TALLINN UNIVERSITY OF TECHNOLOGY
SCHOOL OF ENGINEERING
Department of Civil Engineering and Architecture

LIFE CYCLE ASSESSMENT OF REUSABLE PLASTIC TABLEWARE

KORDUSKASUTUSNÕUDE ELUTSÜKLI ANALÜÜS

MASTER THESIS

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AUTHOR’S DECLARATION

Hereby I declare, that I have written this thesis independently.
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THESIS TASK

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1. To conduct LCA of 3 reuse systems offered by 3 companies in Estonia and of a single-use system.

2. To compare the results of the LCAs on reuse system to the single-use system.

3. Analyse the results and propose solutions for ecodesigning the reuse system.

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CONTENTS

PREFACE

Since January 1st, 2024, single-use tableware is prohibited in public events in Estonia. Several service providers offer reusable plastic tableware for events in Estonia. This service includes the transport of the dishes to and from the event and washing of the dishes. Studies have shown that reusable tableware demonstrates significant reduction in environmental impacts. The impulse for this study came from the Circular Economy Department of the Tallinn Strategic Management Office who was interested to learn about the environmental performance of reusable tableware systems provided in Estonia.

In this study a comparative life cycle assessment according to the ISO standards 14040/14044:2006 is conducted, where the tableware reuse systems provided by three companies are juxtaposed with a hypothetical single-use system in an Estonian context. The impact assessment was done using OpenLCA software and Ecoinvent v3.11 database. The data for the reuse systems was collected from the three companies.

The results of the life cycle assessment indicate that the reuse systems of two companies have a better environmental performance than the single-use cup and plate already from 25 uses. For the third company, the environmental impacts are less significant compared to the single-use system after 50 uses. The most relevant environmental impact categories in this study turned out to be climate change and fossil resource use.

Keywords: life cycle assessment, circular economy, plastic tableware, single-use plastic, reuse, master thesis

List of abbreviations and symbols

CE - Circular Economy

CO – Carbon monoxide

CO2 – Carbon dioxide

EoL - End-of-Life

EPS - Expanded polystyrene

EU - European Union

GHG – Greenhouse Gas

LCA - Life Cycle Assessment

LCIA - Life Cycle Impact Assessment

N - Nitrogen

NH3 - Ammonia

NOx – Nitrogen oxides

P - Phosphorus

PC - Polycarbonate

PE - Polyethylene

PET - Polyethylene terephthalate

PLA - Polylactic Acid

PP - Polypropylene

PS – Polystyrene

SAN - Styrene Acrylonitrile

SO2 – Sulphur dioxide

SOx – Sulphur oxides

SUP – Single-use plastic

1. INTRODUCTION

Globally approximately 400 million tonnes (MT) of plastics are produced annually and it is increasing year by year (Plastics Europe, 2024). Due to its durability, versatility and cheap production plastics are used in many areas of life, and very often for disposable food packaging. The growing demand for takeaway food increases the use of single-use plastic (SUP) containers, which follow the linear "take-make-discard" model. Coupled with increasing amounts of plastic retail food packaging the linear consumption patterns with incorrect disposal and insufficient recycling capacities are causing concern among scientists and lawmakers alike (Directive 2019/904).

Large events, the take-away food industry, in many cases even offices, hotels and hospitals serve food and drinks to their customers on SUP tableware daily. That is an enormous amount of plastic, which is used only for a few moments and is thereafter mostly incinerated, landfilled or, in worst cases, discarded as litter causing plastic pollution in our terrestrial and aquatic environments. 50% of marine litter in the European Union (EU) are SUP items such as forks, knives, plates and cups Directive 2019/904). It is evident that the amount of SUP used needs to be reduced to minimise plastic waste and avoid plastic pollution.

To shift from a linear system to a circular economy (CE), it is essential to eliminate waste and pollution and keep products and materials in circulation (Ellen MacArthur Foundation, n.d.). In the context of plastic food and beverage packaging for on-site consumption this means substituting SUP with reusable tableware. Prompted by the ban of SUP tableware for selling food or drinks at events in Estonia (Pakendiseadus, 2004), a number of local companies with product-as-a-service business models have emerged. These companies rent reusable tableware for events, taking care of the transportation and the washing of the tableware.

In the effort to transition into a more circular economy, it is important to understand if substituting SUP for reusable alternatives does in fact reduce environmental impacts and what are the effects this transition entails. Although studies have demonstrated that reusable options generally perform better than SUP tableware, many parameters that are sometimes related to the geographical context, can influence the environmental impact of the reuse system (Lewis et al., 2021).

The aim of this thesis is to understand the environmental burden of reuse systems for tableware in Estonia in comparison with a SUP alternative. In order to quantify the environmental impact a life cycle assessment (LCA) with a cradle-to-grave approach is conducted in accordance with the International Organization for Standardization (ISO) 14040/14044 standards (ISO, 2006). LCA is a widely recognised approach to study the environmental aspects of a product or service across its entire life cycle. The life cycle model is created in OpenLCA programme (developed by GreenDelta). For the reuse system, mostly primary data collected from the companies is used. For the single-use system, secondary data from literature is applied. For this thesis reusable tableware from three service providers in Estonia are compared to a single-use alternative. The environmental impact of reusable and single-use tableware will be analysed in 16 categories across all life cycle stages employing the Environmental Footprint v3.1 method. This will allow to determine potential shifts in the environmental impacts across the two systems.

Furthermore, sensitivity analyses are conducted to determine the importance of transportation distances and return rate on environmental impacts during the life cycle of the reusable tableware.

The second chapter begins with an introduction of the EU policy on plastics. This is followed by an overview of the Estonian legislation regulating the use of SUP and an outline of targets for recycling. The core of the second chapter constitutes an in-depth presentation of existing LCAs comparing single-use and reuse systems.

The third chapter describes the methodology of an LCA and details the LCA conducted in the framework of the thesis. It includes the goal and scope definition, the life cycle inventory and the description of sensitivity analyses.

The fourth chapter presents the results of the life cycle impact assessment.

In the fifth chapter the results are interpreted by analysing the outcomes of the sensitivity analyses and baseline study. Most relevant impact categories are identified and are then evaluated by life cycle stage. The results of the LCA are also compared to the case studies introduced in the second chapter.

In conclusions, recommendations are made for ecodesigning the reuse system in Estonia based on the insights from the LCA and the matching case studies.

2. THEORETICAL OVERVIEW

Plastic is a durable, lightweight and cheap material which revolutionised manufacturing, resulting it being ubiquitous. It is estimated that approximately 8300 million tons (Mt) of plastic have been produced since large-scale production started in the 1950s. 60% of this plastic - around 4900Mt – is no longer in use and has ended up in landfills or in the natural environment. It is also a sobering fact that only 9% of that plastic has been recycled and approximately 12% has been incinerated. As plastic is so cheap and easy to shape and mold, around 42% of all nonfiber plastics have been used for packaging, which is mostly single use. (Geyer et al., 2017)

However, the durability of plastic is also what makes it a large source of pollution. Weak or non-existent waste management systems along with the abundance of plastic products and littering have allowed plastic to wind up in terrestrial environments but also aquatic ecosystems and eventually the ocean (Meijer et al., 2021). Since plastic either remains intact in the natural environment or breaks down into smaller fragments (Barnes et al., 2009), it accumulates in the ecosystems and food chains, thereby posing a threat to humans (Genovesi et al., 2022). European Commission has indicated that more than 80% of marine litter in the EU is plastic and SUP items represent 50% of the total (Directive 2019/904).

Plastic also endangers human health. Microplastics have been detected in the human bloodstream, with polyethylene (PET) and polystyrene (PS) being the most common types (Leslie et al., 2022). Microplastic has even been found in stool samples, with polypropylene (PP) being the most prevalent (Zhang et al., 2021). In addition, chemicals that are used to produce plastics might have adverse effects on human health. For example, bisphenol A (BPA), which is used to produce polycarbonate (PC), can interfere with the normal performance of the hormonal system, have reprotoxic effects and has been linked to certain types of cancer (Ma et al., 2019). Styrene, which is used to produce polystyrene (PS) and expanded polystyrene (EPS), is a recognised carcinogen (National Toxicology Program, 2011).

40% of plastics in the EU is used to produce packaging and this, in turn, constitutes 36% of municipal solid waste (Regulation 2025/40). Often the argument in favour of SUP is that they can be recycled and thereby the environmental impact of the product is reduced. In effect, most of the plastic waste generated in the EU is incinerated for energy recovery and this has even increased 15% since 2018. Furthermore, a whopping 17% of plastic packaging waste was landfilled and only 37,8% was recycled in 2022, which is set to miss the target of 50% by 2025 established in the Packaging and Packaging Waste Directive (Plastics Europe, 2024). In 2022, Estonia recycled 44% of the total plastic packaging waste generated that year (Eurostat, 2022). Furthermore, of the 58,8 MT of plastic produced in the EU in 2022, only 13,5% was made from recycled plastic (Plastics Europe, 2024).

EU's mechanical recycling capacity in 2021 was only 11 Mt (Plastics Recyclers Europe, 2021). Therefore, half of the generated plastic waste is exported to third countries for treatment. In 2023 Türkiye was the largest destination of recyclable plastic waste with 22% of total extra-EU plastic exports, followed by Malaysia (21%) and Indonesia (19%) (Eurostat, 2023). Considering that these countries are also topping the charts in mismanaging plastic waste, meaning it is not properly recycled or incinerated, plastic may leak to the sea due to poorly managed landfills (Meijer et al., 2021). It's also important to recognise that many SUP items are small (straws, cutlery etc.) and they are often not recycled as they fall through the screens of the recycling equipment (The Association of Plastic Recyclers, 2018).

One solution to reduce the amount of plastic waste and plastic pollution is to transition into circular economy by applying a waste management concept higher in the waste hierarchy than recycling and recovery, and that is implementing reuse systems and preventing plastic waste in the first place (European Commission, 2018). As packaging forms almost half of all nonfiber plastics produced (Geyer et al., 2017), it makes sense to try to curb the amount of single-use packaging. In the case of food containers including tableware reusable alternatives to single-use items have already been introduced in many countries around Europe and more and more event organisers, including in Estonia, are adopting reusable tableware to improve the environmental footprint of their operations (Campbell et al., 2020; Pladerer et al., 2008; Vercalsteren et al., 2006; Walker et al., 2024).

However, both the single-use and the reuse system have its benefits and drawbacks. The single-use packaging has a lower mass and lower energy consumption, but it also has a higher littering potential. Reusable packaging has lower littering potential and higher recyclability, but on the other hand the use stage has a significant effect on the environmental impact of the reuse system (Yadav et al., 2024). Because the scope of the environmental impacts of the two systems are not clear, LCA studies have been carried out to quantify the environmental performance of both systems.

2.1 Regulations

The EU has admitted that the rise of take-away food and drink in single-use packaging has led to increasingly inefficient and linear production and consumption patterns and has therefore taken several initiatives to curb plastic pollution (Directive 2019/904). EU has actively aimed to reduce SUP in food packaging since 2022 but has addressed the problem of plastic packaging waste in its legislation for decades.

The Waste Framework Directive establishes the main concepts and definitions for waste management, including what qualifies as waste, recycling, and recovery. It also establishes the waste hierarchy which prioritises waste prevention in waste management, followed by reuse, recycling, recovery and disposal in a descending order. The waste hierarchy laid down in this directive is referred to in many of EU's legislations. (Directive 2008/98)

The Directive on Packaging and Packaging Waste (PPWD) enacted measures to prevent packaging waste, by promoting the reuse of packaging and encouraging the return and collection of packaging waste for recycling. The directive also set recycling targets for packaging waste: 50% of plastic packaging waste must be recycled by 2025. PPWD is now a part of the Packaging and Packaging Waste Regulation. (Directive 94/62)

EU's Strategy for Plastics in a Circular Economy aims to have all plastics packaging in EU either reusable or recyclable by 2030, encouraging innovation in design and business models. The strategy proposes that measures should be developed to minimize the unnecessary production of plastic waste, particularly from SUP items and to promote the reuse of packaging. The strategy also highlights the importance of innovation and circular solutions in tackling plastic waste. (European Commission, 2018)

In 2019 the EU's Single-Use Plastics Directive (SUPD) was passed, which aims to cut down the volume of ten SUP products that form half of the littering on the beaches in the EU. For items that have sustainable alternatives and are more easily replaceable the directive restricts their access to the EU market. Such items include single-use plates, cutlery and cups made of expanded polystyrene (EPS). In addition, it is emphasized that SUP items made of bio-based and biodegradable plastics also fall under this directive. (Directive 2019/904)

The European Green Deal, presented in the end of 2019, sets the ambitious goal for Europe to be the first climate-neutral continent in the world. It recognizes the existential threat that climate change and environmental degradation pose to Europe. The Green Deal builds on three main principles: achieving net-zero greenhouse gas emissions by 2050; economic growth that is independent of resource consumption and inclusivity, leaving no one and no place behind. The green deal also explicitly indicates the efficient use of resources and achieving a circular economy. It includes a roadmap with actions that aim to boost circular economy in the EU and that sets a goal to adopt a new circular economy action plan. (European Commission, 2019)

One of the goals of the new Circular Economy Action Plan (CEAP) is to expand the Ecodesign Directive beyond energy-related products, making the Ecodesign framework applicable to a wider range of products and ensuring it supports circularity. In the CEAP the Commission reiterates the need to limit the use of disposable items such as cups and plates. It also highlights creating incentives to build new business models that are based on product-as-a-service. (European Commission, 2020)

The Ecodesign for Sustainable Products Regulation (ESPR) that entered into force in July 2024 establishes a framework of ecodesign requirements for almost all product categories. These requirements include but are not limited to setting guidelines for carbon and environmental footprint methods and limiting the amount of waste, including packaging waste. ESPR underlines the importance of uptake of circular business models which support decoupling economic growth from resource use, one of the main principles of the Green Deal. ESPR is key to fulfilling the objectives of the 2020 Circular Economy Action Plan. (Regulation 2024/1781)

In the end of 2024 EU adopted the Packaging and Packaging Waste Regulation which addresses the entire lifecycle of packaging and re-highlights the importance of applying the waste hierarchy in the member states. It also requires Member States to reduce packaging waste 5% by 2030, 10% by 2035, and 15% by 2040, compared to 2018 levels. In addition, the regulation aims to restrict the use of BPA in food packaging and other food-contact materials. It’s important to note that packaging in this regulation means also cups, food containers and plates that are intended to be filled with food/drink or already contain food and beverages at the point of sale. (Regulation 2025/40)

The Waste Act of Estonia sets general requirements for waste management organisation and prevention of waste. A part of the SUPD is transposed in the Waste Act. In that it includes a definition of single-use plastic and restriction of certain SUP products on the market. (Jäätmeseadus, 2004)

The PPWD is transposed in the Packaging Act of Estonia, which establishes that at least 50% of plastic packaging waste must be recycled by 31 December 2025. The act also sets up measures, in line with SUPD, to reduce consumption of SUP products and restricts the use of food containers made of PS. A step towards circular economy was made on January 1, 2024, in Estonia, when only reusable tableware was allowed for the purpose of serving food and drinks at public events in Estonia. (Pakendiseadus, 2004)

Tallinn was the first municipality in the country to implement a restriction on SUP already before it came into force with the national Packaging Act, when it banned the use of disposable plastic tableware at public events since October 1, 2019 (Tallinna jäätmehoolduseeskiri, 2019). On June 1, 2023, Tallinn banned the use of all disposable tableware, including those made of biodegradable plastic (Tallinna jäätmehoolduseeskiri, 2023).

