle assessment
Disposable plates

ABSTRACT

Plastic pollution of the natural environment world-wide is ubiquitous. More than 80% of marine litter is made of plastics, 70% of which originates from disposable items, so plastic disposables need to be replaced with disposables made from renewable materials. However, it is important to investigate the environmental impact of renewable alternatives through their life cycle, in order to support sustainable consumption and production. In this study, the carbon footprint of disposable plates made from two different renewable materials (paper, tree leaves) were analysed using life cycle assessment. The leaf plate was produced in India and the paper plate in Finland, but both were used and disposed of in Sweden. The results showed that the leaf plate had higher carbon footprint, due to long-distance transport and use of fossil fuel-based electricity for production. Scenario analysis indicated that the emissions associated with the leaf plate were lower when replacing air freight with sea transport and with economies of scale in expanded production. For the paper plate, the processing stage was shown to contribute most life cycle emissions. These could be lowered by applying a biodegradable coating. In comparison the leaf plate had the benefit of being biodegradable, but this benefit was not enough to compete with the paper plate which was consider the less environmentally damaging alternative. However, in order to increase sustainability in the food supply chain, it will not be enough to just improve the material use for single use plates, especially since the idea of single use materials could be seen as less sustainable, but improved materials have the potential to offset the anticipated growth of the food service sector where single use items are widely used.

1. Introduction

Global plastic production reached 322 million tonnes in 2015 and is expected to double over the next 20 years (European Commission, 2018). However, our love for plastic has not always been as strong, e.g. in 1950 global production of plastic was only around 2 million tonnes (Geyer et al., 2017). Around that time, plastic started to be mass-produced and its main market changed from military to everyday products such as food packaging, cosmetics packaging, textiles and similar (Andrady and Neal, 2009; Parker, 2018). The world realised the benefits this versatile material could bring, e.g. from health and safety in the food industry to energy savings in transport (Andrady and Neal, 2009). Nowadays, around 40% of all plastic produced world-wide is used in single-use packaging (Geyer et al., 2017).

Rapidly increasing plastic production and use have given rise to rapidly increasing plastic waste generation. Globally, a mere 9% of all plastic waste generated until 2015 was recycled, 12% was incinerated and 79% was disposed of in landfill or the natural environment[1] (Geyer et al., 2017). The highest recycling rates are in Europe, where around 30% of plastic is recycled (European Commission, 2018), followed by China, where 25% of all plastic produced is recycled. These rates are still very low and need to be improved, since most plastic waste is left in the environment (as litter), landfilled or incinerated. This gives rise to greenhouse gas (GHG) emissions, thus contributing to anthropogenic cli-

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[1] These values exclude bio-based plastic produced from renewable biomass sources.

https://doi.org/10.1016/j.spc.2020.08.004

mate change (Eriksson and Finnveden, 2009; European Commission, 2018; IPCC, 2014). Most plastic today is fossil-based, with bio-based plastic, derived from e.g. maize, potato, sugar cane or sugar beet (IfBB, 2017), currently making up only 1% of global annual plastic production (European Bioplastics, 2019).

In addition to GHG emissions generated from production and waste management of plastic, plastic rubbish entering oceans causes damage to the global environment. It is estimated that more than 80% of marine litter is made up of plastics (European Commission, 2018). This plastic degrades to small microplastics <5 mm in size, which are eaten by marine species and can enter the human food chain (European Commission, 2018). Apart from polluting the marine environment, plastic is causing the death of marine species (*BBC*, 2018; Borunda, 2019; Parker, 2018). Therefore, it has been decided that the top 10 single-use plastic items making up over 70% of marine litter (including disposable plates and cutlery, drinking straws and cotton buds) are to be banned from the EU market from 2021 (*European Parliament News*, 2018).

If the current “to-go” culture based on fast/ready-made food and beverages packaged in single-use materials is to continue once plastics are banned, alternative materials for making single-use items must be developed. The industry has recognised the necessity for change and has begun to introduce disposables made from renewable materials (e.g. maize, sugar cane, wood, grass, leaves) onto the market (Duni, 2019(a); Leafymade, 2000; Vegware, 2019). The material used to make disposable tableware is crucial, since this has substantial impacts on the end-of-life choices of the product, as demonstrated by Fieschi and Pretato (2018). Compostable tableware can reduce the overall environmental impact of food waste management, as it can be composted in a thermophilic environment together with food waste. Fossil-based, non-compostable disposables need to be incinerated or landfilled, which has a higher carbon, water and resource footprint (Fieschi and Pretato, 2018). However, if bio-based and compostable disposables are to replace fossil-based plastic disposables, they will need to be produced on a much larger scale than at present. Therefore, it is crucial to study the environmental impact of these alternatives throughout their life cycle, in order to support sustainable consumption and production.

Previous research in this field has compared the environmental impacts of disposable cups made from different materials, in order to identify the most environmentally friendly type (Garrido and Alvarez del Castillo, 2007; Häkkinen and Vares, 2010; Van der Harst et al., 2014; Van der Harst and Potting, 2013; Woods and Bakshi, 2014; Gautam et al., 2020). Most of those studies compared bio-based materials with fossil-based materials, and some found that bio-based materials scored higher than fossil-based in terms of global warming potential (Häkkinen and Vares, 2010; Gautam et al., 2020). However, multiple-dataset comparisons based on life cycle assessments of disposable cups have shown that no one type of cup material is consistently better than others (Van der Harst and Potting, 2013). Three processes increasing the environmental impact have been identified: production of the basic material for the cup, cup manufacturing and waste processing (Van der Harst et al., 2014). Garrido and Alvarez del Castillo (2007) concluded that in order for a reusable cup made of polypropylene to have a smaller environmental impact than a single-use cup, it needs to be used at least 10 times, indicating that multiple use of items is key to reducing their environmental impact.

The main aim of this study was to assess the global warming potential of two biobased alternatives to fossil-based disposable plates and identify the processes with potential for improvement. Additional aims were to identify processes that contribute a large share of the impacts and to assess possible improvements to both products that could lower the total impacts during their life cycle.

2. Material and methods

Life cycle assessment (LCA), following ISO standards 14040:2006 and 14044:2006 as described in Matthews et al. (2014), was used for the analysis. The functional unit was defined as one flat disposable plate 20 ± 2 cm in diameter, which seems to be a common size of disposable plates on the market (Duni, 2019; Leafymade, 2000; Little Cherry, 2019). The impact assessment category used was global warming potential (GWP_{100}), also known as carbon footprint. Although assessment of a single impact category gives a limited perspective on the environmental performance, GWP_{100} was selected here due to the ongoing climate debate and the fact that renewable materials are often marketed as climate friendly. Indicators for this category are the main greenhouse gases (GHG) emitted from processes, defined by the Kyoto Protocol as carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), sulphur hexafluoride (SF_6) and nitrogen trifluoride (NF_3) (UNFCCC, 2012). However, in this study only CO_2, CH_4 and N_2O were considered. The characterisation model adopted was the IPCC model, as it is the most up-to-date and scientifically robust model available (European Commission, 2011) GWP_{100} was based on the IPCC's Fifth Assessment Report (2014), which sets the respective GWP_{100} of CH_4 and N_2O at 28-fold and 265-fold that of the corresponding mass of CO_2.

Two different product systems were selected for comparison in this study, a leaf plate and a coated paper plate (Fig. 1). Both plates have similar functions, and similar size and weight (9.3 g for the leaf plate and 9 g for the coated paper plate). The leaf plate was a disposable plate made of leaves from the sal tree (*Shorea robusta*), grown in India. This plate was selected because it is a new product on the Swedish market and often referred to as a sustainable alternative (even though it has a long transport distance), and because interest in this type of plate is on the rise in Sweden and the rest of Europe. The coated paper plate was a disposable plate produced in Finland. This product was selected due to its shorter supply chain than the leaf plate and its long history and popularity (market leader) in Sweden, so its production is relatively advanced and well-studied in comparison with the leaf plate. The mass-produced nature of the paper plate is also reflected by the price, as it costs roughly 0.5-1.0 SEK/plate, while the small-scale produced leaf plate costs roughly 2.5 SEK/plate.

It was assumed that both types of disposable plate were used and disposed of in Uppsala, Sweden, and all transport therefore included the distance to this city. The product systems of the plates were largely similar (Fig. 1). The life cycle stages studied included the processes directly involved in producing the plates and raw materials, but also transportation, packaging and waste management of the used plates. The use phase was not excluded from the assessment, but was assumed to have no impact, since transportation was assumed to be conducted by bicycle or on foot, and therefore negligible. Furthermore, single-use plates do not require any input before use, since the food served was not included in the study. Food leftovers on the plates were considered outside the scope of this study, even though they can have a significant climate impact (Malefors et al., 2019; Matzembacher et al., 2020). Only direct inputs were considered, such as use of vehicles or machines, while production of vehicles or machines was not included.