Other EU countries have taken similar actions against single-use items. Belgium made the use of reusable cups mandatory in public events from 1 September 2023 (Royal Decree 2022020004). Since 1 January 2023 single-use tableware in fast-food restaurants is banned in France (Loi n° 2020-105). In the Netherlands disposable cups are not allowed for consumption on HoReCa, offices or events premises since 1 January 2024 (Netherlands Enterprise Agency, 2025).

2.2 Case studies

Together with the first emergence of reusable cups at public events and ambitions of public events, such as music festivals and football games, to be more sustainable, studies comparing the environmental performance of SUP and reusable cups have been conducted for decades. The studies have used the LCA methodology to quantify the environmental impact through the product’s whole life cycle. There are also many LCAs carried out for various reusable packaging systems, including tableware and takeaway containers. As this thesis is focused on the life cycle assessment of reusable plastic tableware from a system point of view, results of recent studies examining reuse systems of plastic packaging are presented. All studies listed below are cradle-to-grave LCAs unless stated otherwise.

2.2.1 Plastic tableware

SUP tableware like cups and plates are used at public events around the world, because they are easy and cheap to obtain, convenient to use and dispose of. Single-use tableware is usually lightweight, as a higher weight of the material impairs the environmental performance of the disposable product (Lewis et al., 2021). Disposable cups are commonly made of PET (polyethyleneterephthalate), PS, PP, paperboard coated with polyethylene (PE) and biodegradable PLA (polylactide) (Cottafava et al., 2021). Materials for single-use plates are PP, PS, paperboard coated with PE or not (Lewis et al., 2021).

Reusable tableware must have similar properties like stackability and functionality as single-use tableware to ensure that they are adopted by the catering companies (Pladerer et al., 2008). Reusable cups and plates are commonly also produced from PP or PC, but in order to make the tableware long-lasting, reusable tableware is generally heavier. However, the weight of reusable tableware has a lesser effect on its environmental performance because the mass per use for reusable tableware is lower than for disposable tableware (Lewis et al., 2021).

2.2.2 Biodegradable tableware

Biobased and biodegradable plastics are sometimes seen as a sustainable alternative to fossil-based plastics, as they are generally thought to be compostable (Genovesi et al., 2022). It has been shown that disposable bio-based plastic tableware (PLA and Polybutylene succinate (PBS)) has a lower impact than single-use PP items (Genovesi et al., 2022). Yet, some studies have demonstrated that although single-use biodegradable tableware perform better than the disposable fossil-based counterpart in most categories, they have a larger impact in ozone depletion and aquatic eutrophication (Genovesi et al., 2022; Goodrum et al., 2024). This means that the environmental impacts are shifted rather than avoided (Jürgens & Enders, 2024). The larger impacts in ozone depletion and eutrophication are due to the various steps required to cultivate sugar cane or other raw materials needed for bioplastic production. The composition of the compostable items and the processing technology of the raw materials influence the results, too (Genovesi et al., 2022). If the biobased plastic were produced from organic waste material instead of plants that need to be fertilised and irrigated, it could be a more sustainable alternative to fossil-based plastics (Jürgens &

Enders, 2024). Still studies have demonstrated that even when biobased plastics like PLA have smaller environmental impacts in some categories than PP, reusable containers made of PP still perform better overall than single-use PLA plastics (Hitt et al., 2023). This is caused by high methane emissions from the composting process of the biodegradable plastic, which impairs its environmental performance (Pladerer et al., 2008; Walker et al., 2024).

2.2.3 Reuse vs single-use systems

Many studies conclude that the reuse scenario has generally lower environmental impacts than the single-use alternative (Aggarwal, 2024; Cottafava et al., 2021; Genovesi et al., 2022; Goodrum et al., 2024; Greenwood et al., 2021; Hitt et al., 2023; Lewis et al., 2021; Miele et al., 2024; Walker et al., 2024; Yadav et al., 2024). When comparing the two systems – single-use vs reuse – it becomes evident, that the performances of the two systems vary across the life cycle stages. The environmental impacts of a single-use system are largest in the production and end-of-life (EoL) stage (Genovesi et al., 2022; Lewis et al., 2021). In contrast, for the reuse system the manufacturing stage has the highest impact only when the reusable container is used for a few times (Aggarwal, 2024). However, when the number of uses increases the impact of the production stage decreases and the most significant is the use stage due to washing of the containers (Castellani & Cardamone, 2022; Lewis et al., 2021; Yadav et al., 2024). Even in cases where a sensitivity study was conducted to compare the differences in EoL scenarios or electricity production, reuse systems demonstrate better environmental performance than single-use systems (Greenwood et al., 2021; Hitt et al., 2023).

In addition to an LCA, Hitt et al. (2023) analysed the life cycle costs of the two systems and concluded that the reusable container with 20 uses also has lower life cycle costs by 25%–36% compared to single-use containers. The majority of life cycle costs stem from the initial purchase, with the reusable container's initial cost making up about 93% of its total life cycle costs, and the single-use container's initial cost accounting for approximately 99% (Hitt et al., 2023). In contrast, a study in a Swedish student restaurant, the system of reusable takeaway containers proved to be 3,3% more expensive than the single-use counterpart, with the assumed number of uses also to be 20 (Aggarwal, 2024). These contrasting results highlight the importance of the design of the reuse system.

However, there are also LCA studies that have shared mixed results (Campbell et al., 2020; Potting & van der Harst, 2015; Vercalsteren et al., 2006.) or even the reverse (Blanca-Alcubilla et al., 2020; Castellani & Cardamone, 2022). The study by Castellani and Cardamone (2022) concluded that single-use tableware made of paperboard (some coated with PE) outperforms reusable tableware made of PP. In the case of that particular study it must be noted that it was commissioned by the European Paper Packaging Alliance. Therefore, some assumptions, such as the rather low reuse rate (50) or the decision to increase energy use for drying of the dishes by 30% could support the perspectives of the organisation that has commissioned the study (Walker et al., 2024). Still, in line with the Castellani and Cardamone (2022) study others also have demonstrated that the PE coated paperboard cup is the best performing single-use cup, when compared to PP, PET and PLA disposable cups (Cottafava et al., 2021). Regards to the study by Blanca-Alcubilla (2020), which assessed the environmental impact of reusable tableware in the aviation sector, the largest impact stems from the flight stage because reusable tableware, e.g. steel cutlery, is a lot heavier than single-use counterparts, increasing the greenhouse gas (GHG) emissions during the flight.

Furthermore, there have also been studies where the results do not allow to decide for preference of either the reusable or any disposable cup. Vercalsteren et al. (2006) reached the conclusion that there was no significant difference between the environmental impacts of the reusable PC cup and the single-use PP, PLA and PE lined cardboard cups. Interestingly, Potting and van der Harst (2015) argued that the impact results for reusable ceramic cups are too uncertain and too close to the disposable cups (PLA, PS) and therefore it cannot be concluded that the reusable cup should be preferred. Cottafava et al. (2021) inferred that the single-use option is preferable since the transport distances during the use phase of the reusable container outweigh the gains in other areas.

Many studies have calculated a breakeven point for reuse options, which demonstrates the number of times the reusable item needs to be used before it achieves the same environmental performance per use as the SUP option. This means that in order for the reusable option to be efficient, it needs to be used more times than the breakeven point value (Aggarwal, 2024). However, the breakeven point values vary from 6 to 1571 depending on the type of product (tableware or take-out container), design of the reuse system and the single-use alternative compared, but also on the chosen impact category (Cottafava et al., 2021; Hitt et al., 2023; Miele et al., 2024; Yadav et al., 2024). For example, when taking into account only the global warming potential (GWP), the breakeven point for reusable PP containers in Finland is as low as 6 (Yadav et al., 2024). In Sweden the breakeven point of GWP for reusable bio-PP containers was calculated to be 7, but when considering all the impact categories the value was 20 (Aggarwal, 2024). However, when a reusable PLA cup was compared to a single-use PET cup in the eutrophication impact category, the breakeven point was 1571. In the climate change impact category, the breakeven point for the same comparison was only 24 (Cottafava et al., 2021).

As with any study, the methodological decisions like determining the functional unit have a great impact on the results (Jürgens & Endres, 2024). In the case of reuse systems, the results are also dependent on the system boundary, whether primary or secondary data is used, and which assumptions are made in the beginning of the study (Yadav et al., 2024). The results of the LCA of the reuse system heavily depend on the assumptions made on the number of reuses of the tableware as well as the washing method chosen by the service provider (Genovesi et al., 2022). In addition, the location of the production and the electricity mix used for manufacturing, EoL scenarios and user behaviour all influence the environmental impact of the reuse system (Lewis et al., 2021).

2.2.4 Use stage

As mentioned previously, the most impactful stage for the reuse system is the use phase, which is in turn affected by the design of the return system. Due to the washing process, the reuse system can use up to 99% more water and 62% more energy than the single-use counterpart (Aggarwal, 2024). Another important factor determining the impacts of the use stage is the transport (Campbell et al., 2020; Yadav et al., 2024).

Washing

Several studies have demonstrated that more than a third of the environmental impact of reuse systems is caused by energy consumption during the washing in the use stage (Cottafava et al., 2021; Walker et al., 2024). The environmental metrics during the washing phase are strongly influenced by how the tableware is washed. For instance, if the reusable containers are handwashed, the impact depends on the user preferences: whether the water used during handwashing is hot or cold or if paper towels are used for drying (Potting & van der Harst, 2015). However, it has been shown, that washing reusable tableware in a dishwasher clearly reduces the environmental impacts compared to handwashing (Potting & van der Harst, 2015; Cottafava et al., 2021). Meta-analysis conducted by Lewis et al. (2021) also concludes that the most efficient choice in the washing phase is to use dishwashers, which is followed by handwashing in cold water and finally, the least efficient option is to handwash in hot water. Furthermore, the type of dishwasher also plays a role. Compared to residential dishwashers industrial washing can reduce the GHG emissions by 92% (Yadav et al., 2024).

In the case of takeaway containers, an additional unforeseen step might be added by some consumers, who wash the container at home before bringing it back to a collection point. This, however, increases the electricity consumption during the use stage. The impact of this excessive washing at home can increase the primary energy demand such that it surpasses that of most single-use containers (Hitt et al., 2023). From here follows that customers should be informed about best practices in the use phase to achieve the desired customer behaviour when buying food or drinks in reusable containers.

Moreover, it has been reported that 10% of the impacts in the reuse system arise from the use of detergents during washing (Walker et al., 2024). The effects of the detergent have been shown to increase the impacts of eutrophication and ozone layer depletion. These impacts are related to the fertilizers required for cultivating palm and coconut from which the surfactants in detergents are derived. A solution would be to use eco-friendly alternatives that do not contain palm oil (Miele et al., 2024).

The differences in electricity mix in different countries also have an important effect on the environmental metrics during the washing phase. In countries where renewable energy forms a large part of the grid mix, the environmental impacts caused in the washing stage are reduced. (Aggarwal, 2024; Lewis et al., 2021; Pladerer et al., 2008)

Transportation

Depending on the reuse system transportation can either be a) from tableware rental place to the venue where they are used to the centralised washing facility directly (Campbell et al., 2020) or b) from the restaurant to the place where the food is consumed and from there either back to the restaurant (Castellani & Cardamone, 2022) or c) to designated containers around the municipality, from where these are collected and transported to the central washing facility (Yadav et al., 2024). In the first and third scenario, the transport is organised centrally by the tableware or container provider by trucks or vans. In the second, decentralised, scenario the customer must return the container to the restaurant, where it is washed.

For the first scenario the GHG emissions from transport across 77 km to the central washing facility range from 15% to 25% of the whole life cycle (Campbell et al., 2020). Cottafava et al. (2020) demonstrated that the distance to the central washing facility must not exceed 50 km to minimize the environmental impact compared to single-use containers.

In the case of b) if 5% of customers drive their cars only to return used containers, the reuse system results in higher GHG emissions and primary energy consumption compared to the single-use system. Interestingly, the GWP would increase by 162% even if the customer drives an electric car (Hitt et al., 2024). This is because, when the customer travels to the restaurant only to return the empty container, then no emissions can be allocated to e.g. buying a new meal or doing other errands (Yadav et al., 2024).

In the case of c) the reuse system shares common containers across several restaurants and the used empty containers can be returned to various locations around the municipality for subsequent collection and central washing. This decreases the chance that the customer only travels to return the container and the customer’s transport can be omitted (Yadav et al., 2024). A study by Hitt et al. (2025) found that in this scenario, the transport for decentralised collection by trucks forms 58% of the GHG impacts during the use stage, with washing contributing 42%. However, it is important to note that in this study the production of trucks and collection bins were also accounted for (Hitt et al., 2025).

2.2.5 Number of uses

The effectiveness of the reuse system is highly influenced by the number of times the tableware is used (Genovesi et al., 2022; Vercalsteren et al., 2006). The breakeven point sets the theoretical value that should be exceeded to have the reuse system perform better than the single-use system. If the number of uses is limited by the producer at a lower value than the breakeven point, then the reuse system will always have larger environmental impacts than the single-use system (Baird et al., 2022). In the case of reusable plastic tableware many producers estimate that their products can be used ca 100-150 times (Pladerer et al., 2008).

Nevertheless, numerous other aspects can influence the number of uses. Reusable tableware can be lost from the reuse system when the user does not return it to the collection point. The loss rate can range from 2% to 20% depending on the return system implemented (Campbell et al., 2020). When reusable cups are branded with the event logo or other design, the return rate is lower, because customers might take the cup home as a souvenir. In that case, the loss rate can even exceed 25% (Pladerer et al., 2008). Although, one can assume that the cup is used at home, the cup is still lost from the system, thereby increasing the environmental impact of the system, as new cups need to be produced (Campbell et al., 2020; Pladerer et al., 2008). From here it follows that any reuse system should aim to achieve as high a return rate as possible (Campbell et al., 2020).

A few studies have also highlighted the importance of structural integrity of the reusable tableware and appearance for the customer, as signs of wear and tear may lead to the customer rejecting the reusable container (Aggarwal, 2024; Baird et al., 2022). This means that the container must maintain its appearance well over the calculated breakeven point to demonstrate a smaller environmental impact than the single-use system (Greenwood et al., 2021).

2.2.6 End of Life

The EoL stage of the single-use systems can form up to 25% of the whole life cycle impacts. In the reusable system the EoL contributes little to the impacts (Lewis et al., 2021). Still, it is an important stage where assumptions made can affect the results of the LCA. Recycling technologies encompass mechanical recycling, chemical recycling, and solvent-based recycling. Closed-loop mechanical recycling or solvent-based recycling show the best results in terms of material quality. Nevertheless open-loop mechanical recycling, which results in a reduction in quality forms the majority of post-consumer plastic recycling (Jürgens & Endres, 2024). Many studies demonstrate that recycling disposable containers from most common materials (PP, PLA, PET) is preferred over incineration for energy recovery (Cottafava et al., 2021; Potting & van der Harst, 2015). In fact, mechanical recycling produces around 60% fewer emissions per ton compared to incineration (United Nations Environment Programme, 2023).

It has been shown that recycling single-use cups is more effective than reusing cups that are eventually incinerated, regardless of the number of uses (Cottafava et al., 2021). However, this is also dependent on the energy mix of the country. In case of incineration, credits for energy recovery can significantly impact the results of the LCA.

When waste is incinerated with energy recovery in a country with fossil-based energy production, it contributes to reducing greenhouse gas emissions in electricity production, as it substitutes some of the electricity generated by burning fossil fuels (Potting & van der Harst, 2015). However, as material recovery is higher in the waste hierarchy than energy recovery, this outcome should be taken with caution (Jürgens & Endres, 2024). For instance, in a country where the energy mix has a higher proportion of renewable energy, the credits derived from incineration compared to recycling are lower, which aligns with the waste hierarchy (Aggarwal, 2024).

When it comes to biodegradable plastics, composting and digestion yield poorer results compared to other EoL options, particularly incineration, primarily due to high methane emissions and the absence of credits for material or energy recovery (Cottafava et al., 2021; Potting & van der Harst, 2015). Also, as biobased plastic does not contain nutrients then there are no useful products generated during composting and no ecological benefits (Pladerer et al., 2008; Potting & van der Harst, 2015). Furthermore, emissions from composting and anaerobic digestion contribute to increased impacts in acidification category (Genovesi et al., 2022). In some cases, the composting process increases the environmental impact of the EoL stage to 55-80% (Goodrum et al., 2024). Interestingly, these findings are not in line with the waste hierarchy, whereby material recovery, including composting, is preferred over energy recovery (Walker et al., 2024). Anaerobic digestion however has shown better results than incineration (Potting & van der Harst, 2015).

Another important aspect to consider is that bioplastics are in practice often not composted, but rather incinerated, because they cannot be separated from the fossil-based plastic in the organic waste fraction (Utilitalia, 2020). In addition, even if bioplastics are labelled as compostable under EN 13432 (European Committee for Standardization, 2007), which mandates disintegration within 12 weeks and complete biodegradation within 6 months, this is not always achievable with the current industrial composting practices in some European countries (Pladerer et al., 2008; Utilitalia 2020).

Furthermore, plastics labelled 'compostable' may not be suitable for home composting, potentially misleading consumers. This has led to some countries, like Belgium, to prohibit the label ‘biodegradable’ from packaging, because it might inadvertently convey to consumers that the product can be discarded anywhere in nature and strangely encourage littering (van Doorsselaer & Koopmans, 2021).

3. LIFE CYCLE ASSESSMENT

3.1 Methodology

Life Cycle Assessment (LCA) is a comprehensive and standardized method used to evaluate the environmental impact of a product or service throughout its entire life cycle. It ensures that improvements in one area do not simply shift the environmental burden elsewhere. By quantifying emissions, resource consumption, and related impacts, LCA provides crucial insights for decision-making to foster sustainability. As such the LCA is a useful ecodesign tool that helps determine the life cycle stages of a product, which need to be improved to reduce the product's environmental impacts. (Jolliet et al., 2016)

The two standards followed in this study are the ISO 14040/14044:2006, which set the principles and framework for the LCA. The LCA consists of the following stages: goal and scope definition, life cycle inventory analysis (LCI), life cycle impact assessment (LCIA), and life cycle interpretation of results (Figure 3.1). (ISO, 2006)

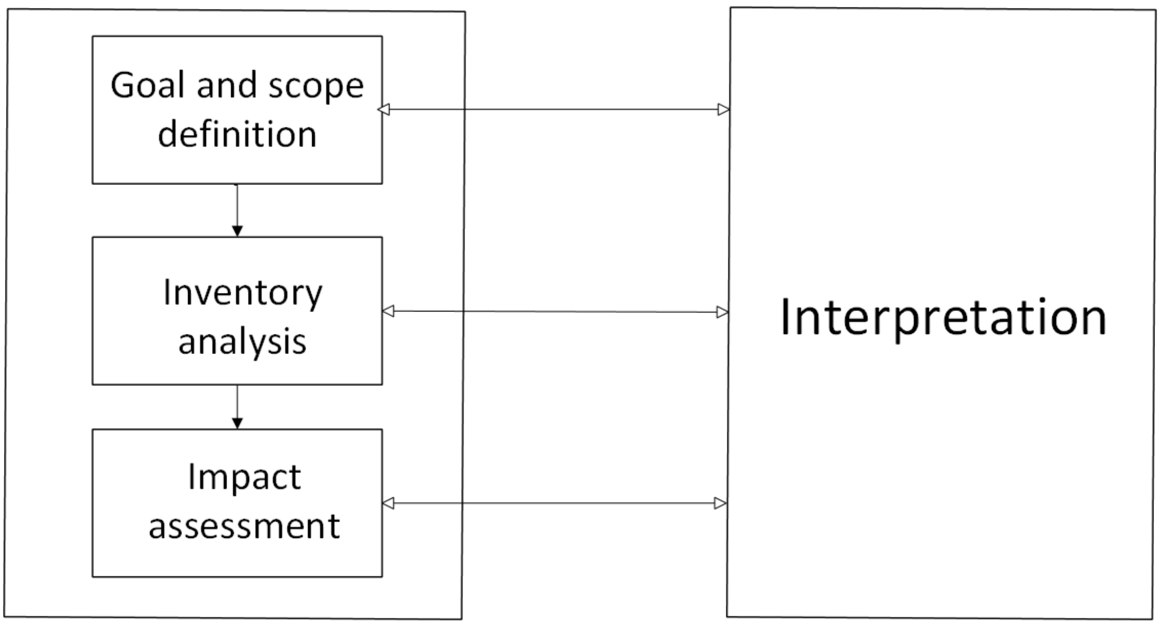

Figure 3.1 The four phases of the life cycle assessment (ISO, 2006)

In the goal and scope definition it is clarified why an LCA is conducted, what are the different scenarios studied and who is the intended audience. In addition, the functional unit and the system boundary are defined. In this phase also assumptions and limitations are disclosed and the methodology for the life cycle impact assessment clarified.

In the life cycle inventory analysis data on all the flows of energy, material and other resources are collected and quantitatively described throughout the different stages of the life cycle. The amounts of the various flows are multiplied by the reference flows to correspond to the functional unit.

In the life cycle impact assessment, the inventory data are linked to impact categories to quantify the environmental burden. For the calculations in this LCIA the Environmental Footprint (EF) v3.1 method is used because the data is high-quality, and it is endorsed by the European Commission (Regulation 2024/1781). In the EF v3.1 impact assessment method there are 16 impact categories: particulate matter, human toxicity (cancer), human toxicity (non-cancer), climate change, ozone depletion, ionising radiation, photochemical ozone formation, acidification, eutrophication freshwater, eutrophication marine, eutrophication terrestrial, ecotoxicity freshwater, land use, water use, fossil resource use (energy carriers) and resource use of minerals and metals (European Commission, 2021).

The life cycle interpretation phase aims to identify the life cycle stages which have the highest environmental burden and where changes can result in lowered impacts. In this phase recommendations for product improvement can be made based on insights gathered from the previous steps in the LCA.

3.2 Study

3.2.1 Goal and scope definition

This LCA aims to evaluate the environmental impacts of existing reuse systems for tableware that are used in public events in Estonia. The tableware analysed in this study are a 400 ml drink cup and a main plate. The drink cup is used to serve cold beverages. The main plate here means a flat dish that is meant to hold warm or cold solid food. Photos of samples of the tableware are exhibited in Appendices 1 and 2.

Although SUP tableware is not allowed during public events in Estonia, in order to better understand the environmental impacts of the reuse systems and juxtapose it with the alternative, the environmental performance of the reuse system will be compared to that of the single-use system.

The goal of the study is to compare the environmental performance of the single-use system and the reuse system of tableware of three different service providers in Estonia (Company A, B and C). The study could be of interest to the three existing service providers of the reuse system in Estonia, who, by referring to the results of this study can use ecodesign tools to improve the environmental performance of their service. The study can also be of interest to other companies looking to develop their business model in the field of event management or takeaway food business. Finally, the study can provide policymakers with the knowledge on designing new policies to encourage reuse systems.

Reuse system

During public events, food and drinks are often served by different caterers. In principle there are two possible ways to serve food and drinks to customers at an event: using SUP tableware or renting reusable tableware. Figure 3.2 illustrates a typical reuse system with a deposit system at a public event in Estonia. In the reuse system the service provider purchases reusable plastic tableware, which is shipped to the company's warehouse. Once an event organiser rents reusable tableware from the company, the tableware is delivered from the warehouse to the venue, where it is used to serve food and drinks by the catering companies at the event. In most public events, a deposit system is applied for reusable tableware, whereby the customer pays a small deposit for the cups and plates, which they will get back, once they have returned the tableware to the designated collection points on the event grounds. In some cases, the collection points are managed by the service provider of the reuse system. In other cases, they are staffed by volunteers of the event. At the end of the event, the collected returned tableware is transported back to the warehouse where it is washed and dried in a dishwashing machine and stored until the next event, when the tableware will be used again. In the reuse system under study no additional rinsing or washing is done at the event grounds. The reusable cups and plates can be used ca 100-125 times, as specified by the companies.

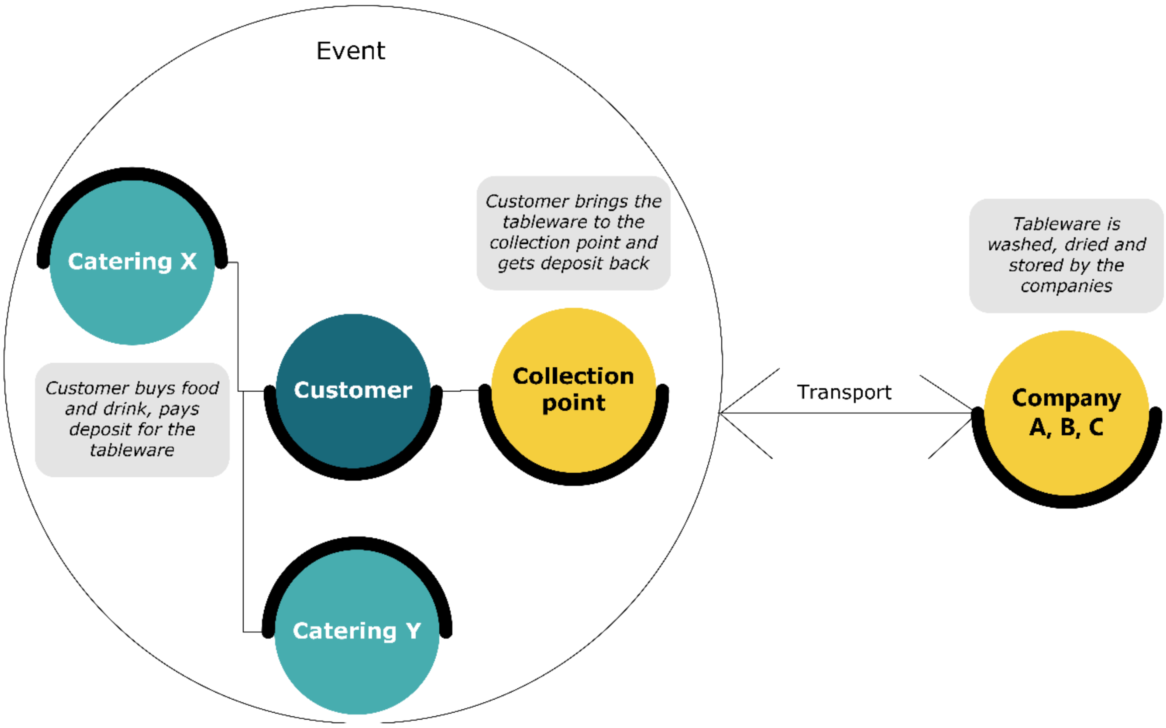

Figure 3.2 Illustration of a reuse system at a public event

Single-use system

The single-use system follows the conventional linear consumption pattern of using SUP tableware once, after which it is discarded. In this study, the single-use cups are produced from PP and the disposable plates from paperboard, as these have been the most common disposable tableware used during events. The disposable tableware is produced at a manufactory, where it is packaged and distributed to a wholesaler and from there it is delivered to the catering companies at the event, who use the tableware to serve food and drinks to eventgoers. Once the customer has finished their meal or drink, the tableware is collected in a waste container at the event grounds. After the event the waste is sent to a waste treatment facility. As the SUP tableware is contaminated with food it is not recycled and will be incinerated in a waste-to-energy plant.

In this study biodegradable disposable tableware was not considered, because there is no technology in Estonia for composting bioplastic. In some countries, like Italy, the SUP directive has been transposed into national law by still allowing the use of disposable tableware made of bioplastics (Walker et al., 2024). In Estonia, the

Packaging Act explicitly mentions that the ban on disposable tableware at public events also applies to bio-based plastics and therefore is not a suitable alternative to SUP tableware used at public events.

The functional unit (FU) is defined as 1000 meals served in 25 events. For one meal a set of tableware consisting of a cup and a plate is needed. For every meal a new set of SUP tableware is used. To fulfil the FU description 25 000 sets of SUP tableware are needed. Taking into account a 5% loss rate, 2250 sets of tableware are needed in the reuse system.

The system boundary of this LCA is "cradle-to-grave" and is shown in more detail in Figure 3.3 for the reuse system and 3.4 for the single-use system. The analysis includes environmental impacts from material extraction, manufacturing, distribution, use and EoL of the tableware.

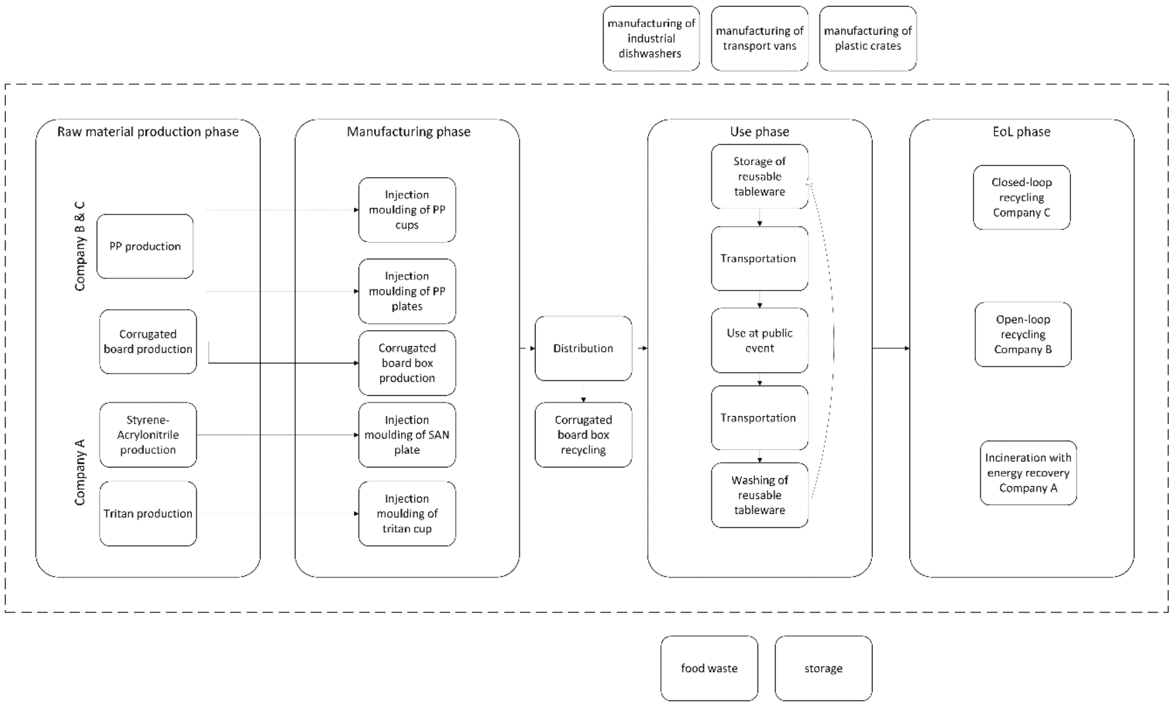

Figure 3.3 System boundary for the reuse system of three companies (A, B and C)