At the end of life, the paper plate would be incinerated as part of the municipal solid waste stream in Uppsala, while the leaf plate would be anaerobically digested as part of a biodegradable waste stream. System expansion was used to calculate the impact of energy recovery replacing wood chips and diesel. Material recycling was not included as an option, as single-use plates are heavily contaminated by food leftovers and it would be complicated to recycle

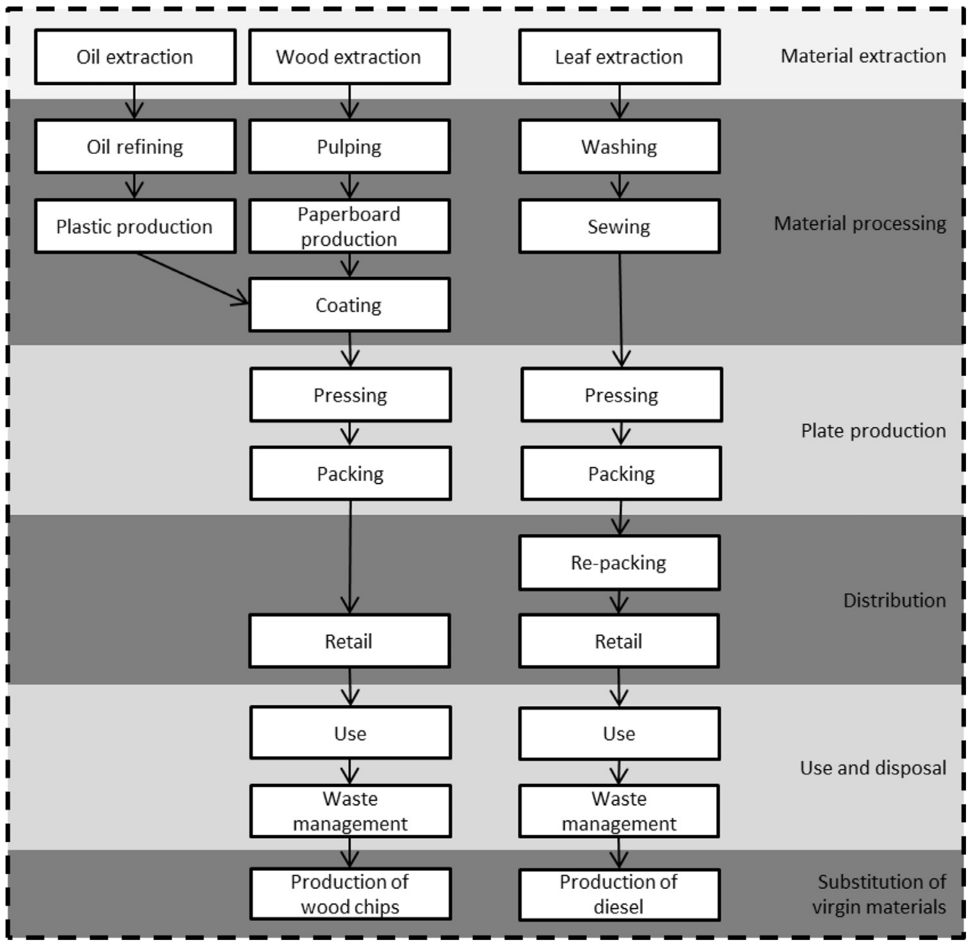

Fig. 1. Schematic overview of the product systems of the two disposable plate types assessed.

them. They are also not included in the producers' responsibility for collection and recycling of packaging material applied in Sweden. Landfilling of organic waste was also not included, since this is forbidden in Sweden (SFS 2001:512).

In order to assess the effect of possible changes in the life cycle of the two products, several improvement analyses were conducted. Since the company producing the leaf plate was a start-up business, in one improvement analysis it was assumed to be a well-established business in the future, with transport and production benefiting from larger volumes. Another improvement analysis considered the impact of the leaf plate when used in India, reducing the need for long-distance transportation but also foregoing the opportunity for efficient waste management. In the case of the paper plate, an improvement analysis considered replacement of the coating with a biodegradable layer so the plate could be disposed of by anaerobic digestion together with food waste, instead of being incinerated.

2.1. System description

2.1.1. Leaf plate

Several commercial companies produce leaf plates (Eco-gecko, 2019; Leafymade, 2000; Little Cherry, 2019). The leaf plate produced by Leafymade was selected as representative in the leaf plate scenario in this study. Data for this system were collected in close collaboration with Leafymade through regular visits to the company, emails and phone calls.

Leafymade is a young start-up located in Uppsala, Sweden. Production of the leaf plates takes place in the state of Odisha in eastern India. Tribal women collect leaves of sal trees in the local rainforest. The leaves have special properties such as water resistance, rigid structure and long-term colour retention. Picking leaves only requires energy from manpower and no other raw materials apart from leaves are needed in this stage of production.

When leaves are picked, they are washed first and then sewn together. Washing uses ~10 mL water per plate (six leaves) (Mehta, 2019). No other natural or artificial materials are used for this unit process and the only output is waterborne dust washed off the leaves. Sewing is done on foot-operated sewing machines, so no electricity is required. One plate is made of six stitched leaves and requires 40 cm cotton thread, of which 5 cm is wasted (Mehta, 2019). At the end of this stage, the sewn leaves are stacked, bundled and transported from Daringbadi to Bhubaneswar (246 km) in a light diesel truck.

In Bhubaneswar, the sewn leaves are pressed in an electrical heat-press. Depending on the quality of the leaves, a small amount of water (~2 mL/plate) may be sprinkled on the leaves in order to increase the elasticity before pressing (Mehta, 2019). Based on data on daily energy consumption and daily plate production at the facility (obtained from Leafymade), it was calculated that electricity use per plate was 9 Wh. After pressing one plate, the cut-off pieces of leaves generate approximately 390 cm^2 or 12 g of solid waste per plate.

After the production stage, plates are packed in corrugated cardboard boxes, with 1350 plates per box. One shipment consists of 19 boxes, comprising in total 260 kg gross weight and 237 kg net weight according to the shipment list used for calculations (Mehta, 2019). In each shipment, 25,485 plates are transported from India to Sweden. Here, the weight of an average plate was calculated as 9.3 g, as the exact mass can vary.

Once the plates are ready to be shipped, they are sent from Bhubaneswar to the port city of Kolkata (442 km) by light truck. In some cases when Leafymade urgently needs stock (8% of all shipments so far), plates are transported from Bhubaneswar to Stockholm, Sweden, by air (Mehta, 2019). Otherwise (92% of all shipments so far), the goods are transported by ship from Kolkata to Gothenburg, Sweden (15,663 km) (Mehta, 2019). Compared with shipping goods by air, the sea transport can take about a month longer (Searoutes, 2019). From Gothenburg, the goods are transported to Uppsala (453 km) by fixed-body truck. From Stockholm airport, the goods are transported to Uppsala (71 km) by rigid truck.

2.1.2. Paper plate

Since its invention, the disposable paper plate has been manufactured by many different companies worldwide, each with a specific manufacturing process. The paper plate produced by the Finnish company MiniMaid was selected as a case in the present study, due to the relatively close proximity of the production site to Uppsala, Sweden. Data for this plate were collected in collaboration with MiniMaid through communication by emails and phone calls. MiniMaid is a private label manufacturer, i.e. its customers choose the parameters of their desired plate, including thickness of paperboard or number of plates per pack. Therefore, the figures used here were for a hypothetical average paper plate produced in 2018 (Knutar, 2019).

As only the production of paper plates takes place at MiniMaid, all the data for upstream processes had to be acquired from one of MiniMaid's suppliers. For the purposes of this study, data from one significant supplier of paperboard, an integrated pulp and paperboard mill located in Husum, Sweden, were used (Knutar, 2019). Attempts to interview Husum mill were not successful, so information on wood supply had to be obtained from literature sources. Data for the first stage of the paper plate life cycle (extraction of wood) were derived from González-García et al. (2009), who investigated the whole process of extraction of wood from site preparation and logging to wood transport from forest landing to the pulp mill gate. Their data were considered representative, since the pulp mill they studied and the pulp and paperboard mill used in the paper plate scenario are located close to each other in northern Sweden (González-García et al., 2009; Grahn, 2019). The pulp mill studied by González-García et al. (2009) obtains 25% of its wood supply from Baltic countries, 30% from southern Sweden and 45% from central Sweden, transported by a combination of truck and ship. All the energy required in silviculture, logging and transport was considered as input to the first phase of the paper plate's life cycle. The total non-renewable energy consumed in these processes is 370 MJ/m^3 of wood (González-García et al., 2009). The outputs of these processes are the total GHG emissions associated with energy use, which amount to 36.1 kg CO_2e/m^3 wood (González-García et al., 2009).

On arrival at the pulp mill, wood logs need to be debarked and processed into wood chips, which in turn are turned into pulp by three main methods: mechanical, chemical or biopulping (Das and Houtman, 2004). Mechanical pulping involves applying mechanical forces to grind wood against a rotating stone (Das and Houtman, 2004). The wood chips can be pre-treated by steam (thermo-mechanical pulp) or a combination of steam and sodium sulphite (chemi-thermomechanical pulp) (CEPI, 2019; Das and Houtman, 2004). Mechanical pulping is energy-intensive and gives higher yields, but lower-strength fibres, than chemical pulping (Das and Houtman, 2004). Chemical pulping uses chemicals in a cooking process to remove lignin from the wood and separate it into cellulose fibres (CEPI, 2019). This gives lower yield, but fibres of higher strength than mechanical pulping (CEPI, 2019; Das and Houtman, 2004). In biopulping, lignin-degrading fungi are applied (Das and Houtman, 2004). The integrated pulp and board mill that produces the paperboard used for MiniMaid's paper plates employs both chemi-thermomechanical pulping and chemical pulping methods (Metsä Board Husum, 2019).