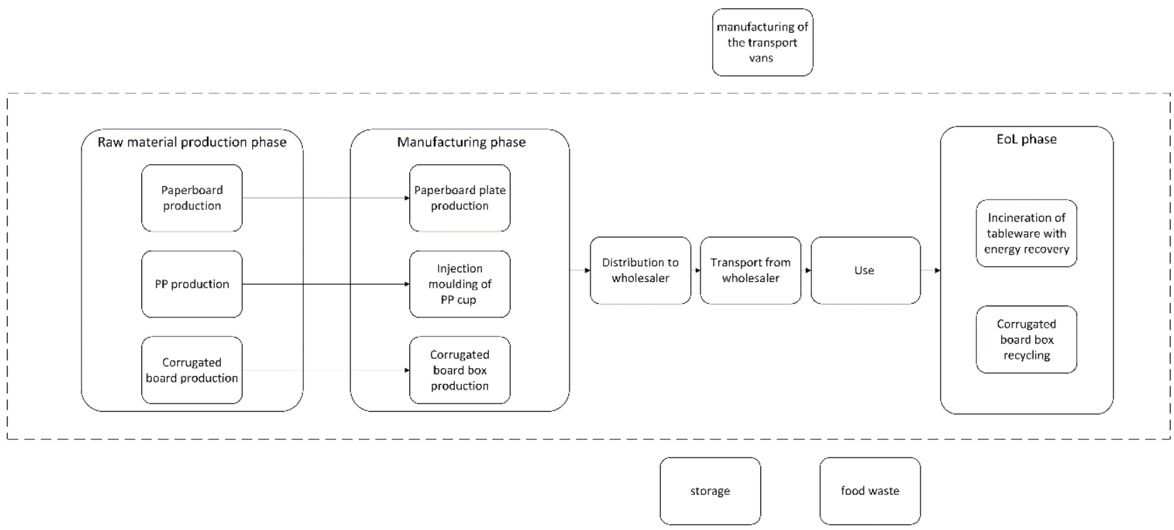

Figure 3.4 System boundary for the single-use system

The following limitations are considered in the study:

- Although food waste has a significant impact on greenhouse gas emissions, food waste is not considered in this study, because it is assumed that the amount of food waste for either system is similar.
- For the reuse system, the manufacturing of the industrial dishwashers is not taken into account because their lifetime is at least 10 years and therefore has minimal impacts to the functional unit (Walker et al., 2024).
- For the reuse system, the manufacturing of the vehicles transporting the tableware to and from the warehouse is not considered because their lifetime is relatively long with minimal impacts to the system.
- Storage of the cups is excluded from the system boundary. It is assumed that the storage of SUP and reusable tableware are similar.
- For the delivery of reusable tableware to events plastic crates are used in all three companies. Due to the longer lifespan and high number of reuses in the reuse system, the impact of the crates is omitted.

3.2.2 Life Cycle Inventory

Data collection

Data for the reuse systems were gathered from the three companies (A, B and C) providing the tableware rental service in Estonia. Samples of chosen cups and plates were weighed at the warehouses of the companies. Semi-structured interviews with the companies' employees were also conducted to gather information about the reuse system, loss rate, the process of washing, typical number of uses per tableware etc. Data for the single-use system were collected from the wholesalers' and producers' webpages and literature. Samples of SUP cup and plate were also weighed. For any additional data, the globally recognized Ecoinvent (v3.11) life cycle database was utilized.

Table 3.1 The weights and materials of reusable and SUP tableware

		CUP		**PLATE**	
		Per item	Reference flow	Per item	Reference flow
COMPANY A	Weight, gr	37	81 400	121	266 200
	Material	tritane		SAN	
COMPANY B	Weight, gr	38	83 600	76	167 200
	Material	PP		PP	
COMPANY C	Weight, gr	47	103 400	79	173 800
	Material	PP		PP	
SINGLE-USE	Weight, gr	6	150 000	21	525 000
	Material	PP		paperboard	

Production

Tableware provided by companies B and C are produced from PP granulates in various locations in Italy and France. The PP cups and plates are injection moulded. In the case of company A the cup is produced from tritan in Estonia and the plate from styrene acrylonitrile (SAN) in Sweden. Both the cup and plate are injection moulded.

For the single-use system the paperboard plate is produced in Latvia. The disposable PP cup is produced in Poland by thermoforming. Data about the production of raw materials and manufacturing of the tableware are taken from Ecoinvent database. Table 3.1 presents the weights and materials of the tableware analysed in this study. In addition, the table shows the reference flows of each cup and plate for the baseline scenario.

Distribution

The reusable tableware is transported from the manufacturers to the companies A, B and C by lorries. In the single-use system the SUP tableware is transported first to a wholesaler and from there to the event. Transport distances were calculated in Google Maps (Table 3.2). For both systems corrugated cardboard boxes with the dimensions 600x400x400mm are used for the distribution of the tableware. In the reuse system the box is discarded upon arrival to the company's warehouse. In the single-use system it is used to transport the tableware to the event and is discarded together with the used tableware at the event. Data for the impact of transport and packaging were taken from Ecoinvent database.

Table 3.2 Data for transport of tableware from manufacturer to companies

		Distance, km	**Means of transport**	**Type of vehicle**
Company A	cup	25	lorry	transport, freight, lorry, 7.5-16 metric ton, diesel, EURO 6 | transport, freight, lorry, 7.5-16 metric ton, diesel, EURO 6 | Cutoff, Europe
	plate	588	lorry	transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | Cutoff, Europe
		290	ferry	transport, freight, sea, ferry, heavy fuel oil | transport, freight, sea, ferry, heavy fuel oil | Cutoff, U - Glo
Company B	cup	2461	lorry	transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | Cutoff, Europe
	plate	2200	lorry	transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | Cutoff, Europe

		Distance, km	Means of transport	Type of vehicle
Company C	cup	2600	lorry	transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | Cutoff, Europe
	plate	2600	lorry	transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | Cutoff, Europe
SUP	cup	1545	lorry	transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | Cutoff, Europe
	plate	320	lorry	transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | transport, freight, lorry, 16-32 metric ton, diesel, EURO 6 | Cutoff, Europe

Use stage

The reuse system for the three companies is similar. The tableware is transported to the public events, which all take place in the city centre of Tallinn. A deposit system is implemented, to ensure the highest possible return rate. After the event the used tableware is transported to the companies' washing facility where it is cleaned before the next use. The washing machines in all companies utilise the available washing capacity fully every time.

Data on electricity, water and detergent consumption in companies A and B were derived from the websites of the dishwashing machines' manufacturers and from literature (Castellani & Cardamone, 2022; Meiko, n.d.; Metos, n.d.). Company C provided the exact data. According to Company C 75% of the energy used during the summer, the most active event season, comes from their solar panels. This is also accounted for.

All companies use Euro 6 compliant vans to transport tableware to and from the events. Distances between the events in the city centre and the companies' washing facilities were calculated in Google Maps and are presented in Table 3.5.

Based on interviews with the three service providers in Estonia, the loss percentage of reusable tableware at events with a deposit system is around 5%. This has been taken into account in the reference flows of the reuse system.

In the single-use system no impacts occur during the use stage. The environmental impacts of a scenario where SUP tableware causes littering during the use phase cannot be measured by current LCA methods.

End of Life stage

The three reuse systems analysed in this study have different EoL scenarios. Company A does not have a determined procedure for the end-of-life stage of the tableware. This is because the tableware has not been used for a long time (approximately 1 year) and there has been no reason yet to dispose of the tableware due to wearing. Therefore, the individual items of tableware removed from use in Company A are disposed of in mixed municipal waste. As most of the mixed municipal waste in Estonia is incinerated (Jäätmete infopäring, n.d.), the EoL of the plates in Company A is incineration with energy recovery. Company B has an arrangement to send tableware that is removed from use to PP pellet manufacturers, which means that 100% is recycled. Company C sends its decommissioned tableware back to the producer for closed-loop recycling. The transport in the EoL stage for all companies is considered.

Similarly to Company A, also in the single use system, the EoL of the tableware is incineration with energy recovery as the used SUP tableware is discarded in mixed municipal waste.

Assumptions

- Although the diameters of the plates provided by the companies vary slightly (+/- 1 cm), the plates are used to serve the same amount of food, thus the FU is the same.

- Also, the cups vary in their height and diameter, but their volume is the same, 400 ml, therefore the FU is the same.

- Raw materials like tritan, PP, SAN and paperboard are manufactured in the same factory as the production of the tableware and no transportation of raw materials is considered during the production stage.

- Since manufacturing of tritan as a raw material is not an available flow in the Ecoinvent v3.11, manufacturing of PET granulates, bottle grade, was used in the tritan cup production stage instead, as tritan is derived from PET (Wankai New Materials Co.Ltd., 2024).

- For the packaging in the distribution stage of both systems a similar corrugated board box with the dimensions 600x400x400mm was selected. In the EoL stage the box is recycled.

- All 25 events of the FU in the baseline scenario take place in the city centre of Tallinn, to not give preference to the location of any of the companies.

- For the closed-loop recycling of the tableware used by Company C it is assumed that the tableware is manufactured 50% from recycled PP pellets and 50% from virgin material.

3.2.3 Sensitivity analyses

To assess whether some data and modelling assumptions influence the results of the LCA five sensitivity analyses are performed. In the first scenario the number of events is increased from 25 to 50. In the second scenario the number of events is increased to 100. Both the 1st and 2nd scenario increase the FU for both systems (Table 3.3, Table 3.4). In the third scenario, the location of the events is changed to the city centre of Tartu, a town located 180 km from Tallinn. In this scenario the distance between the event grounds and the off-site washing facility is increased ca. 10 times (Table 3.5). In the fourth scenario the loss rate in the reuse system is increased from 5% to 10%. In the fifth scenario the loss rate in the reuse system was reduced to 0.

Table 3.3 The reference flows of plastic cups in the different scenarios analysed (weight in kg)

CUP	UNIT	BASELINE	SC 1	SC 2	SC 3	SC 4	SC 5
COMPANY A	0,037	81,4	127,65	222,15	81,4	125,8	37
COMPANY B	0,038	83,6	131,1	226,1	83,6	129,2	38
COMPANY C	0,047	103,4	162,15	279,65	103,4	159,8	47
SINGLE-USE	0,006	150	300	600	150	150	150

Table 3.4 The reference flows of plates in the different scenarios analysed (weight in kg)

PLATE	UNIT	BASELINE	SC 1	SC 2	SC 3	SC 4	SC 5
COMPANY A	0,121	266,2	417,45	719,95	266,2	411,4	121
COMPANY B	0,076	167,2	262,2	452,2	167,2	258,4	76
COMPANY C	0,079	173,8	272,55	470,05	173,8	268,6	79
SINGLE-USE	0,021	525	1050	2100	525	525	525

Table 3.5 The transport distances during the use stage in the different scenarios analysed (km)

	DISTANCE	BASELINE	SC 1	SC 2	SC 3	SC 4	SC 5
COMPANY A	22	550	1100	2200	9650	550	550
COMPANY B	18	450	900	1800	9000	450	450
COMPANY C	18	450	900	1800	8850	450	450

4. RESULTS

4.1. Life cycle impact assessment

In the LCIA the emissions data aggregated during life cycle inventory on material and energy inputs/outputs are first assigned to relevant impact categories during classification. Next, the respective data are multiplied with a characterisation factor to calculate the contribution of inputs/outputs to respective impact categories (European Commission, 2021; Jolliet et al., 2016). In this LCIA the EF v3.1 method is applied, which includes sixteen impact categories, described in detail in Table 4.1. The impact categories were assessed using the OpenLCA software.

Table 4.1 Environmental Footprint impact categories (European Commission, 2021)

EF impact category	Unit	Description
Climate change	kg CO2-Eq	Takes into account all inputs and outputs where GHG are emitted which contribute to the increase of the average global temperatures. All GHG emissions are converted to their equivalent of global warming potential of 1 kg of CO2.
Ozone depletion	kg CFC-11-Eq	Accounts for the depletion of the stratospheric ozone layer by all ozone-depleting substances emitted during the life cycle. These emissions are converted to their equivalent of kilograms of trichlorofluoromethane.
Human toxicity, carcinogenic	CTUh	Refers to potential cancer-related health impacts on human beings caused by absorbing toxic substances from the air, water and soil. The unit of measurement is Comparative Toxic Unit for humans (CTUh).
Human toxicity, non-carcinogenic	CTUh	Refers to potential non-carcinogenic health impacts on human beings caused by absorbing toxic substances from the air, water and soil. The unit of measurement is Comparative Toxic Unit for humans (CTUh).
Particulate matter	disease incidence per kg of PM2.5 emitted	Takes into account the potential adverse impacts on human health caused by emissions of Particulate Matter (PM) and its precursors (e.g. NOx, SO2). It is measured as the change in mortality due to PM emissions.
Ionising radiation	kBq U235-Eq	Accounts for the potential adverse health impacts by radioactivity. It is measured in equivalent of kilobequerels of Uranium 235.
Photochemical ozone formation	kg NMVOC-Eq	Refers to emission of substances (volatile organic compounds and CO) that in the presence of NOx and sunlight cause the formation of surface-level ozone. The measurement unit is equivalent of kilograms of Non-Methane Volatile Organic Compounds.

EF impact category	Unit	Description
Acidification	mol H+-Eq	Takes into account emissions of NOx, NH3 and SOx which contribute to the acidification of soils and water. The emissions are converted to the equivalent of moles of hydron.
Eutrophication, terrestrial	mol N-Eq	Refers to the emission of nutrients (nitrogen and phosphorus) that increase the growth of algae or other plants that inhibit the normal function of an ecosystem. It is measured in equivalent of kilograms of nitrogen (kg N eq).
Eutrophication, freshwater	kg P-Eq	Refers to the emission of nutrients (mainly phosphorus) that increase the growth of algae and other vegetation, which cause low levels of oxygen in water. It is measured in equivalent of kilograms of phosphorus (kg P eq).
Eutrophication, marine	kg N-Eq	Refers to the emission of nutrients (mainly nitrogen) that increase the growth of algae and other vegetation, which lead to blooms of blue-green algae and loss of biodiversity in the marine ecosystems. It is measured in equivalent of kilograms of nitrogen (kg N eq).
Ecotoxicity, freshwater	CTUe	Takes into account the release of substances that have potential toxic impacts on ecosystems, changing the structure and function of the ecosystem. The unit of measurement is Comparative Toxic Unit for ecosystems (CTUe).
Land use	Dimensionless (Pts)	Accounts for the changes of soil quality due to the use and transformation of land for agriculture, roads, housing, mining etc. It is measured in points (Pts).
Water use	m3 world Eq deprived	Refers to the available water in an area after the extraction from lakes, rivers or groundwater for human needs and demand from ecosystems has been met. It is expressed in cubic metres (m3) of water use related to the local scarcity of water.
Resource use, fossils	MJ, net calorific value	Accounts for the amount of materials that contribute to the use of non-renewable resources such as fossil fuels. It is measured in MJ.
Resource use, minerals and metals	kg Sb-Eq	Accounts for the amount of materials that contribute to the use of non-renewable abiotic resources such as minerals and metals. It is measured equivalents of kilograms of antimony.

The most relevant impact categories for this study were identified as those which contribute most to the single scores of the systems. The single score is the sum of the weighted results of all environmental impact categories, indicating the total environmental impact of a system (European Commission, 2021). For the systems under study the most relevant categories are climate change and fossil resource use (Figure 4.1). In addition, acidification and eutrophication were analysed, since in LCA studies where environmental metrics of plastic products are compared most often these 4 aforementioned categories are included (Jürgens & Endres, 2024). The full overview of the results of LCIA on all 16 impact categories in all scenarios is presented in the Appendices (Appendices 3-6). It must also be noted that toxicity categories, like human toxicity (cancerous and non-cancerous) and ecotoxicity of the freshwater have not yet reached a high robustness of the impact model and are not considered conclusive (Antony & Gensch, 2017; Walker et al., 2024).