The pulp yield from wood is normally around 55% for chemical pulping (FEFCO and CEPI Containerboard, 2015), meaning that 1000 kg of wood yields 550 kg of pulp. To find more information about GHG emissions from the processes at the integrated mill in Husum, the mill's website was searched for environmental profiles of their products. Only two kinds of paperboard suitable for a paper plate application were found, one uncoated and one coated (Metsä Board, 2019). The uncoated paperboard was selected as representative, since MiniMaid has its paperboard coated in a separate factory (Grahn, 2019). Hence, the environmental profile of "MetsäBoard Natural FBB 175-325 g/m^2" was selected. Based on the environmental profile of this paperboard, the process of making pulp and subsequent paperboard emits 44 kg of fossil CO_2/tonne paperboard (Metsä Board, 2018). However, 82% of the total fuels used in all mills within the Metsä Board group are derived from biomass (wood, bark, black liquor[2]) (Metsä Board, 2018). According to the environmental profile of the paperboard investigated, it comprises 84% pulp, 8% moisture, 5% binders and 3% pigments and fillers (Metsä Board, 2018).

Each paperboard grade produced by machines is tailored to a specific standard, but the overall basic process of paper and board making is similar (Ottenio et al., 2004). After pulp has been obtained, it can be bleached depending on its final use (Australian Packaging Covenant, 2019; Iggesund, 2019; Ottenio et al., 2004). In this study, unbleached pulp was assumed to be used for the paper plate. Unbleached pulp is screened, cleaned and diluted in water (Iggesund, 2019; Ottenio et al., 2004). Chemicals are then added to the mixture of raw fibres and water, which is pumped to the headbox, a device controlling the flow of the mass (Ottenio et al., 2004). The headbox feeds the stock onto the wire section, a woven mesh conveyor belt (Ottenio et al., 2004). As the paper mass travels on the conveyor belt the water is drained away, leaving fibres on the mesh. By the time the mat of fibres arrives at the end of the wire section, it has become a sheet of paper (Ottenio et al., 2004). A paperboard machine has a number of formation devices in headboxes and wires, which manufacture multi-ply sheets that are combined later in the process (Ottenio et al., 2004). The moist sheets of paperboard move to the press section, where more water is squeezed out, which binds the fibres together (Ottenio et al., 2004). The sheets are then dried by steam. Halfway through the drying process, the paperboard can be coated with pigments and binding agents (Ottenio et al., 2004).

Based on customer specifications, paperboard can be coated with a number of soak-proof materials such as polyethylene (PE), water-based barrier (MiniMaid Ab, 2019) or a compostable PLA layer (Shah et al., 2008). MiniMaid provides its customers with the option of PE-based coating or water-based dispersion coating based on dispersion of solid material dissolved in water (MiniMaid Ab, 2019). The solid material used can be titanium dioxide (TiO_2) nanoparticles in combination with polyolefin copolymers (Mates et al., 2016) or fluoroacrylic copolymer in combination with hydrophilic bentonite nanoclay (Mates et al., 2014), in proportions in relation to water of 5% and 3%, respectively (Mates et al., 2014, 2016). One plate uses 0.39 g of water-based dispersion barrier or 0.55 g of PE coating (Knutar, 2019). As the exact compo-

[2] Black liquor is the waste product from the kraft process when digesting pulpwood into paper pulp, removing lignin, hemicelluloses and other extractives from the wood to free the cellulose fibres (Climate Technology Centre and Network, 2016).

Table 1
Specification of modes of transportation of the leaf plate.

Transport type	Route	Vehicle type	Distance (km)	Cargo load factor (%)	Cargo capacity (m^3 or t)	Shipment volume/ weight (m^3 or t)	Source
Sea route	Leg 1 (DAR-BHU)	Van	246	100	8 m^3	4 m^3	(NTMCalc 4.0, 2019; Mehta, 2019)
	Leg 2 (BHU-KOL)	Van	442	50	2.25 t	0.260 t	(NTMCalc 4.0, 2019; Mehta, 2019)
	Leg 3 (KOL-GOT)	Bulk carrier	15 663	55	15,000 t	0.260 t	(NTMCalc 4.0, 2019; Mehta, 2019; Searoutes, 2019)
	Leg 4 (GOT-UPP)	Rigid truck (7.5-12 t)	453	40	6 t	0.260 t	(NTMCalc 4.0, 2019; Mehta, 2019)
Flight route	Leg 1 (DAR-BHU)	Van	246	100	8 m^3	4 m^3	(NTMCalc 4.0, 2019; Mehta, 2019)
	Leg 2 (BHU-STO)	Belly freighter - cargo	6,996	65	14 t	0.014 t	(NTMCalc 4.0, 2019; Mehta, 2019)
	Leg 3 (STO-UPP)	Van	71	20	1.5 t	0.014 t	(NTMCalc 4.0, 2019; Mehta, 2019)

sition of bio-coating applied on MiniMaid's paper plates was unknown and considering that some bio-coating is still based on elements found in plastic, the end of life of the paper plate coated with such material is ambivalent. Therefore, a PE coating was assumed in the paper plate scenario, since the process of PE production is relatively well-known and the end of life of a PE-coated paper plate is unambiguously incineration with energy recovery.

The process of coating can be performed at the paperboard mill or at a separate factory (Knutar, 2019). At MiniMaid, most of the coating is done at separate factories (Grahn, 2019). The distance between the paperboard mill and coating factories is 10 to 50 km (Knutar, 2019). Coated paperboard is later transported to MiniMaid by a 20-tonne truck (Grahn, 2019) over a distance of 330 km. The entire truckload of 40 tonnes of paperboard is shipped to MiniMaid at once (Knutar, 2019) and the sheets of paperboard are pressed into plates. The electricity required for pressing one plate is 2.8 Wh (Knutar, 2019). The weight of one plate is 8.4 g, while with a layer of PE coating it is 9 g (Knutar, 2019). A pressed plate is packed using 0.05 g polyolefin shrink-film per plate (Knutar, 2019). Stacks of plates are placed in corrugated cardboard boxes, each weighing 227 g, with an average number of 660 plates per box (Knutar, 2019). The packed paper plates are ready to be shipped to customers. A standard delivery comprises 2 million plates in a fully loaded truck (Knutar, 2019).

2.2. Inventory data

This section explains the software and methods used to process the collected data above, in order to calculate GHG emissions of the two plates.

2.2.1. Leaf plate

In order to calculate GHG emissions from transportation of leaf plates throughout their life cycle, the Network for Transport Measures (NTM) Calculator Advanced 4.0 was used ("NTMCalc 4.0", 2019). Table 1 presents a summary of the variables influencing the carbon footprint arising from transportation of the leaf plate. The leaf plate can be transported by sea or air. The sea route comprises four different legs. For the first and second leg, from Daringbadi to Kolkata (688 km), the vehicle type "van" was used, as it corresponds best to the light truck used in India. It was assumed that the van runs on fuel corresponding to Diesel B5-EU and that its fuel consumption is 8.5 L per 100 km.

In order to estimate the route of the ship from Kolkata to Gothenburg (15,663 km), the sea routes calculator was used (Searoutes, 2019). This route was then entered into the NTM Calculator. The vehicle type used for this calculation was "bulk carrier", as its weight corresponds best to the actual vessel weight used for transportation of Leafymade's shipments. On arrival in Gothenburg, the plates were assumed to be further transported to Uppsala (453 km) by "rigid truck 7.5-12 t" with fuel consumption of 17.8 L per 100 km.

The flight route comprised three legs. The first leg was identical to the sea route. For the second leg, from Bhubaneswar airport to Stockholm airport (6,996 km), "belly freighter – cargo" was chosen in the NTM Calculator. Cargo carrier capacity was 14,000 kg and the default cargo load factor was 65%. The weight of shipment was 14 kg. For the last leg of the journey from Stockholm to Uppsala (71 km), the vehicle type "van" with the same specifications as described above was chosen in the NTM Calculator.

2.2.1.1. Processing. As processing of leaves is done on foot-operated sewing machines, no electrical power is required. The only process in the production phase of the leaf plate that utilises electric power is heat pressing. A small proportion of electricity produced in India is made from renewable energy sources, but most still comes from coal (Central Electricity Authority, 2019). Therefore, in order to quantify GHG emissions of the electricity used in pressing the plates, data for GHG emissions from the Indian electricity mix had to be acquired. For this calculation, the "Standard values for emission factors v.1.0." dataset compiled by the European Commission (2014) was used. This dataset states that, per 1 MJ of electricity produced in India, 292 g CO_2e are emitted to the atmosphere. The electricity used per plate was 31 kJ.

2.2.1.2. Packaging. Packaging can be divided into two subgroups, packaging used for transport from business to business and packaging of plates into retail packs designed for the end-customer.

In business-to-business packaging, the leaf plates are packed in corrugated cardboard boxes with dimensions (LxWxH) 0.63 m x 0.42 m x 0.42 m (Mehta, 2019). Each box weighs 1 kg (Mehta, 2019). In calculating GHG emissions from packaging, figures from a Finnish comparative study were used (Koskela et al., 2014). That study compared the environmental impacts of reusable plastic crates with those of corrugated cardboard boxes, using LCA, and took into account GHG emissions from manufacturing of the boxes, their use, the delivery routes to retailers and waste management/recycling of the boxes. A corrugated cardboard box with dimensions (LxWxH) 0.54 m x 0.33 m x 0.11 m and 0.2 kg weight was considered, for which GHG emissions were 0.9 kg (Koskela et al., 2014). These figures were scaled up here to the weight of the corrugated cardboard box used by Leafymade (1 kg). The final GHG emissions from Leafymade's box were in agreement with values in another study examining the environmental impact of corrugated cardboard boxes (Yi et al., 2017). When the figures from that study were scaled up to the weight of Leafymade's box, the value for GHG emissions was almost identical to that based on the Finnish study by Koskela et al. (2014).