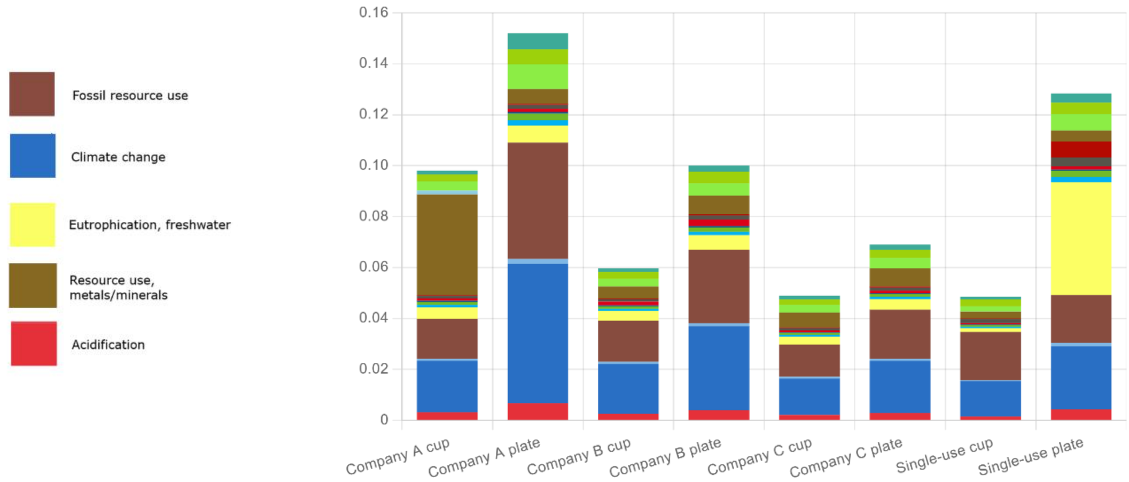

Figure 4.1 Single score of tableware in both systems in baseline scenario

In the baseline scenario cups perform generally better than plates across all 16 impact categories in both systems. From all the tableware items, the plate by Company A demonstrates the largest environmental impacts (Figure 4.1). In fact, the plate by Company A has the highest environmental impact in 10 categories, including acidification, climate change and fossil resource use. The best performing tableware item is the single-use cup, which is indicated by the lowest impacts in 11 categories. The cup by Company C has the lowest environmental burden from the reusable tableware (Table 4.2).

Table 4.2 Results of environmental impact assessment in 16 categories in the baseline scenario

Indicator	Company A cup	Company A plate	Company B cup	Company B plate	Company C cup	Company C plate	Single-use cup	Single-use plate
Acidification	2,83	6,03	2,27	3,61	1,89	2,55	1,33	3,88
Climate change	725,40	1 963,05	704,86	1 181,25	512,38	734,83	497,17	886,80
Climate change: biogenic	0,84	2,15	0,80	1,17	0,86	1,09	1,18	3,32
Climate change: fossil	721,43	1 957,60	700,97	1 176,66	508,75	730,76	495,58	876,74
Climate change: land use and land use change	3,13	3,30	3,09	3,41	2,77	2,97	0,41	6,75
Ecotoxicity: freshwater	2 539,59	5 560,78	2 271,96	3 472,58	2 042,69	2 580,38	1 289,36	4 056,95
Energy resources: non-renewable	12 210,40	35 670,30	12 703,40	22 504,10	9 914,09	15 034,40	14 792,30	14 659,60
Eutrophication: freshwater	0,26	0,39	0,21	0,33	0,18	0,24	0,08	2,54
Eutrophication: marine	0,64	1,39	0,58	0,91	0,45	0,59	0,39	1,36
Eutrophication: terrestrial	5,71	12,33	4,93	7,83	3,88	5,26	3,38	11,66
Human toxicity: carcinogenic	3,33E-07	5,83E-07	2,95E-07	4,35E-07	1,98E-07	2,48E-07	1,07E-07	3,85E-07
Human toxicity: non-carcinogenic	6,93E-06	8,40E-06	1,01E-05	1,81E-05	5,56E-06	7,21E-06	3,76E-06	9,46E-06
Ionising radiation: human health	77,75	129,65	76,27	134,71	68,21	104,21	137,28	288,64
Land use	2 909,73	4 233,43	3 052,32	4 799,50	2 924,62	4 161,70	2 240,16	64 701,00
Material resources: metals/minerals	3,34E-02	4,90E-03	3,98E-03	6,16E-03	4,96E-03	6,00E-03	2,30E-03	3,62E-03
Ozone depletion	1,22E-03	1,85E-05	1,80E-05	3,25E-05	1,72E-05	2,53E-05	2,01E-05	1,97E-05
Particulate matter formation	2,42E-05	6,47E-05	2,04E-05	3,19E-05	1,96E-05	2,71E-05	1,36E-05	4,31E-05
Photochemical oxidant formation: human health	2,29	4,92	2,28	3,96	1,84	2,70	2,30	3,78
Water use	210,92	862,86	187,89	307,76	199,66	276,64	135,54	487,70

Figure 4.2 shows the results of the impact assessment by assigning the item with the largest value in a category the 100% marker and the other items in the analysis are displayed in relation to the highest result. The disposable plate and the plate by Company A exhibit the largest values across categories.

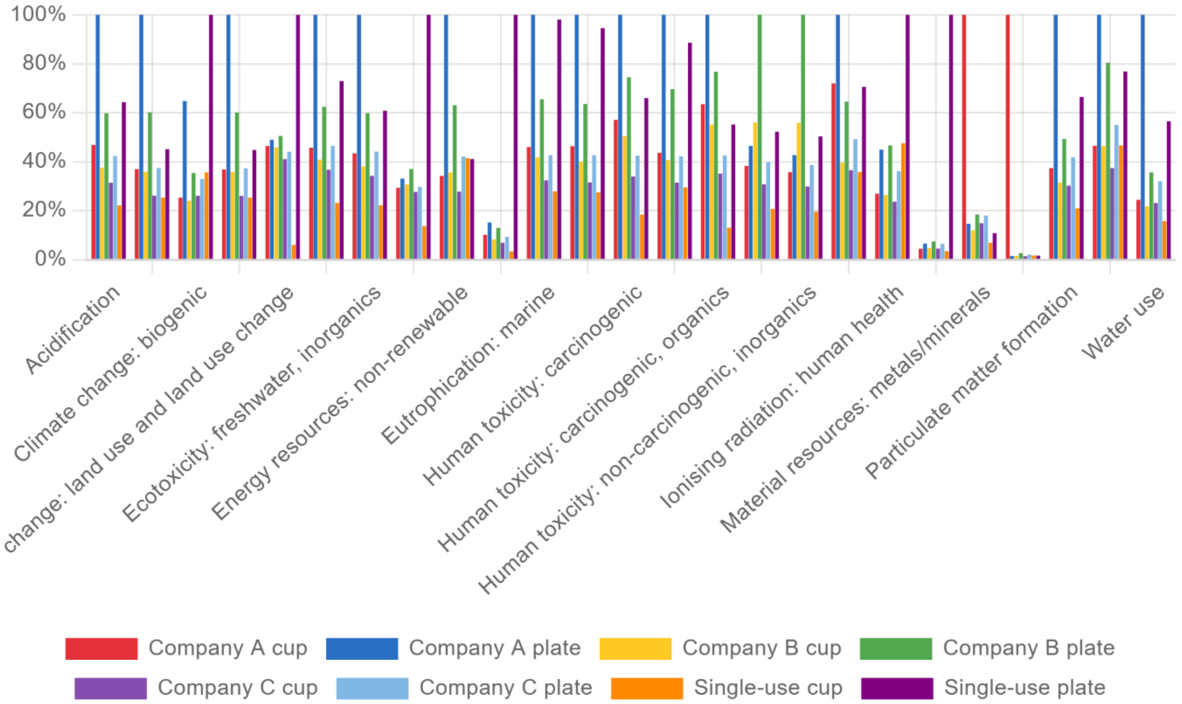

Figure 4.2 Results of the LCIA of the baseline scenario

In the baseline scenario the cups both in the reuse as well as in the single-use system have rather similar environmental impacts in all categories (Table 4.2) with the single-use cup performing slightly better in the climate change (Figure 4.3) and freshwater eutrophication (Figure 4.4) categories. In the fossil resource use category, the single-use cup has a higher impact than the cups in the reuse system (Figure 4.5).

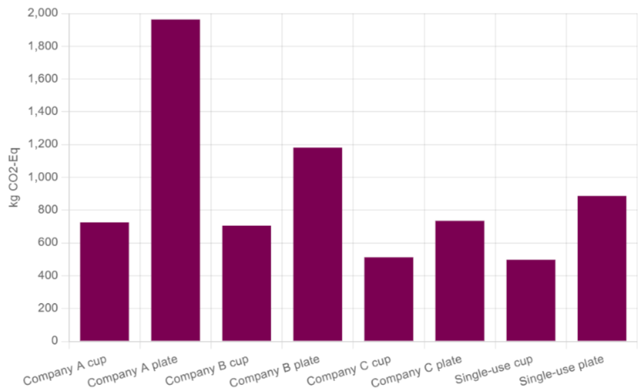

Figure 4.3 Comparison of reuse systems A, B and C with single-use system in climate change impact category for baseline scenario

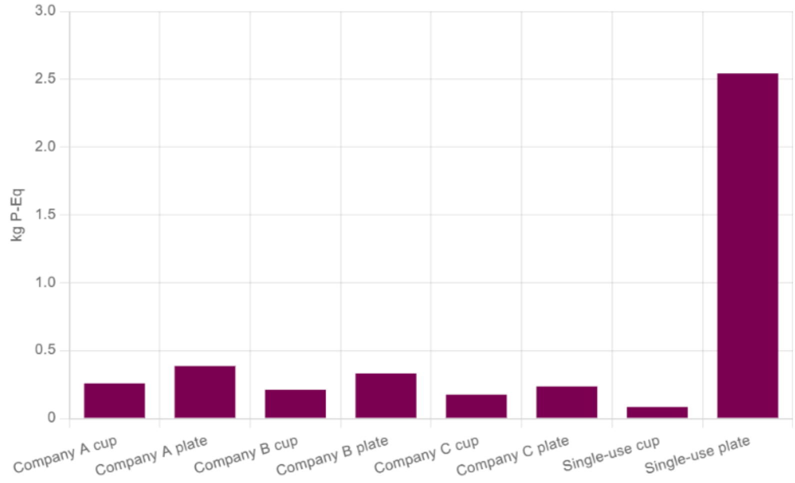

Figure 4.4 Comparison of reuse systems A, B and C with single-use system in freshwater eutrophication impact category for baseline scenario

The best performing plate is by the Company C, which demonstrates the lowest values in almost all impact categories. The environmental burden by tableware in Company B generally falls somewhere between that of Company A and Company C as illustrated on Figure 4.5 and Figure 4.6. In the baseline scenario reuse systems by Company B and C outperform the single-use system, while the reuse system by Company A shows higher environmental impacts than the single-use option.

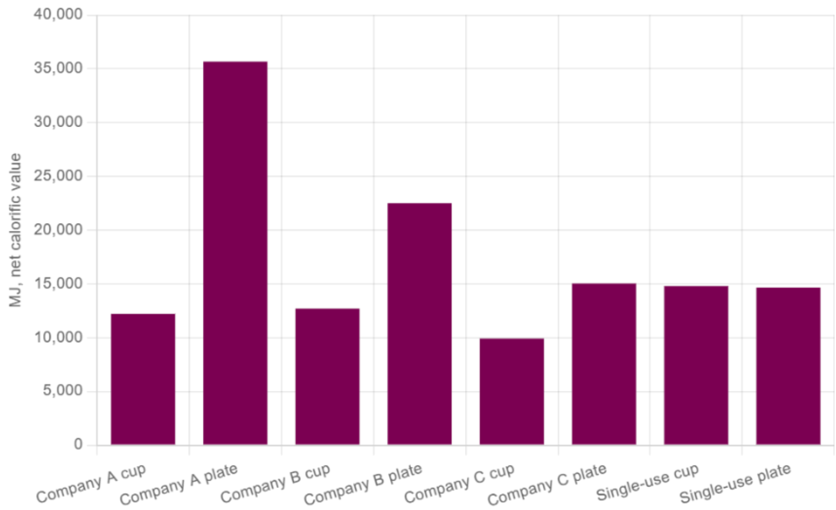

Figure 4.5 Comparison of reuse systems A, B and C with single-use system in fossil resource use category for baseline scenario

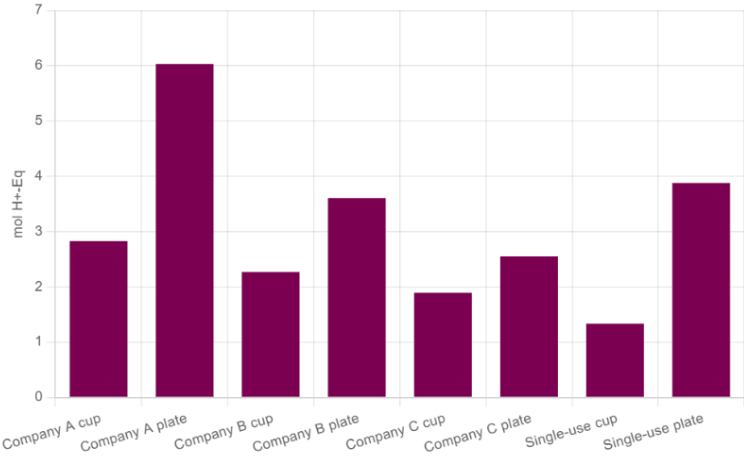

Figure 4.6 Comparison of reuse systems A, B and C with single-use system in acidification impact category for baseline scenario

4.2 Sensitivity analyses

4.2.1 Scenario 1

In scenario 1 the number of the events was increased to 50. This has several implications. In the single-use system more tableware items are needed for catering at the events. For the reuse system more tableware is required due to the loss rate of 5%. In the reuse system the total distance travelled during the use stage also grows by 100%.

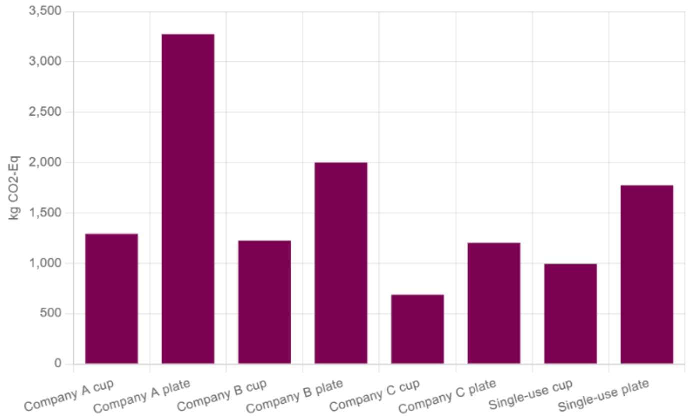

Figure 4.7 Comparison of reuse systems A, B and C with single-use system in climate change impact category for scenario 1

The results for all items across all impact categories have grown. As in the baseline scenario the worst performing tableware item in the climate change category is the reusable plate by Company A which is followed by the plate by Company B and the disposable plate (Figure 4.7). In the acidification category the difference between the impact from the single-use and the plates by Company B and C has increased (Figure 4.8).