The retail packaging for the leaf plates is made of bio-based polyethylene (Bio-PE), which is 100% recyclable, non-biodegradable and made from sugar cane (Braskem, 2019). The length of the pack (31 cm) was calculated based on the overall length of the bio-PE role supplied by the supplier and the number of packs made from it. The width of the pack (25 cm) was obtained from the

Table 2
Specification of modes of transportation of the paper plate.

Route type	Route	Vehicle type	Distance (km)	Load factor (%)	Cargo capacity (m^3/t)	Shipment volume/ weight (m^3/t)	Source
Production	Leg 1 (HUS-HOL)	Truck with trailer 50-60 t	93	100	40 t	40 t	(NTMCalc 4.0, 2019; Knutar, 2019)
	Leg 2 (HOL-VAS)	Ro-Ro ship[a]	103	70	10,000 t	60 t	(NTMCalc 4.0, 2019)
	Leg 3 (VAS-COA[b])	Truck with trailer 50-60 t	30	100	40 t	40 t	(NTMCalc 4.0, 2019; Knutar, 2019;)
	Leg 4 (COA-TER[c])	Truck with trailer 50-60 t	135	100	40 t	40 t	(NTMCalc 4.0, 2019; Knutar, 2019; Grahn 2019)
Sales	Leg 1 (TER-VAS)	Rigid truck 20-26 t	135	100	19 t	19 t	(NTMCalc 4.0, 2019; Knutar, 2019; Grahn 2019)
	Leg 2 (VAS-HOL)	Ro-Ro ship	103	70	10,000 t	26 t	(NTMCalc 4.0, 2019)
	Leg 3 (HOL-UPP)	Rigid truck 20-26 t	579	100	19 t	19 t	(NTMCalc 4.0, 2019; Knutar, 2019; Grahn 2019)

[a] Roll-on/roll-off ship designed to carry wheeled cargo that is driven on and off the ship on its own wheels ("NTMCalc 4.0", 2019).
[b] Coating factory where plates are coated.
[c] Terjärv, the town in Finland where MiniMaid is located (MiniMaid, 2019).

supplier (Högström, 2019). The supplier also reported that in order to manufacture the bio-HDPE used for packaging, 23-27g/m^2 Bio-PE are needed (Högström, 2019). As the area of one pack is 0.16 m^2 and 25 g of Bio-PE are needed for 1 m^2, one pack uses 4 g. There are 12 plates in one pack, and therefore the amount needed per functional unit is 0.3 g. According to a cradle-to-gate LCA study of bio-PE production, which included ethanol production, bio-ethylene production, polymerisation to bio-HDPE and final transport of the polymers from Brazil to Europe, production of bio-HDPE emits 2.45 kg CO_2e/kg bio-HDPE (Tsiropoulos et al., 2015).

The processing of bio-HDPE resin to the bio-HDPE film used for packaging requires 0.5 kWh/kg bio-HDPE film produced (Högström, 2019). As one plate needs 0.3 g bio-HDPE, the electricity required for the production of film per plate is 0.15 Wh. In order to calculate GHG emissions from this process, data on GHG emissions from the electricity mix for Sweden had to be acquired, since the production of bio-HDPE film takes place in Sweden. This value was found in Moro and Lonza (2018), who accounted for upstream production and import and export of electricity for each member state of the EU. For Sweden, they found the carbon intensity of electricity to be 47 g CO_2e/kWh when taking into consideration upstream electricity production and import and export of electricity (Moro and Lonza, 2018).

2.2.1.3. Disposal. In order to calculate GHG emissions from the specific waste management methods available for the leaf plate, the methodology from Eriksson et al. (2015) was applied. As their study area was Uppsala municipality, the waste management facilities they investigated were the same as assumed for the leaf plate.

There were three different options for disposal of the leaf plate: composting, incineration with energy recovery and anaerobic digestion. It has previously been calculated that composting in Uppsala emits 0.043 kg CO_2e/kg composted waste (Eriksson et al., 2015). This includes production of windrows, the composting process, production of soil amendment, machinery use and transport to the composting facility. The compost produced in the system is currently used for covering landfill, and thus does not replace any other product or service (Eriksson et al., 2015). Incineration with energy recovery is another option. Based on the heat content of sal tree leaves (242.8 J/g) identified in Singh et al. (2016), the GHG emissions and the amount of substituted wood chips were calculated. Finally, the GHG emissions from anaerobic digestion of a leaf plate were calculated using values for water content (0.45%) and heat content of the leaf (242.8 J/g) obtained from Singh et al. (2016). As anaerobic digestion produces biogas, in this calculation substituted diesel used by city buses in Uppsala was considered. Since biogas production requires electricity use and the biomass needs to be transported to the biogas plant, emissions from these processes were also factored in. Anaerobic digestion was selected as the default option for the leaf plate, as this is where organic waste normally ends up in Uppsala.

2.2.2. Paper plate

For the first stage of the paper plate life cycle, the calculations of GHG emissions from transportation were based on González-García et al. (2009). As mentioned above, the pulp mill investigated in González-García et al. (2009) was assumed have the same criteria as the integrated pulp and paperboard mill used in the paper plate scenario. In that study, the total GWP of silviculture, logging and transport of wood was 36.1 kg CO_2e/m^3 wood (González-García et al., 2009). Around 58% of the GHG emissions originated from transport of wood and the remaining 42% from logging and silviculture (González-García et al., 2009). In order to calculate the GHG emissions released from transport of wood per plate, the amount of wood needed for production of one plate had to be calculated as follows: the weight of one uncoated plate was 8.4 g, of which 84% consisted of pulp. Therefore, the weight of pulp per plate was 7.1 g. Assuming a pulp yield from wood of 55% (FEFCO and CEPI Containerboard, 2015), the amount of wood needed for the production of 7.1 g pulp was 12.9 g of dry wood. However, González-García et al. (2009) based their calculations on wood with moisture content 40% and density 399 kg/m^3, so the moisture content had to be accounted for on a per-plate basis. Thus, the total weight of wood required per plate was 21.5 g, with 40% made up of moisture and 60% of dry wood. Therefore, if processing and transporting 665 kg of solid wood under bark (40% moisture content) emitted 36.1 kg CO_2e, then the GHG emissions from these processes per plate would be 1.2 g. Since 58% was associated with transport of the pulpwood to the pulp mill gate, the GWP of wood transport required for one paper plate was 0.7 g.

The following calculations of GHG emissions from transportation in the subsequent stages of the paper plate life cycle were made using the NTM Calculator Advanced 4.0 ("NTMCalc 4.0", 2019). After the pulp and paperboard were produced, sheets of paperboard were assumed to be shipped from Husum to MiniMaid. This route was called "production route" and was divided into four legs. Specifications for each leg are shown in Table 2.

The first leg of the route was from Husum, Sweden, to Holmsund, Sweden (93 km). The vehicle type "truck with trailer 50-60 t" was selected, as its typical cargo capacity (40 tonnes) was the same as the cargo capacity of the truck used for transportation of paperboard to MiniMaid (Knutar, 2019). It was assumed that the truck ran on Diesel B5-EU and its fuel consumption was 68 L per 100 km. The shipment weight was 40 tonnes and, as the cargo capacity was 40 tonnes, the cargo load factor was 100%.

The second leg of the journey was by ferry from Holmsund, Sweden, to Vaasa, Finland (103 km). The default ship size was 10,000 tonnes and the default cargo load factor in the NTM Calculator was 70%. The shipment weight (60 tonnes) included the weight of the paperboard and the truck. From Vaasa, the paperboard was transported to a factory for coating (30 km), by the same truck with the same specifications as in the first leg of the journey. The shipment weight was assumed to remain unchanged even after coating (40 tonnes), as was the cargo load factor. The fourth leg of the journey was from the coating factory to MiniMaid (135 km). The same data as above were inserted into the NTM Calculator (see Table S2).

After the paperboard was converted to paper plates, they were packed and shipped to Uppsala. This route was called “sales route” and was divided into three legs (see Table S2). A shipment of 2 million plates was sent at once to Uppsala. As the average number of plates per box was 600 and one box weighed 277 g (Knutar, 2019), it was calculated that the shipment of 2 million plates was packed in 3334 boxes and the weight of the boxes only was 924 kg. The weight of 2 million plates (17.9 tonnes) was calculated based on the weight of one coated plate (9 g). Thus, the total weight of the shipment was calculated as 19 tonnes. This shipment travelled from the production line in Terjärv, Finland, to Vaasa, Finland (135km). The vehicle type “rigid truck 20-26 t” was used for this and subsequent journeys. It was assumed that this truck ran on Diesel B5-EU and its fuel consumption was 35 l per 100 km. The shipment of 19 tonnes took up its full cargo capacity, so 100% cargo load was assumed.

In Vaasa, the truck was loaded on the ferry and travelled to Holmsund, Sweden (103 km). The ship size and the cargo load factor were as specified above. The shipment weight was 26 tonnes, comprising the shipment itself and the weight of the truck. From Holmsund to Uppsala, Sweden (579 km), the goods were transported by the same rigid truck as in the first leg of this journey.

2.2.2.1. Processing. Processing of material started with processing of wood, which included silviculture and logging. As mentioned above, about 42% of the total GHG emissions arise from the phase of wood harvesting derived from silviculture and logging (González-García et al., 2009). This is due to the use of diesel, petrol and lubricant oils to power and maintain the machinery used in these processes (González-García et al., 2009). Since the total GHG emissions arising from the phase of wood harvesting were calculated above as 1.2 g per plate, the proportion allocated to silviculture and logging was thus 0.5 g.