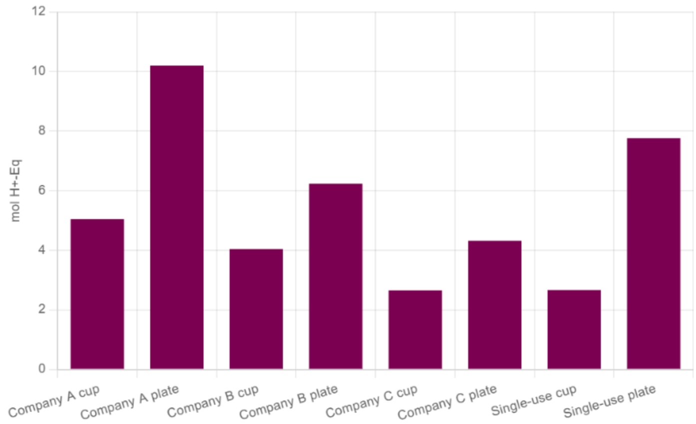

Figure 4.8 Comparison of reuse systems A, B and C with single-use system in acidification impact category for scenario 1

The best performing tableware is the cup from Company C, which in this scenario shows smaller environmental burden than even the single-use cup (Figure 4.9). The single score results demonstrate that the environmental impacts of the disposable plate and the plate by Company A are similar in this scenario, with the reusable plate showing slightly lower impact values. In this scenario the reuse systems by Company B and C still outperform the single-use system. The plate by Company A has a slightly lower environmental burden that the disposable plate, but the cup by Company A has larger environmental impacts than the single-use cup, thus impairing the performance of the reuse system by Company A.

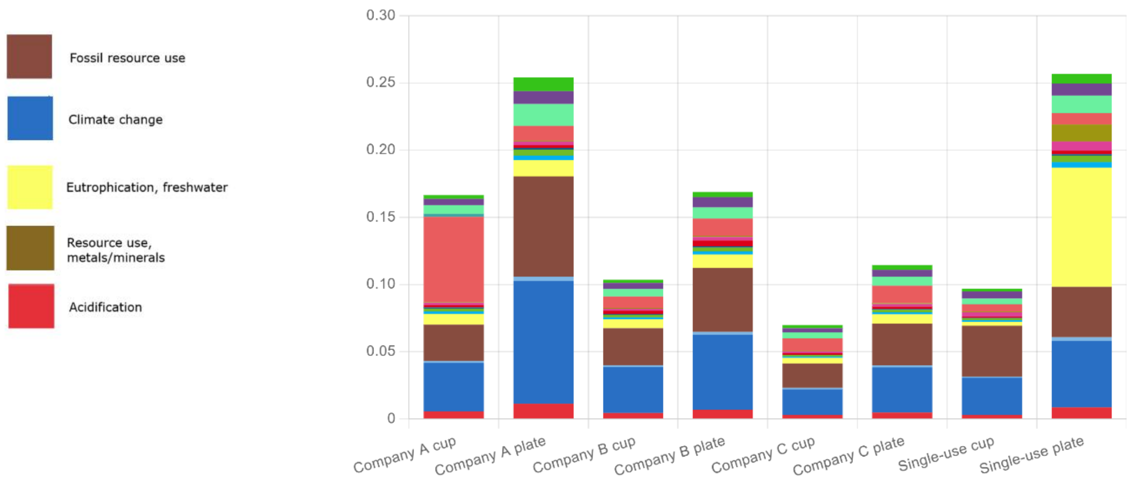

Figure 4.9 Single score of tableware in both systems for scenario 1

4.2.2 Scenario 2

In the 2nd scenario the number of the events was increased to 100. Similarly to scenario 1, the number of tableware items increases for both systems. In the reuse system the distance travelled in the use stage increases as well.

All the reusable cups demonstrate lower CO_2 emissions than the disposable cup in this scenario (Figure 4.10). In the climate change and fossil resource use impact category, the single-use plate has surpassed the plate by Company A and is now the worst performing tableware item (Figure 4.11).

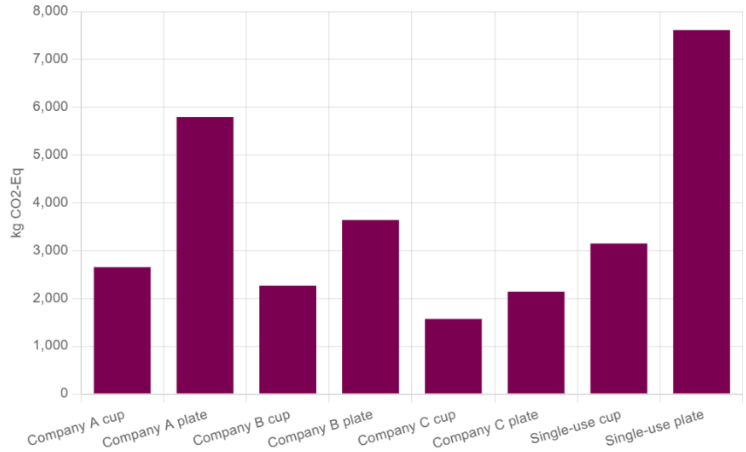

Figure 4.10 Comparison of reuse systems A, B and C with single-use system in climate change impact category for scenario 2

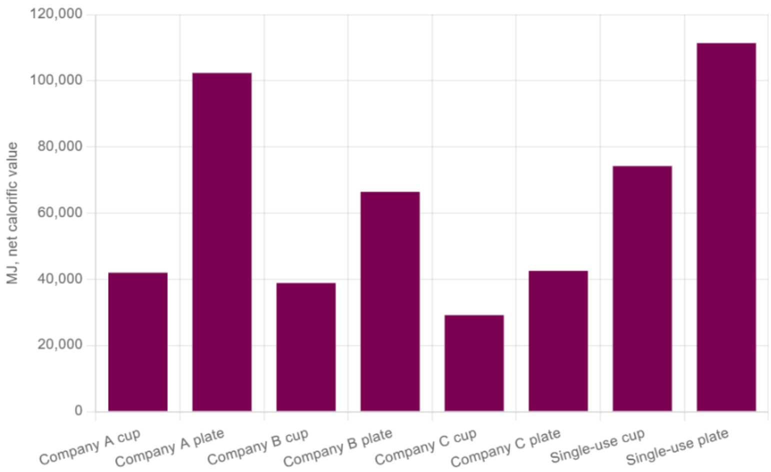

Figure 4.11 Comparison of reuse systems A, B and C with single-use system in fossil resource use impact category for scenario 2

In the acidification and eutrophication categories the differences between the reusable plates and the disposable plate are more pronounced, with the latter showing the biggest environmental burden of all the tableware items (Figure 4.12). In this scenario all the reuse systems perform better than the single-use system.

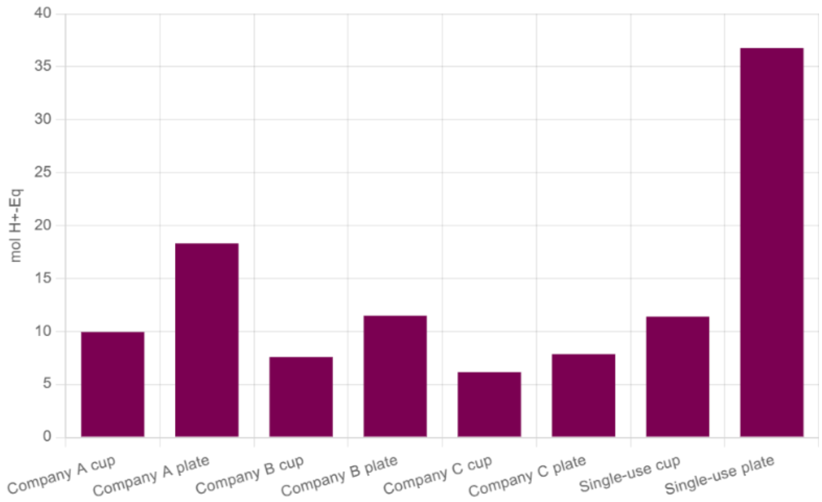

Figure 4.12 Comparison of reuse systems A, B and C with single-use system in acidification impact category for scenario 2

4.2.3 Scenario 3

In the 3rd scenario the location of the event was moved 180 km from Tallinn with the number of the events staying the same (25). This has implications to the use stage in the reuse system, as the transport distance between the washing facility/warehouse and the event grounds is increased more than tenfold. For the single-use system this increases the transport distance in the EoL stage, since the only municipal waste incineration plant is located outside Tallinn. As there are wholesalers for SUP tableware in Tartu and Tartu is also located nearer to the manufacturers of the SUP tableware, the transport distances in the distribution stage for the single-use system are reduced by ca 25%.

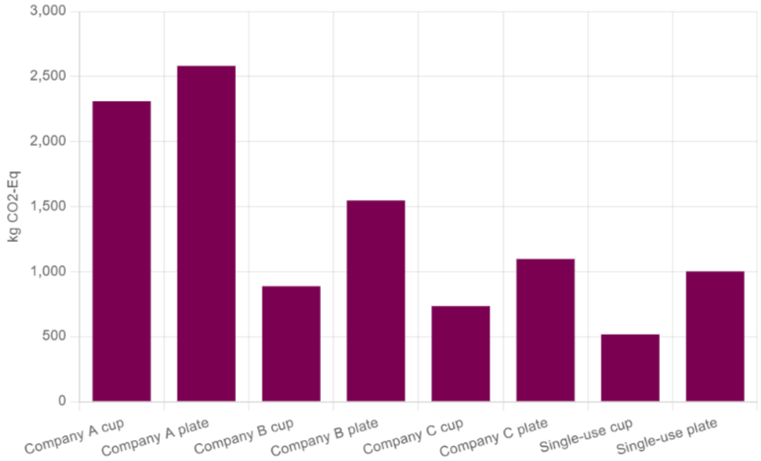

Figure 4.13 Comparison of reuse systems A, B and C with single-use system in climate change impact category for scenario 3

The impacts in the reuse system have increased in all categories by more than 100% compared to the baseline scenario. The environmental impacts in the single-use system also increased, but less than in the reuse system as can be seen in Figure 4.13. In the climate change impact category, the disposable tableware is the best performing system in this scenario. Company C demonstrates the lowest impact in the reuse systems. Although in the freshwater eutrophication impact category the single-use plate still has a significantly larger environmental impact as all the other tableware (Figure 4.14), in this scenario the single-use system is generally outperforming the reuse system.

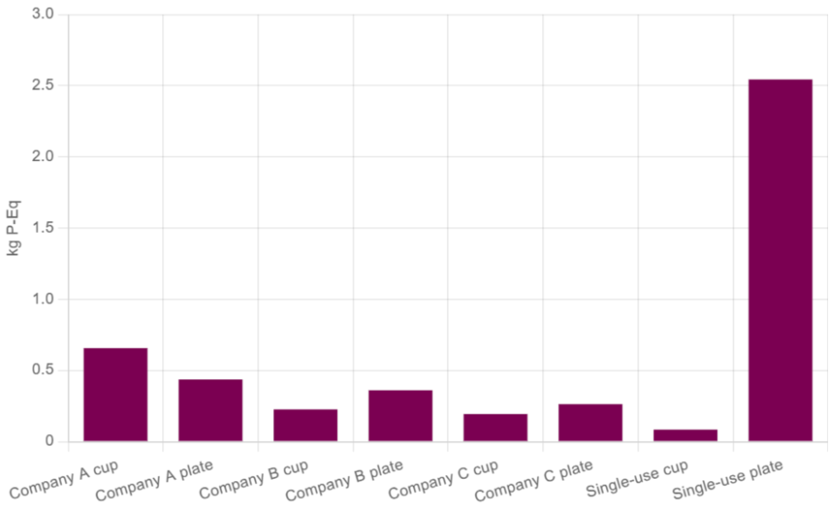

Figure 4.14 Comparison of reuse systems A, B and C with single-use system in freshwater eutrophication impact category for scenario 3

4.2.4 Scenario 4

In the 4th scenario the loss rate of tableware in the reuse system was increased from 5% to 10%. This has implications on the reuse system, as more tableware needs to be produced to serve the meals in the FU. No changes occur in the single-use system.

As can be seen in Figures 4.15 and 4.16 the environmental burden of the reuse system has increased compared to the baseline scenario and in the climate change and fossil resource use impact categories it exceeds that of the single-use system both for cups and plates. The biggest impact in both categories is exerted by the tableware of Company A. Tableware by Company C and the single-use system demonstrate the lowest impacts in all categories, though in this scenario, the single-use tableware performs generally better.

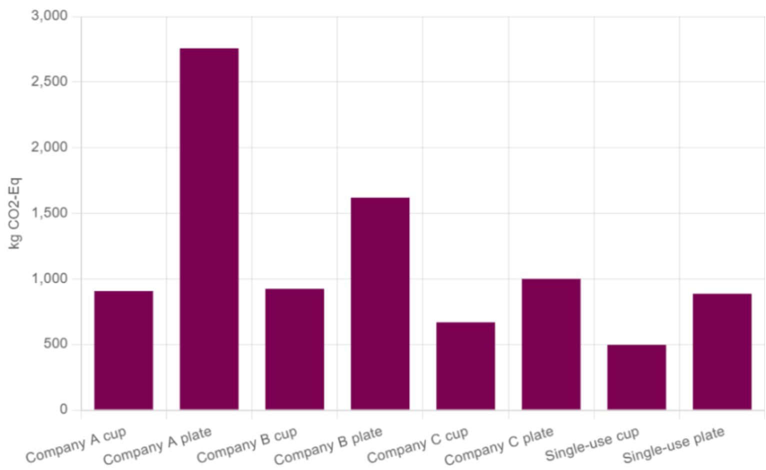

Figure 4.15 Comparison of reuse systems A, B and C with single-use system in climate change impact category for scenario 4

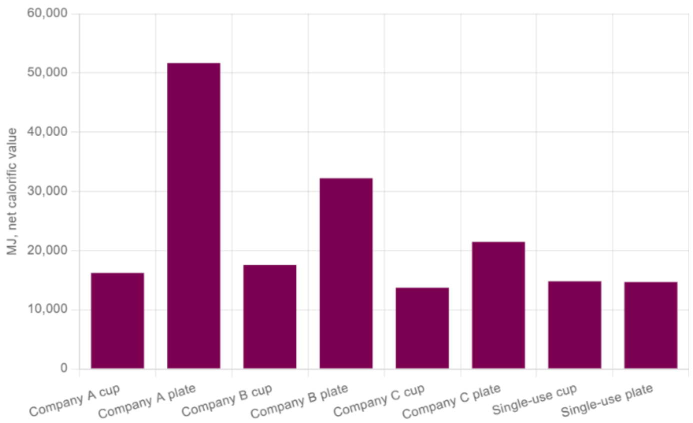

Figure 4.16 Comparison of reuse systems A, B and C with single-use system in fossil resource use impact category for scenario 4

4.2.5 Scenario 5

In the 5th scenario the loss rate of tableware in the reuse system was reduced to 0. This means that in the reuse system no new tableware items need to be produced due to loss from the system. For the single-use system this does not mean any change.

In this scenario the impacts for the reuse systems have decreased compared to the baseline scenario. The reuse systems also fare generally better than the single-use system (Figure 4.17), although in the acidification category the SUP cup has a slightly lower impact than the reusable options (Figure 4.18).