The processes of pulp and subsequent paperboard making generated 44 kg of CO_2/tonne paperboard produced (Metsä Board, 2018). The weight of one plate was 8.4 g (Knutar, 2019). Hence, the GHG emissions from the pulp and paperboard needed for production of one paper plate were calculated to be 0.4 g.

Next, the paperboard was coated with a layer of polyethylene. It was assumed that low-density polyethylene was applied on the paperboard. According to research by Plastics Europe (2014), the production of 1 kg of LDPE emits 1.9 kg CO_2e. Since one plate uses 0.55 g of LDPE coating (Knutar, 2019), the GHG emissions associated with this process were 1 g.

Finally, upon arrival of the paperboard at MiniMaid, it was pressed into paper plates. Pressing of one plate requires 2.8 Wh (Knutar, 2019). To quantify the amount of GHG emitted from this process, the carbon intensity of the Finnish electricity mix had to be researched. The value was found in Moro and Lonza (2018), who accounted for upstream production and import and export of electricity for each member state of the EU. The carbon intensity of Finnish electricity was thus found to be 211 g CO_2e/kWh (Moro and Lonza, 2018).

2.2.2.2. Packaging. As in the leaf plate system, two different types of packaging were considered, packaging used to protect the goods during transport and retail packaging.

The protective packaging used for transport was corrugated cardboard boxes, each weighing 277 g (Knutar, 2019). For the purpose of calculating GHG emissions from the production of one box used by MiniMaid, the Finnish study by Koskela et al. (2014) was used as a basis for calculations, as in the leaf plate scenario. According to that study, 190 g of corrugated cardboard box emits 0.9 kg GHG. Thus, a box weighing 277 g was calculated to emit 1.3 kg GHG. As there were 600 plates per box (Knutar, 2019), GHG emissions per plate were 2 g.

The retail packaging for MiniMaid plates is made of polyolefin shrink-film (Knutar, 2019). Polyolefin is a collective term for polyethylene and polypropylene thermoplastics (Plastics Europe(a), 2019). These can be of different types; LDPE (low-density polyethylene), LLDPE (linear low-density polyethylene), HDPE (high-density polyethylene) or PP (polypropylene) (Plastics Europe(a), 2019). For this study, LDPE was chosen, as it is commonly applied in packaging shrink-film (Barlow and Morgan, 2013; Plastics Europe(b), 2019). According to an extensive review of 52 plants producing polyolefins in Europe, the GHG emissions for production of 1 kg of LDPE amount to 1.9 kg CO_2e (Plastics Europe AISBL, 2014). In order for the LDPE resin to be used in packaging, it needs to be converted into shrink-film. The amount of electricity required for this conversion was assumed to be the same as that used for producing bio-HDPE film from bio-HDPE resin. This figure (0.5 kWh/kg bio-HDPE film produced) was obtained from the producer of the bio-HDPE film (Högström, 2019). To calculate GHG emissions released from this process, information on the carbon intensity of Finnish electricity mix (211 g CO_2e/kWh) was taken from Moro and Lonza (2018).

2.2.2.3. Disposal. For disposal of the paper plate, the methodology from Eriksson et al. (2015) was applied to quantify GHG emissions from the waste management options available. Their study area was Uppsala municipality, and therefore the waste management facilities would be the same as those available for the paper plate.

Since the paper plate is coated with a plastic (LDPE) layer to make it soak-proof, the option of composting and anaerobic digestion had to be excluded, as it would not decompose fast enough. Thus, the only possibility left for disposal of the paper plate was incineration with energy recovery. In order to calculate GHG emissions and the amount of wood chips substituted by incineration of a coated paper plate, the calorific value of the paper plate had to be researched. The low heat value (LHV) of paperboard (14.8 MJ/kg) was taken from Phyllis2 database (ECN.TNO, 2019) and was assumed to correspond to the LHV of the paper plate studied. As an LHV for coated paperboard could not be found, emissions from burning the plastic LDPE layer were not accounted for in these calculations.

2.3. Scenario analysis

Life cycle assessment can be applied not only to analyse current systems of products and their impacts, but also to assess impacts of possible future changes to the systems. As carrying out an LCA can reveal critical points for improvement in product systems, future scenarios can also serve as projections of impacts of these improvements when implemented.

Since the company producing the leaf plate is a start-up business, we assumed that it can become a well-established business. Business expansion would bring some inevitable changes to the current mode of production, so it was important to estimate the impact of these changes on the environment. A scenario where the leaf plate company was an established business would also place it on an equal footing with the paper plate company. With respect to the paper plate, using fossil-based plastic for its coating eliminated all waste management options other than inciner-

Table 3
Specification of modes of transportation of the leaf plate used in scenario analysis.

Route	Vehicle type	Distance (km)	Load factor (%)	Cargo capacity (m³/t)	Shipment volume/ weight (m³/t)	Source
Leg 1 (DAR-BHU)	Rigid truck 20-26 t	246	100	56 m³	56 m³	(NTMCalc 4.0, 2019; Mehta, 2019)
Leg 2 (BHU-KOL)	Rigid truck 20-26 t	442	100	20.4 t	20.4 t	(NTMCalc 4.0, 2019; Mehta, 2019)
Leg 3 (KOL-GOT)	Bulk carrier	15,663	55	15,000 t	20.4 t	(NTMCalc 4.0, 2019; Mehta, 2019; Searoutes, 2019)
Leg 4 (GOT-UPP)	Rigid truck 20-26 t	453	100	20.4 t	20.4 t	(NTMCalc 4.0, 2019)

ation with energy recovery. Thus, if the paper plate were to be coated with a biodegradable layer, it could be digested anaerobically or composted. In addition, the European Parliament will ban oxo-degradable plastic packaging from the market starting in 2021 (European Parliament News, 2018). It is therefore likely that the plastic packaging currently used for the paper plate will need to be replaced with a biodegradable material.

2.3.1. Leaf plate

In the leaf plate scenario analysis, the plates were shipped only via the sea route and it was assumed that 2 million plates were shipped at once from India to Sweden. This number was chosen as it corresponds to the number of plates the paper plate company ships to customers. If 2 million plates were to be shipped at once, leaf plate production would need to grow exponentially, since it would take over seven years for this number of plates to be produced at the current speed of production at Leafymade.

For transportation, all legs of the journey of the leaf plate were adjusted to suit the scenario with 2 million plates shipped at once. Since the cargo load of all vehicles used on the route between India and Sweden in the current scenario is shared with some other goods, it would be possible to load more plates (and less other goods). Therefore, if the vehicle cargo capacity stayed the same, but the number of leaf plates transported increased, their share of the GHG emissions would increase and the result per plate would stay the same. To reduce the carbon footprint per plate, a vehicle with greater cargo capacity would need to be utilised for the same journey, fully loaded only with leaf plates.

The type of vehicle used between Daringbadi and Bhubaneswar (246 km) was changed from “van” to “rigid truck 20-26 t” with a full volumetric cargo capacity of 56 m^3, as set in the NTM Calculator (2019) (Table S3). Leafymade’s shipment would take up the truck’s full cargo capacity, which was estimated to equate to a volume of 356,790 plates, based on the information that 4 m^3 correspond to 25,485 plates (Mehta, 2019).

The subsequent journey from Bhubaneswar to Kolkata (442 km) would also be carried out by a “rigid truck 20-26 t”, as opposed to a van in the current scenario (Table 3). As 25,485 packed plates weigh 260 kg (Mehta, 2019), 2 million packed plates were calculated to weigh 20.4 tonnes. Thus, the weight of shipment on this journey would be 20.4 tonnes, which is also the full cargo capacity of the truck.

For sea transport from Kolkata, India, to Gothenburg, Sweden (15,663 km), if a ship with the same specifications as in the current scenario were used (full only to 55% of its cargo capacity), the carbon footprint per plate would stay the same. This is because the higher share of GHG emissions generated by a 20.4 tonne heavy shipment, as opposed to 260 kg, would also be divided by a larger number of plates transported (2 million in comparison with 25,485). To reduce GHG emissions from this journey, the ship would need to be fuller, e.g. to 75% of its full cargo capacity. As this is outside Leafymade’s influence, the same default cargo load (55%) as in the current scenario was assumed.

For the last leg of the route, from Gothenburg to Uppsala (453 km), the “rigid truck 7.5-12 t” was changed to “rigid truck 20-26 t” with the same specifications as defined for the second leg of the journey (Table 3).

2.3.1.1. Processing. For processing, it was assumed that with larger production volumes (and necessary investments), the leaf plate could be produced with similar need of input energy as the paper plate. This would mean a three-fold increase in energy efficiency for the leaf plate. For the calculation, the same electricity emission factor for India as in the current scenario (292 gCO_2e/MJ) was assumed.

If plate production were to increase three-fold, sewing of leaves would increase. Sewing is now done on foot-operated sewing machines. No data were collected on the productivity of the workforce in sewing leaves, so it cannot be claimed with certainty that a three-fold increase in production could still be supported using only foot-operated sewing machines. In this case, more workers would need to be employed. However, it is probable that expanded manufacturing of plates would lead to electric sewing machines being used, especially if production were to increase 78-fold. Hence, a rough estimate was made of the production capacity of electric sewing machines and the workforce. It was estimated that an electric sewing machine requires 100 W power (Storgaard, 2018) and it would be used for six full hours per day. This would require 0.6 kWh per day for one machine. It was also estimated that one machine would produce one plate per minute, which would yield 360 plates per working day. In order to produce 2250 plates per day, six electric sewing machines would be required. Thus, six machines working for six hours would need 3.6 kWh of energy to produce 2250 plates.