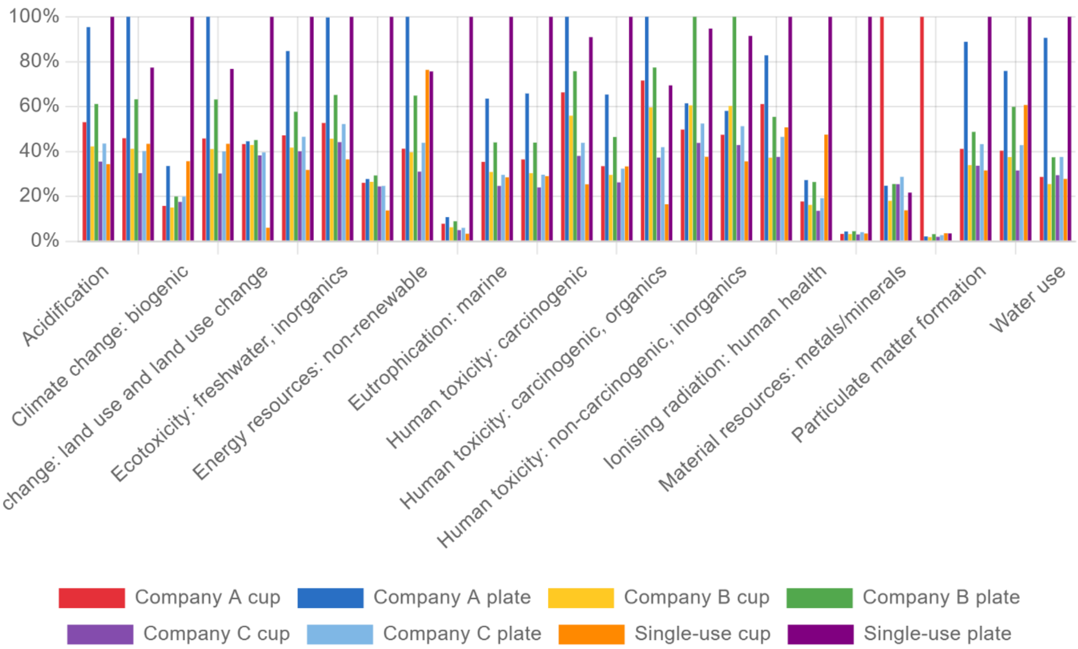

Figure 4.17 Results of the LCIA of the scenario 5

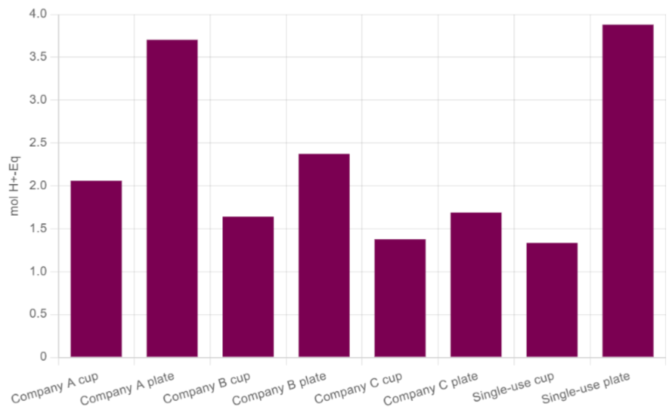

Figure 4.18 Comparison of reuse systems A, B and C with single-use system in acidification impact category for scenario 5

In the climate change and fossil resource use categories, the tableware by company A still has the highest result, but the difference between the single-use system is now smaller than in the baseline scenario (Figure 4.19, Figure 4.20).

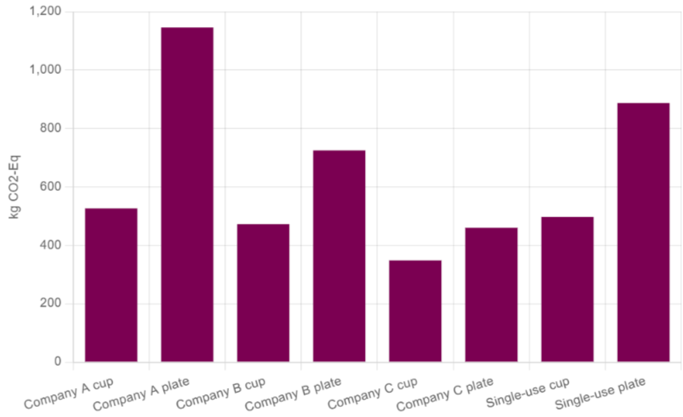

Figure 4.19 Comparison of reuse systems A, B and C with single-use system in climate change impact category for scenario 5

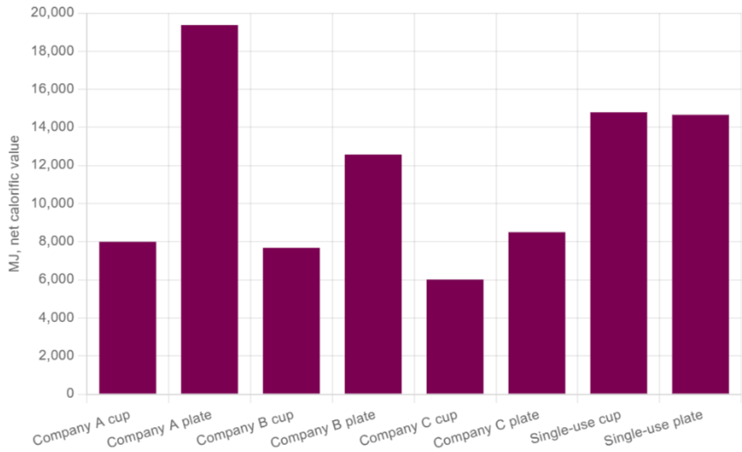

Figure 4.20 Comparison of reuse systems A, B and C with single-use system in fossil resource use impact category for scenario 5

5. DISCUSSION

5.1. Life cycle interpretation

The aim of this study is to compare the environmental performance of the reuse system of tableware from three different service providers in Estonia with a single-use system. The results indicate that with already 25 uses the reuse systems perform better than the single-use system. The exception is the reuse system of Company A, which has a lower environmental burden than single-use system after more than 50 uses. However, if solely the climate change impact category is considered, only the reuse system by Company C achieves a better performance than the single-use alternative with 25 uses. The reuse systems by Companies A and B show lower GHG emissions than the single-use system after 100 uses.

The results of the LCIA clearly demonstrate the effect of the tableware's mass on the exerted environmental impacts. Plates are 2-3x heavier than cups and have a bigger environmental burden in all categories and both systems. Also, the biggest differences in the environmental impact emerge between the plates, which corresponds to the larger range of weights of plates compared to the weights of the cups. The environmental burden of reusable cups is similar across categories. The tableware from Company C consistently outperforms other reuse systems as well as the single-use system in most categories in the baseline and 1st and 5th scenario. The plate by Company A and the single-use plate demonstrate the largest impacts.

It is important to recognise that the results of an LCA always present the environmental performance of a system or a product at a fixed time and location with specific parameters and thus cannot be applied to other similar products or systems (Campbell et al., 2020). Particularly whenever changes are made in the system the LCA should be repeated to ensure accuracy and relevancy for better decision-making. Still, the results of this study could offer important takeaways for Estonian event planners and service providers, as such study in local conditions has not been conducted in Estonia to the author's knowledge.

It is essential to acknowledge that assumptions made before the LCA have a significant impact on the results. For instance, in a study by Potting and van der Harst (2015) it was concluded that the environmental impacts of a reusable and SUP cup are too similar, because both types can be used several times. The assumption that disposable cups can be used several times before throwing them away, which is not common as they are designed for single use only and their aesthetics and structural integrity after one use does not promote their reuse, also influenced the results of the analysis.

In this study a number of assumptions were made due to missing data. The assumption that raw material like PP, SAN, tritan and paperboard are produced in the same location as the manufacturing of the tableware may potentially have an increasing effect on climate change and fossil resource use impact categories due to the additional transportation step in the production stage if the raw material is extracted in another location.

In the case of Company C, the actual rate of recycled material used to produce the tableware is unknown, thus it was assumed that it is 50%. Any change in the amount of recycled PP in the production would influence the results of the LCIA for Company C.

Instead of tritan, PET production was used for the LCIA of the cup by Company A, as tritan production is not an available process flow in the Ecoinvent database v3.11. Tritan is a high-temperature modified version of Polycyclohexanedimethylene Terephthalate Glycol-modified (PCTG) which itself is a derivative from PET (Wankai New Materials Co.Ltd., 2024). Therefore, PET production was chosen as a substitute in the analysis. Consequently, the environmental burden of the cup by Company A might be underestimated because the production process of tritan is probably more energy intense and carbon heavy than that of PET due to incorporation of glycol and heat required during the process.

5.1.1 Most relevant impact categories

Based on the analysis of the single scores for the tableware in the baseline scenario (Figure 4.1), the most relevant impact categories in the systems under study are climate change and fossil resource use. Nevertheless, for the cup by Company A resource use of metals and minerals is the most significant category. This is attributable to the production of antimony, which is used as a catalyst in PET production (Filella, 2020). For the disposable plate, the most significant category is freshwater eutrophication, linked to the production process of paperboard, which generates high concentrations of phosphorus in the paper mill sludge.

The question often arises concerning the implications that the shift from SUP to reusable tableware can have in moving the environmental burden from one category to another or from one life cycle stage to another. It is important to understand the possible trade-offs when new business models emerge that aim to contribute to circular economy (Lewis et al., 2021). Figure 5.1. displays the variation between the environmental impacts of the reuse and single-use systems. Apart from the reuse system by Company A, a result reinforced also by the single score analysis, there is very little evidence to suggest a shift of environmental impacts between the reuse systems and the single-use system.

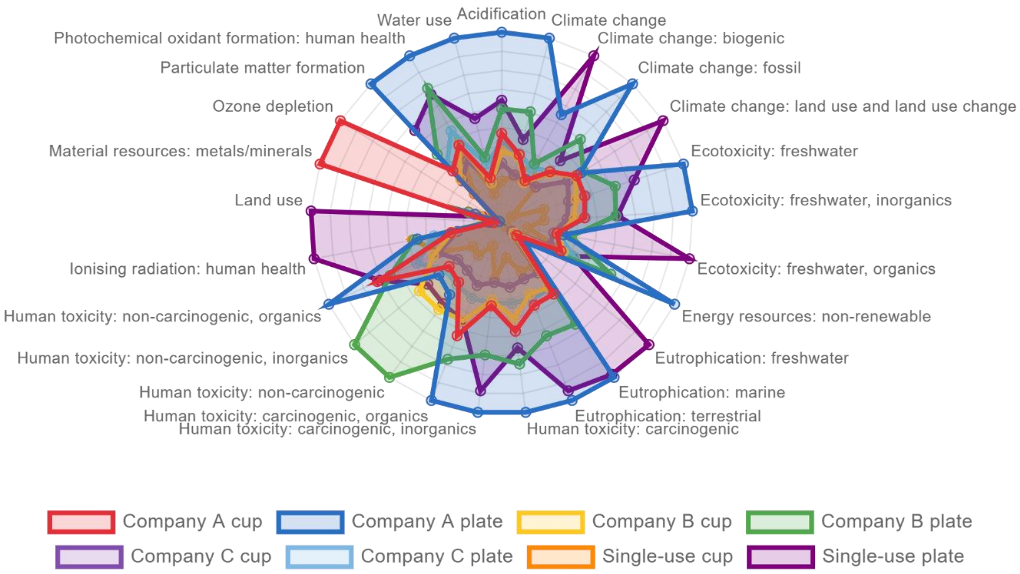

Figure 5.1 Radar chart of the LCIA results in the baseline scenario

5.1.2. Breakdown of impacts by life cycle stage

In this section the study focuses on the climate change impact category in the baseline scenario to delve into the impacts by life cycle stage. As can be seen on Figures 5.2 and 5.3 the extent of the impact varies from one life cycle stage to another and the environmental burden of the tableware has shifter from one life cycle stage to another.

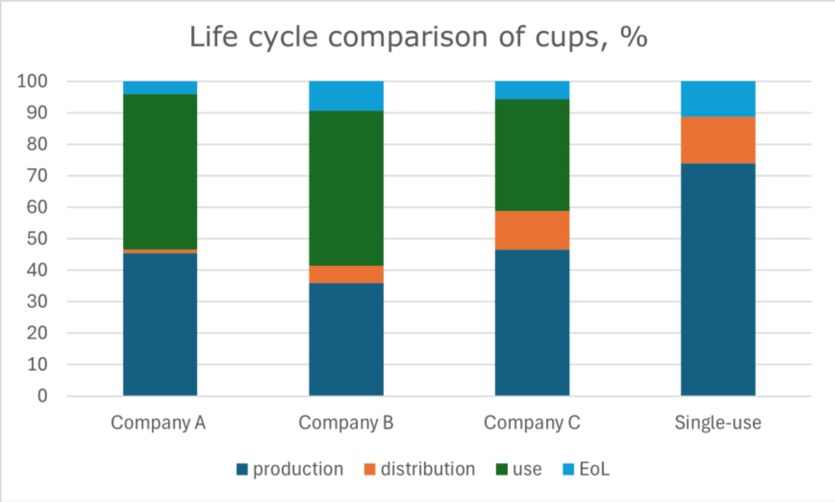

Figure 5.2 Comparison of life cycle stages of cups in climate change category in baseline scenario

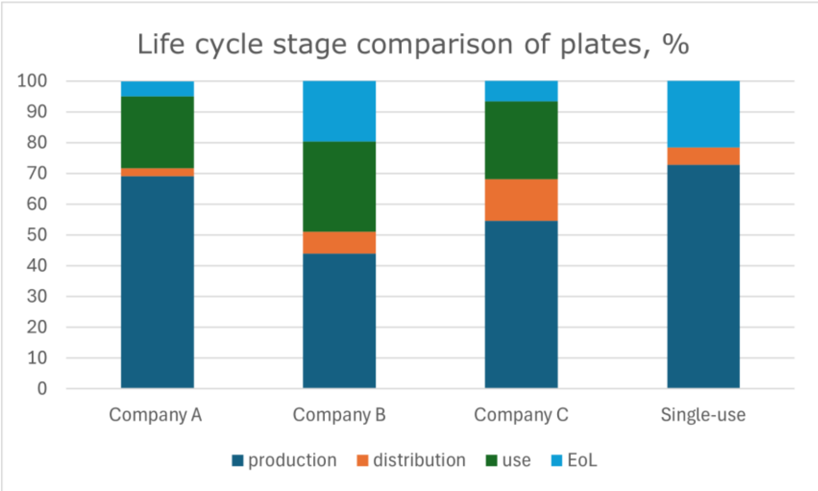

Figure 5.3 Comparison of life cycle stages of plates in climate change category in baseline scenario

Production stage

The production stage of the tableware exerts the largest impact in the single-use system, exceeding 70%. This result has been demonstrated also in other studies (Genovesi et al., 2022; Miele et al., 2024). For reusable tableware, the production stage in the baseline scenario accounts for 35%-50% of the impacts and it is as significant (in case of cups) or more (in case of plates) than the use stage. The high impact of the production stage can be explained by the low count of uses in the baseline scenario – 25. The GHG emissions from the production stage decrease with the increase of the number of times the tableware is used, meaning from baseline to 1st scenario (50 uses) to 2nd scenario (100), which has also been demonstrated by other studies (Aggarwal, 2024).

The lowest impact of the production stage is in the scenario 5 (Figure 5.4), because no additional tableware needs to be produced to substitute the lost ones. In this scenario the environmental impacts of reuse systems decreased across categories and already with 25 uses the reuse system by all companies performed better than the single-use option. This has important implications to the design of the reuse system, as it emphasizes the need to ensure the highest return rate for reusable tableware at events to reduce the environmental burden of the system.