2.3.2. Paper plate

In the potential future scenario for the paper plate, the current LDPE coating was replaced with coating made from polylactic acid (PLA). It can be produced from a number of starch-rich crops such as maize, rice, potato, cassava or sugarcane (Papong et al., 2014) and was considered here as a biodegradable, water-resistant material. For this scenario analysis, PLA from cassava produced in Thailand was applied. PLA was selected as it seems to be the most popular and feasible material for use on paper-based disposables (Häkkinen and Vares, 2010; Van der Harst et al., 2014; Van der Harst and Potting, 2013). As it is biodegradable, the whole paper plate could be composted or anaerobically digested. In addition, plastic packaging currently used for paper plates was replaced with biodegradable PLA-based plastic.

2.3.2.1. Processing. When calculating the carbon footprint of PLA production, data on GHG emissions were taken from Papong et al. (2014). These emissions did not account for the biogenic carbon stored in the plant. Three different scenarios for GHG emissions from production of PLA were considered. The base case scenario, where 2.48 kg CO_2e per kg PLA resin produced were emitted (Papong et al., 2014); the improved production scenario, where the biogas from wastewater treatment of cassava starch production was utilised, reducing net GHG emissions to 1.96 kg CO_2e per kg resin (Papong et al., 2014); and a scenario with further improvement of production, where a combined heat and power (CHP)

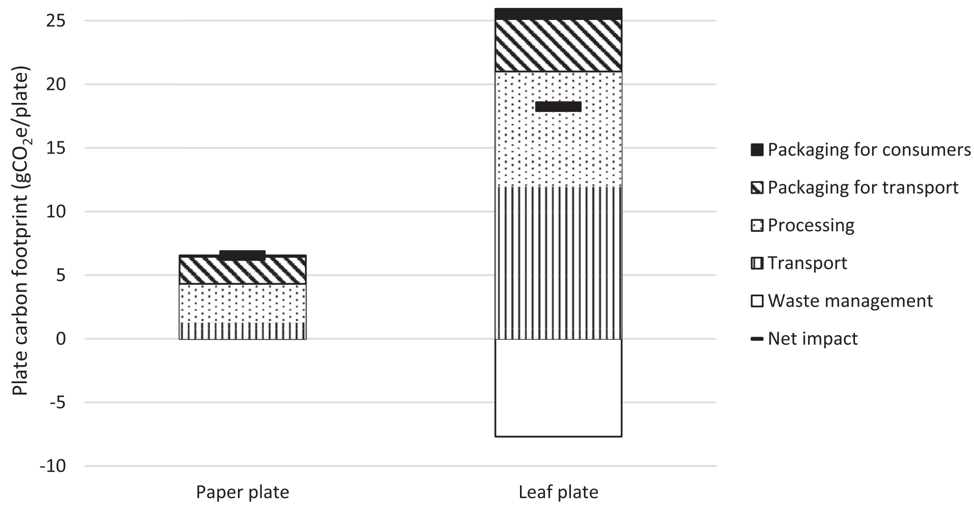

Fig. 2. The carbon footprint of different life-cycle stages and of the overall life cycle of the disposable leaf plate and the paper plate.

system was installed instead of use of grid electricity and steam energy from natural gas, reducing GHG emissions to 1.54 kg CO_2e per kg PLA resin (Papong et al., 2014). Based on these options, the best-case and worst-case scenarios were calculated for the paper plate coating. In the best-case scenario, it was assumed that production of 1 kg PLA resin was responsible for 1.54 kg GHGs. In the worst-case scenario, PLA production generated 2.48 kg GHGs per kg PLA resin. The amount of the coating material needed per plate remained unchanged (0.55 g).

2.3.2.2. Disposal. Since the PLA coating is biodegradable, the whole paper plate could be composted or anaerobically digested in the future scenario. As described earlier, in Uppsala, where the plates would be disposed of, the compost obtained is currently used only to cover landfill in the area, thus not providing nutrients to plants. Therefore, it does not replace any fertiliser and only generates GHG emissions from production of windrows, the composting process, machinery use and transport (Eriksson et al., 2015). Based on the calculation that composting in Uppsala emits 0.043 kg CO_2e/kg composted waste (Eriksson et al., 2015), the emissions associated with one coated paper plate (9 g) were calculated to be 0.4 g CO_2e.

The potential for production of biogas from a coated paper plate was investigated using the methodology applied in Eriksson et al. (2015). The biogas acquired would replace the use of diesel by city buses in Uppsala. In order to quantify the GHG emissions from anaerobic digestion, properties of both paperboard and PLA had to be researched. The moisture content of the paperboard was assumed to be 8%, in accordance with Metsä Board (2018). Values for the volatile matter in the paperboard (78%) and PLA (100%) and the moisture content of PLA (0.1%) were all acquired from the Phyllis 2 database (ECN.TNO, 2019). Polylactic acid was assumed to have the same material properties as low-density polyethylene in the database. The methane yield from paperboard varies depending on its composition and on the pulping method used (Bayr and Rintala, 2012; Carlsson and Uldal, 2009). It was assumed that the methane yield was 0.12 m^3 CH_4 per kg volatile solids (Karlsson et al., 2011, cit. Bayr and Rintala, 2012) contained in the paperboard used in the paper plate. Methane yield of PLA used for the calculations was 0.53 m^3 CH_4 per kg volatile solids, based on Benn and Zitomer (2018). Since anaerobic digestion avoided some use of fossil fuels in comparison with composting, it was used in the scenario analysis of the paper plate.

2.3.2.3. Packaging. The last amendment in scenario analysis of the paper plate life cycle was the material used for retail packaging. Fossil-based LDPE in the current use was replaced by PLA packaging in the future scenario. However, the amount used per plate was so small (0.05 g) that using the more environmentally friendly option gave a negligible improvement.

3. Results

The paper plate had a lower impact on climate change than the leaf plate (Fig. 2). The total carbon footprint of the paper plate was 7 g CO_2e/plate. In contrast, the leaf plate generated larger savings from replaced fossil fuels, but net emissions were still 18 g CO_2e/plate. The disposal option for the leaf plate depicted in Fig. 2 was anaerobic digestion, which was the best-case scenario, avoiding 8 g of fossil GHG emissions. The disposal scenario for the paper plate in Fig. 2 was incineration with energy recovery, which was the only possible scenario, but replacing biofuels and had a negligible impact. When waste management of the plates was not taken into consideration, the leaf plate generated almost four times more GHG than the paper plate per functional unit (Fig. 2).

Closer scrutiny of the results indicated that the hotspots in the life cycle of the leaf plate, contributing the most to the overall impact and with the largest potential for improvement, were transport (12 g CO_2e/plate) and processing of materials (9 g CO_2e/plate). To date, 92% of all shipments of the leaf plate type have been by sea and 8% by air. If all shipments were sent by sea, the leaf plate would emit 5 g CO_2e/plate, while if all shipments were sent by air, its GHG emissions would be 92 g CO_2e/plate. The GHG emissions arising from transport of the leaf plate in the present study were 12 g CO_2e (92% of 5 g plus 8% of 7 g represented 8% of all the shipments so far). Even if sent only by sea, the transport-related carbon footprint of the leaf plate would be 4 g CO_2e/plate higher than that of the paper plate.

The processing-related hotspot in the life cycle of the leaf plate was caused by heat pressing of the plates, which generated 9 g CO_2e. Other manufacturing processes included in the processing stage (collecting, washing and sewing leaves) did not utilise any power other than manpower and thus generated no GHG emissions.

A process making a substantial positive contribution to the carbon footprint of the leaf plate was disposal (-8 g CO_2e/plate). The disposal scenario considered was anaerobic digestion and the saving represented the fossil diesel replaced by biogas generated in anaerobic digestion. Other waste management options were available for the leaf plate, but these either resulted in smaller GHG savings or generated GHG emissions, as opposed to lowering them

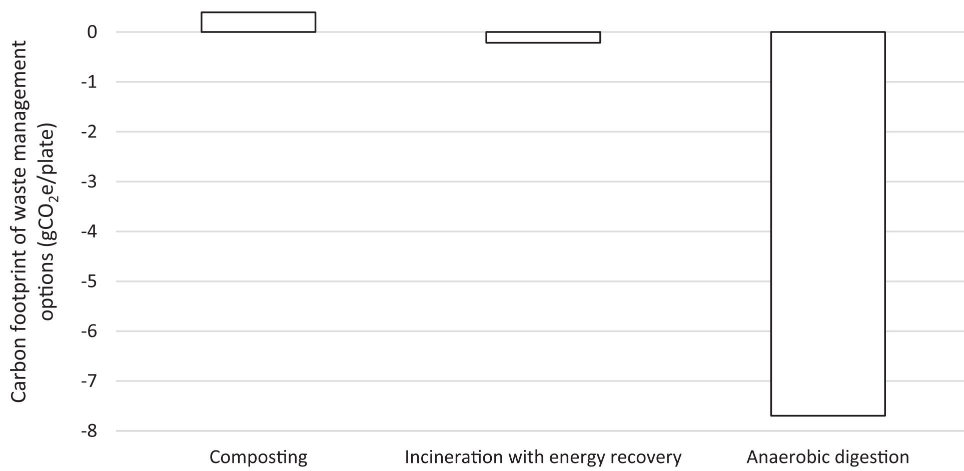

Fig. 3. Global warming potential of the three waste management options for the leaf plate: incineration with energy recovery, anaerobic digestion and composting.

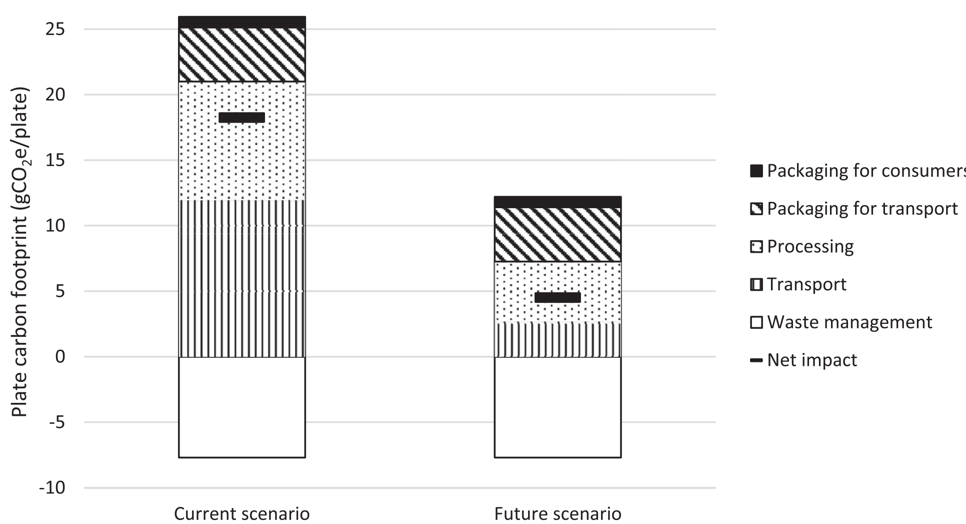

Fig. 4. Global warming potential of the leaf plate in the current scenario and in a future scenario assuming a three-fold increase in production and two million plates shipped to Sweden (by sea).

(Fig. 3). Incineration of a leaf plate with energy recovery could save 0.01 g CO_2e of wood chips, which is normally used for district heating/cooling in Uppsala. On the other hand, composting of the leaf plate gave rise to 0.4 g CO_2e, as it did not replace any other product. The compost produced covered landfill in Uppsala, and hence did not provide nutrients to plants (Eriksson et al., 2015).

In the life cycle of the paper plate, the processing stage contributed most GHG emissions (3 g CO_2e/plate). The processing of the paper plate included extraction of wood, pulp and paperboard making, production of the coating material and pressing, but all these processes combined generated about 6 g less GHG emissions than heat pressing of the leaf plate.

3.1. Results from improvement scenario analyses

In the first improvement scenario analysis, it was assumed that the company producing the leaf plate was a well-established business that could ship two million plates at a time, only via the sea route, and that the productivity of the company was three-fold higher to supply the larger volume. The assumptions on type of packaging and disposal of the leaf plate were the same as in the current system. The disposal option was not changed, as the other disposal options for the leaf plate performed worse in terms of GHG emissions.

In this scenario, the greenhouse gas emissions of the leaf plate life cycle decreased by 13 g, to 5 g CO_2e/plate (Figure 4). The largest decrease was due to transport (-9 g). Transport via the sea route decreased from emissions from 5 g CO_2e/plate in the current scenario to 3 g CO_2e/plate in the future scenario. Processing in the future scenario included emissions from pressing plates (3 g CO_2e) and sewing leaves (2 g CO_2e). If sewing were still done using foot-operated machines, the overall carbon footprint could be further decreased. However, a more realistic option for decreasing the footprint would be an increasing number of plates sewn on electric sewing machines per day.

In the improvement analysis for the paper plate, it was assumed to be coated with a biodegradable layer made from polylactic acid (PLA), which would allow it to be included in the organic waste fraction and therefore digested anaerobically. In addition, fossil-based plastic used for retail packaging was assumed to be replaced with PLA-based plastic. Other processes in the life cycle were assumed to stay the same.

When the paper plate was anaerobically digested, coated with PLA and packed in PLA-based plastic, its total GWP was 3 g CO_2e/plate, as opposed to 7 g CO_2e/plate in the current scenario (Fig. 5). There was a change in emissions from disposal, as anaerobic digestion of the paper plate produced biogas that avoided around 2 g CO_2e/plate more than when it was incinerated. How-

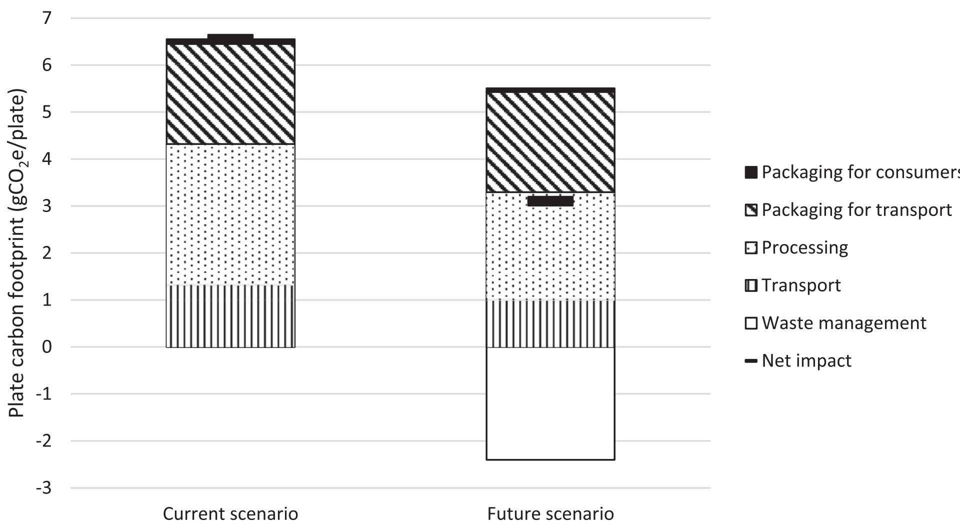

Fig. 5. Global warming potential of the paper plate in the current scenario and in a future scenario where the coating material was assumed to be biodegradable polylactic acid (PLA), the plate anaerobically digested and the retail packaging made from PLA.

ever, the reduction was moderate, due to the low degradation rate of cellulose and hemicellulose and no degradation of lignin in anaerobic conditions (Carlsson and Uldal, 2009; Häkkinen and Vares, 2010). With biodegradable coating, the carbon footprint decreased to 0.8 g CO_2e per plate, from the previous 1 g CO_2e, when the best-case scenario for PLA production (1.54 kg CO_2e/kg PLA) was considered. However, it increased to 1.4 g CO_2e/plate when the worst-case scenario for PLA production (2.48 kg CO_2e/kg PLA) was applied. Therefore, the best-case scenario was used in the improvement analysis for the paper plate. Replacement of fossil-based plastic with cassava-based plastic for retail packaging did not cause any real change in GHG emissions from this process. For comparison, the current GHG emissions per plate with plastic packaging based on low-density polyethylene were 0.09 g CO_2e, compared with 0.08 g CO_2e[3] with PLA-based packaging.

After increasing production and the number of leaf plates shipped, the leaf plate would be able to compete with the paper plate in terms of GHG emissions. However, this would only be the case if the paper plate were kept as in the current scenario. When comparing the future scenarios for both plates, the paper plate would still be the option with less GHG emissions, but the difference would be much smaller than when comparing the current scenarios. This indicates that the leaf plate, the less mature product, has more improvement potential than the market-leading paper plate.

4. Discussion

The results of this analysis showed that a regionally produced paper plate coated with a layer of low-density polyethylene had lower GWP than an uncoated leaf plate, irrespective of shipping distance. When recalculated to the same functional unit, the results were comparable to those in a study by Gautam et al. (2020), which assessed single-use plates made from Areca palm sheets in India and shipped to Portugal. Gautam et al. (2020) reported a carbon footprint of 1033 kg CO_2eq/ton for the Areca palm plate, which is almost half that of the leaf plate considered in the present study (1960 kg CO_2eq/ton), but still higher than that of the paper plate (727 kg CO_2eq/ton). Häkkinen and Vares (2010) compared two paperboard-based disposable cups, one coated with two layers of polyethylene (PE) and the other coated with two layers of PLA based on fermented corn sugars and applied as a biodegradable, water-resistant material. Their results indicated that the cup coated with (bio-based) PLA had slightly higher GWP than the cup coated with PE. Similarly, in the present study a bio-based leaf plate had a higher climate impact than a bio-based paper plate coated with fossil-based plastic. Van der Harst et al. (2014) considered PLA production far from the site of use, but did not find a single best material for disposable cups when comparing the environmental performance of polystyrene, PLA and paper lined with bioplastic across multiple impact categories.

The results of the present study should be interpreted with caution, as it focused solely on the performance of the two disposable plates and only took into account one impact category (GWP). If acidification were taken into consideration, the paper plate might have had a higher impact than the leaf plate, due to the use of various potentially acidifying chemicals in pulp and paperboard making. Furthermore, previous studies have taken into consideration diverse waste management options (including landfill), giving disposable cups varying amounts of credit. Many materials do not decay in landfill, and thus do not release any GHGs (Häkkinen and Vares, 2010). Similarly, recycling certain materials may release less GHG emissions than composting (Van der Harst and Potting, 2013). Thus, waste management choices and associated credit allocations can strongly influence the overall GWP of products (Häkkinen and Vares, 2010; Van der Harst et al., 2014; Van der Harst and Potting, 2013). This is illustrated by a previous review of life cycle assessments of 10 disposable cups made from various materials (Van der Harst and Potting, 2013), which showed that no material was consistently better than the others. The varying results for GWP can be due to multiple factors, such as production processes, waste processes, allocation options, data used etc. (Van der Harst and Potting, 2013). However, a later study identified the three processes with the highest environmental impact as being production of the basic material for the cup, cup manufacturing and waste management (Van der Harst et al., 2014). The hotspots identified in the present study for leaf and paper plates were almost identical (manufacturing, transport, waste management).

4.1. Hotspot analysis

The process contributing most to the GWP of the leaf plate was transport, followed by plate manufacturing. In order to decrease GHG emissions from these processes, it is necessary to examine

[3] The best-case scenario applied, with 1.54 kg GHG/kg PLA resin emitted.

their origin. The leaf plate first travels within India during production and later to Sweden for sale and use. The mode of transportation strongly influences the amount of GHG emissions generated, e.g. transporting the leaf plate by air increased the transport contribution to its carbon footprint by 18-fold compared with transportation by sea. Some plates are sent by air for faster delivery (Mehta, 2019). Production of the leaf plate also has social aspects, e.g. it provides employment for tribal women living in the area in India where the sal tree grows. They collect abundant fallen leaves in the local rainforest and receive training on sewing the leaves from a local non-government organisation, so that they can earn extra income (Mehta, 2019).

The manufacturing hotspot (processing leaves and turning them into a leaf plate) had only one step generating GHG emissions, heat pressing. It was the only process requiring electricity, as leaves were collected manually and sewn on foot-operated sewing machines. However, heat pressing gave rise to 9 g CO_2e/plate, compared with 3 g CO_2e/plate generated in processing materials for the paper plate, which included extraction of wood, pulp and paperboard making, coating and pressing, all requiring different types of energy (including electrical). The key factor behind the difference in GHG emissions was the source of energy. India derives the majority of its electrical energy from coal (Central Electricity Authority, 2019), hence its high carbon footprint. Finland, where the paper plate was produced, produces the majority of its electricity from carbon-free sources (Finnish Energy, 2019) and regards biomass fuel as carbon-neutral, as it is composed of biogenic carbon. Wood processing industries, such as pulp and paper industries, are a great example of efficient use of biomass waste from production for powering internal manufacturing processes. They use black liquor and forest residues to produce the steam and electricity needed for pulp and paper manufacturing (Mantau, 2012), making it carbon-neutral. This was the case for the pulp and paperboard mill considered in the paper plate life cycle (Metsä Board, 2018). Therefore, even though more electricity and other types of energy were used in the whole processing stage of the paper plate system, it still had a lower carbon footprint than processing of the leaf plate.

Although the paper plate performed better than the leaf plate in terms of GHG emissions in this study, there is still room for improvement in the processing stage of its life cycle. Harvesting wood involves use of heavy machinery, which runs on fossil fuels. Based on data in González-García et al. (2009a), which were used for the calculations in this study, silviculture and logging are responsible for 0.5 g CO_2e/plate. Pulp and paperboard making contributed about 0.4 g CO_2e per plate, as a substantial amount of the energy consumed in production came from biomass. Another source of GHGs was pressing paper plates (0.6 g CO_2e/plate). However, the highest contribution of GHG emissions within processing originated from production of the coating material (1 g CO_2e/plate). In this case, the material was fossil-based low-density polyethylene (LDPE). The life cycle of LDPE production comprises a multitude of complicated and energy-intensive processes that give rise to 1.9 kg CO_2e/kg LDPE resin produced (Plastics Europe AISBL, 2014). If a different material were used, coating could potentially be less energy-intensive and thus have a lower impact on overall GWP. The material used for coating also influenced the end-of-life options for the paper plate. If the LDPE coating were replaced with a biodegradable layer, the paper plate could be digested anaerobically or composted.

There is room for improvement in the leaf plate life cycle. For example, pressing the leaf plate used three times more electricity per plate (31 KJ) than pressing the paper plate (10 KJ). Electricity use per plate could be improved by increasing the volumes processed, if machinery could be used more efficiently, or emissions could be reduced by switching to more renewable energy sources. If production increased, large shipments could be dispatched, lowering emissions per plate from transport. Expanded production could also avoid air freight of plates to Sweden, as there would be enough product to build up stocks. The improvement analysis for the leaf plate considered a future scenario where the company was a well-established business and shipped two million plates at a time, only by sea.

4.2. Improvement analysis

The improvement analysis for the leaf plate showed that the GWP could be decreased substantially with all transport by sea and production increased without increasing electricity consumption. If production expanded further, more electricity would be required for sewing leaves. In that case, attention should be paid to increasing production capacity per electric sewing machine. After increasing production and volumes of leaf plates shipped, GWP of the leaf plate was lower than for the paper plate. However, when comparing the future scenarios for both plates, the paper plate would still be the option with less GHG emissions. Both leaf plate scenarios considered the best option for waste management, so no further improvements are possible there. Further improvements are still possible in terms of packaging. For transport between businesses, lighter corrugated cardboard boxes could be used and/or more plates could be packed into each box, although there are already more leaf plates per box (1350) than paper plates (600). For retail packaging, pack size (currently 12) could be increased, decreasing packaging amount per plate.

A radical change that would greatly reduce the carbon footprint of the leaf plate would be to locate production in Sweden and start producing plates from Swedish leaves. The carbon footprint of production would then be lower, due to Sweden's less carbon-intensive electricity production. For comparison, the 5 g CO_2e/plate from processing in the future scenario with expanded leaf plate production in India would decrease to 0.21 g CO_2e/plate with production in Sweden. However, the social aspect of production in India, providing employment and rising standards of living for local inhabitants, would then be lost. If production remains in India, its carbon footprint could be lowered by using renewable sources of energy. Like the pulp and paperboard mill in the paper plate scenario, the leaf plate production unit could be powered by its own biomass waste. As leaf waste gives most energy when digested anaerobically, a small-scale biogas plant or community biogas plant ("Small Scale Biogas Design", 2015) could be installed on the premises. Electricity generation could be backed up by installation of solar panels, if electricity from biomass proved insufficient.

For the paper plate, coating with a biodegradable layer is preferable to a fossil-based layer, even if production of the biodegradable material emits more GHG than the fossil material. This is because the final emissions from decaying biodegradable coating are carbon-neutral, due to carbon uptake by the biological material during plant growth. Biodegradable coating also gives the paper plate more waste management options. If the amount of retail packaging per plate remains the same as at present, choice of packaging material will have almost no impact on the GWP of the whole life cycle of the paper plate. However, as fossil-based plastic has severe impacts on the environment, no matter how much is produced, a material which is recyclable and biodegradable at the end of its life should be used in retail packaging.

In the future case for the leaf plate, increasing the efficiency of production could lead to a rebound effect of increased consumption, with negative consequences. The rebound effect occurs when processes become more efficient and the product becomes cheaper, stimulating higher demand for the product and/or other products, which become more affordable due to the savings made (Berkhout et al., 2000). However, one of the major features of the leaf plate is the social benefits of providing work opportunities in less industrialised areas, hence the higher price compared with the paper plate. Therefore it seems unlikely that this product will replace the much cheaper mass-produced paper plates or plastic equivalents, but will rather be a complement to them. However, as concluded by Herberz et al. (2020), single-use items are harmful to the environment regardless of the material from which they are made. Disposables were invented for sanitary reasons, but their use has now extended to areas of everyday life where they could be easily replaced by reusable counterparts and thus cut down on the waste generated. It is questionable whether a change of material is enough, or whether all consumption of single-use items needs to cease. However, this would require a change of mindset whereby environmental protection and sustainability are placed above comfort.

5. Conclusions

This comparison of two types of disposable plates showed that the leaf plate had substantially higher GWP than the paper plate, due to its long-distance transport and use of coal-based electricity in processing the raw material. When leaf plate transport was only by sea and larger vehicles were fully loaded with more plates, the impact of transportation dropped significantly. With increasing efficiency of production and changing the source of energy used, the GWP of processing also decreased. Thus, it can be concluded that long-distance transport and mode of transportation (sea, air) can strongly influence the overall climate impact of a product and should be considered when choosing shipment options for goods. Efficiency of production and the source of energy used in manufacturing processes are crucial for the GWP of products, as they can be the largest contributors to total carbon footprint.

For the paper plate, a significant amount of the energy used for processing wood fibres came from biomass and the overall carbon intensity of electricity used in the region of production was significantly lower than for the leaf plate. Thus, although manufacturing of the paper plate included many more energy-requiring operations than processing of the leaf plate, it emitted only one-third of the GHGs generated by processing of the leaf plate. Manufacturing was still a hotspot in the life cycle of the paper plate, with the highest GHG emissions originating from manufacturing of the coating material. Coating type (plastic/biodegradable) was thus critical for the carbon footprint associated with manufacture, and also waste management, of the paper plate.

This study assessed two materials that could possibly replace plastic in single-use products. However, since increasing demand for disposables poses a threat to sustainable consumption, it is questionable whether a change of material is enough, or whether all consumption of single-use items needs to cease, which might not be a realistic scenario looking at current trajectories in production and consumption, especially in the food service sector.

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.

Acknowledgements

This research was funded through the H2020 ERA-net Cofund on Sustainable Food Production and Consumption (SUSFOOD2) by the Swedish Research Council for Sustainable Development (Formas), grant number FR-2018/0001. The authors would like to thank the companies Leafymade and MiniMaid for their time, collaboration and willingness to share the information making the study possible.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.spc.2020.08.004.

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