In case of the plate by Company A, the impact production stage has in the baseline scenario is almost equal to that of the disposable plate. This can be attributed to the production process of styrene-acrylonitrile copolymer (SAN), which has a higher energy demand and emits more GHG than the manufacturing of PP or PET (Ecoinvent database v3.11). In addition, the plate by the Company A is 40% heavier than other reusable plates in the analysis and 80% heavier than the single-use plate, which contributes to more emissions both from production and transport (Lewis et al., 2021). The SAN plate also demonstrates a consistently high impact in the acidification category in almost all scenarios, which can be traced back to the production process of SAN that requires ammonia (Karali et al., 2024). Therefore, it can be concluded that under conditions of this study PP has a lower environmental impact in the reuse system than SAN.

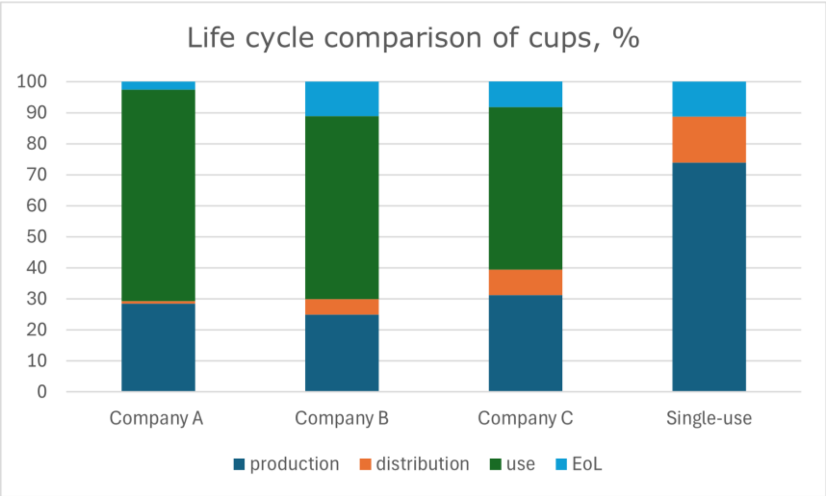

Figure 5.4 Comparison of life cycle stages of plates in climate change category in scenario 5

In the single-use system the choice of the raw material is also important. The paperboard plate shows consistently better results in most scenarios and categories than other tableware items. This is consistent with other studies, which have shown that paperboard tableware is the best performing disposable tableware (Campbell et al., 2020). In future studies SUP tableware produced from other materials such as PET or paperboard with a PE lining could be analysed.

The sensitivity analyses indicate that the reuse rate of the tableware also influences the impacts of the production stage. In the 4th scenario, where the loss rate is 10%, the reuse system has a bigger environmental burden than the single-use system. Contrarily, the reuse system exhibits better environmental performance in scenario 2, where the tableware is used for 100 events compared to the baseline scenario for 25 events and in scenario 5, where the loss rate is 0. These results suggest that it is important to use the reusable tableware as many times as possible to reduce the environmental impacts related to the production stage.

Distribution stage

The distribution stage in the reuse system has a relatively small contribution to the climate change impact category. Although shorter transport distances for the tableware by Company A are reflected also by lower impacts, this does not significantly reduce the emissions generated during its whole life cycle. This is demonstrated by the large environmental burden of Company A compared to the other reuse systems.

In this study, the single-use tableware was sourced from countries rather close to Estonia, which makes the distances travelled in this stage shorter and emissions smaller. It can be expected that if the SUP cups and plates were procured from countries like Türkiye and China, the impacts in the distribution stage would intensify, thereby impairing the overall environmental performance of the disposable tableware.

The distribution stage also includes packaging, but the extent of its effect to total life cycle emissions in all systems is only 1%-2%.

Use stage

For the reusable tableware the largest impact comes from the use stage, which ranges from 40% to 70%. The impacts can be divided to electricity, water, detergent use, wastewater treatment and transportation. The largest contributor to GHG emissions is electricity consumption, which in the baseline scenario forms about a quarter of impacts across the life cycle. Similar results have also been demonstrated by Yadav et al. (2024) and Hitt et al. (2025). For Company C the impacts generated by electricity consumption are lower by 5% due to the use of solar energy, which constitutes a significant reduction in emissions for the system.

In the baseline scenario transport between the events and the washing facility creates about 3% of the total emissions. Detergent use accounts for ca 4%. This is slightly lower than results from a study by Walker et al. (2024), where use of detergents amounted to 10% of the impacts. In this study, water use and wastewater treatment influences the life cycle impacts very little, about 0,2%. However, it must be noted that Estonia is a country that does not suffer from water scarcity. In countries where there is insufficient clean drinking water, the reuse system might not be favoured, because of the increased water use during the use stage.

In scenarios 1 and 2 where the reuse rate of tableware increases, the GHG emissions from the use stage increase, too. This is caused mainly by the upsurge of electricity consumption for the washing phase and not from the increased transport distances. However, in the 3rd scenario where the transport distances for the reuse system are increased tenfold, the impact of the transportation does surpass that of electricity consumption and forms 25% of the total impacts. In future studies, it would be interesting to analyse how many times the tableware needs to be used to demonstrate lower environmental burden than disposable tableware when the events take place a long distance from the service providers.

End-of-life stage

All the reuse systems have a different EoL scenario, which allows investigating the waste treatment options for reusable tableware. However, modelling of recycling scenarios in LCAs is a disputed issue for a long time (Ekvall et al., 2020). Studies use different approaches to model closed-loop and open-loop scenarios which then affect the results of the studies. In some cases, this might even result in outcomes that are unexpected e.g. when incineration displays lower environmental impacts than material recovery although the latter is placed higher in the waste hierarchy (Walker et al., 2024). This study presents a similar case as can be observed in Figure 5.5. The open-loop recycling has the highest GHG emissions compared to incineration and closed-loop recycling.

Since Ecoinvent does not take into account substituting primary material for recycled material in the cut-off version, the open-loop recycling considered here for Company B excludes the benefit from using recycled materials as inputs in other systems. In addition, the lower GHG emissions from incineration may be caused by the credits for avoided Estonian fossil-based electricity production. Conversely, the emissions generated during the incineration of single-use tableware exceed the incineration of reusable tableware. This is because the mass of the disposable tableware generated is bigger than the reusable tableware, as more single-use plates and cups are needed to fulfil the functional unit of the study. The GHG emissions from incineration in the single-use system also exceeds those of the closed-loop recycling by Company C, which also includes the transportation of the tableware back to the producer.

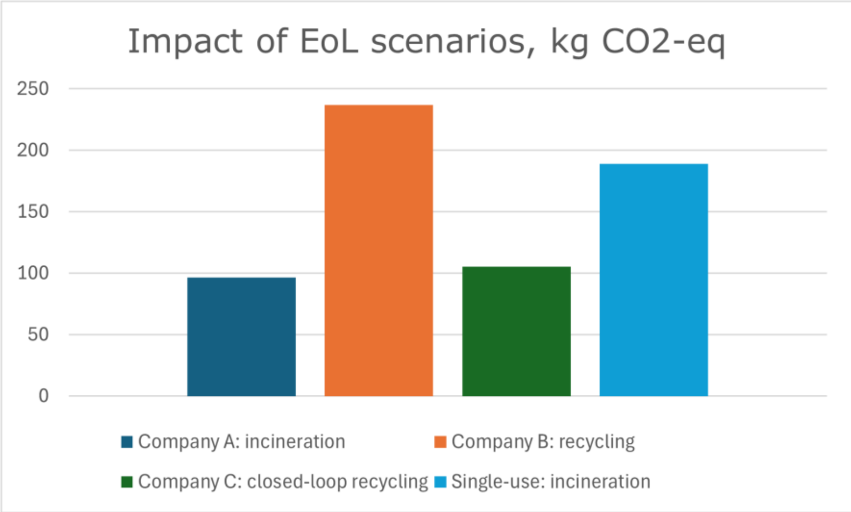

Figure 5.5 Comparison of different EoL scenarios for plates in climate chance impact category, baseline scenario

The EoL stage for disposable tableware forms up to 25% of the whole life cycle impacts in the baseline scenario, whereas in the reuse system EoL contributes little to the total impacts (Figure 5.2; Figure 5.3). This outcome is consistent with other studies (Lewis et al., 2021). Although for the reuse system the EoL stage has less influence on the overall performance of the system than in the case of the single-use system, it is still important to send the tableware for recycling instead of incineration to contribute to circular economy. In addition, as demonstrated by the findings of this study, a closed-loop recycling scenario can aid in reducing the GHG emissions of the life cycle. Here again it must be reiterated that the material selection of the tableware is important, as it not only impacts the performance of the production stage but also the EoL stage.

6. CONCLUSIONS

The findings of this study suggest that the reuse systems, except by Company A, have a lower environmental burden than the single-use system after 25 uses. However, it is also evident that reuse systems can have very dissimilar environmental impacts as is demonstrated by the varying extent of impacts by Company A and Company C. Therefore, it is useful to analyse the life cycle stages of the different systems and propose ecodesign solutions to improve the systems' environmental performances.

The material selection for the tableware is significant. The results of the LCIA clearly show that when compared to PP, the use of SAN plate produces bigger environmental impacts in the production stage. PP shows better results than PET and SAN in most categories. Material choice also influences the environmental impacts from EoL stage, as some materials, such as PP, are easily recyclable and have a local market as Company B has demonstrated. It would also be advisable to prefer reusable tableware with a smaller mass, while keeping in mind its structural durability to prolong the lifetime and increase the number of uses of the tableware.

A particularly important life cycle stage to focus on is the use stage. To reduce the environmental burden of the use stage, it is recommended to invest in energy efficiency and reduce electricity consumption during the washing phase. Likely measures include installing solar panels like Company C or optimising dishwashers. These efforts would also make the system more cost-effective in the long run. Furthermore, detergent use influences the environmental impacts of the system. Detergents that use less or no palm oil for fatty alcohol sulphate production should be preferred (Miele et al., 2024). Although in most scenarios the transportation in the use stage did not have a significant impact, the results from scenario 3 indicate that if the reusable tableware is used at events located further from the site of the washing facility and warehouse, the environmental burden increases and surpasses that of the disposable tableware. Therefore, significant environmental gain would be achieved if there are service providers for reusable tableware in other parts of Estonia as well to have industrial washing facilities closer to events that are taking place outside of Tallinn.

Another element to consider is the design of the reuse system implemented. The sensitivity analyses demonstrated that the increased need for transport and washing in scenarios 1 and 2 do increase the environmental impacts of the reuse system, but the additional tableware produced to replace the lost cups and plates in scenario 4 have a significantly larger influence. Scenario 5, on the other hand, demonstrated how a very effective reuse system generates less environmental impacts than the baseline scenario.

A high return rate of reusable tableware should be the priority when designing a reuse system.

Studies have demonstrated that considering customer behaviour is fundamental when designing reuse systems (Hitt et al., 2023; Yadav et al., 2024). In a well-designed return system that includes a deposit scheme the return rate can reach 99%. Without a deposit scheme the loss rate can even reach 25% (Company A, personal communication, April 1, 2025). To ensure high return rates at large public events, the reusable tableware needs to be collected quickly, with adequate access to return counters provided throughout the event, particularly at the end of the event (Campbell et al., 2020). Furthermore, special designs such as printed brands or logos of the event on reusable cups and plates should be avoided to reduce the probability of users taking them home as a souvenir (Vercalsteren et al., 2006). It was also revealed that when events are organised outside of cities like Tallinn and Tartu, e.g. in smaller towns, people might use the reusable cups or plates as a takeaway container to take some food with them on the road, while driving back to their home. The loss percentage in that case is significantly higher than 5% (Company A, personal communication, April 1, 2025).

From this follows that effective communication is paramount. It may be useful to present the reuse system on posters at the event grounds and describe its effects on the event's sustainability. This would increase the awareness of eventgoers and make them want to be a part of the event's sustainability efforts. It also might be worthwhile to communicate to event organisers and eventgoers not to reject cups or plates that might have some scratches, which do not pose threat to human health. Obviously, food safety is of utmost importance, but if this is ensured, then for sustainability reasons we as consumers should not be too picky.

Another important aspect for why reuse systems should be promoted and preferred is the fact that deposit schemes provide the consumer an incentive to return the packaging and therefore discourage littering (European Commission, 2018). Littering is not reflected in the LCA studies, but can have a significant environmental impact, especially in marine ecosystems (Lewis et al., 2021). In addition to implementing reusable tableware systems at events, transitioning into reuse systems in other areas, e.g. restaurant takeaway containers would contribute to the fight against plastic pollution and marine litter, as well as mounting plastic waste. However, first it is important to conduct a thorough analysis of environmental impacts that may come with this transition to design the most eco-efficient reuse system.

SUMMARY

The global production volume of plastic increases year by year. 40% of plastics in the EU is used for packaging, which includes disposable food containers, cups and plates. This has led to increasing amounts of plastic waste and growing plastic pollution, especially marine litter. The EU has recognized the issue and with the Single-Use Plastics Directive passed in 2019 has restricted the market access of single-use tableware. In Estonia, single-use plastic dishes were banned at public events from January 1, 2024. Many companies have emerged offering reusable plastic tableware rental services as a sustainable alternative to disposable tableware. This service encompasses transporting the dishes to and from the event, as well as washing them. Numerous studies have found that reuse scenarios for food containers typically have lower environmental impacts compared to single-use alternatives. However, the environmental impact of the reuse system is influenced by factors such as the production location, the electricity mix used in manufacturing, end-of-life scenarios, design of the reuse system and user behaviour. Therefore, a study in the local conditions for reuse systems in Estonia was needed.

This thesis aimed to evaluate the environmental impact of plastic tableware reuse systems by three companies in Estonia compared to single-use plastic alternatives. The tableware under study were a 400 ml drink cup and a main plate. 16 categories across all life cycle stages employing the Environmental Footprint method 3.1. were analysed. To quantify this impact, a life cycle assessment (LCA) using a cradle-to-grave approach was conducted, adhering to the International Organization for Standardization (ISO) 14040/14044 standards (ISO, 2006). For the modelling OpenLCA software was used.

Five sensitivity analyses were performed to understand the impact of factors on results. In the first and second scenarios, the study examined how the number of reuses of reusable tableware affects the system's environmental metrics. In the third scenario, it investigated how long distances between the event and the washing facility affect the environmental performance of the reuse system. In the 4th and 5th scenario the aim was to understand what kind of effect the loss rate of tableware has on the environmental performance of reuse systems.

The results of the LCA indicate that the reuse systems surpass the single-use cup and plate in environmental performance after just 25 uses. Only for one company, the environmental impacts become less significant compared to the single-use system after

50 uses. The reuse system performed best when the loss rate of tableware was 0%. The most significant impact categories were climate change and fossil resource use.

The environmental impacts differ between the reuse and single-use systems when analysed by life cycle stage. In the single-use system the impact from the production stage exceeds 70% of the total life cycle impacts. For the reusable tableware, the largest impact comes from the use stage, which ranges from 40% to 70%. The findings of the LCA indicate that for the reusable tableware the largest contributor to GHG emissions in the use stage is electricity consumption during the washing phase, which forms 25% of impacts across the life cycle. Transport in the use stage accounts for ca 3% of the total emissions, while detergent use 4%. Only when the transport distances in the use phase increase by almost 20 times, does the transportation surpass electricity consumption.

The study's findings led to several recommendations for the eco-design of the systems. First, attention must be paid to the material of the reusable tableware and choose tableware that is smaller in mass, durable and at the end of its life easily recyclable. Most important is to ensure that the return rate of the tableware at events is as close to 100% as possible. This can be achieved by implementing deposit schemes and quick collection and avoiding printed designs on tableware. It is important to increase the awareness of eventgoers about reusable tableware and encourage them to return the tableware to the collection points. Small tweaks in the use stage also include using detergents that use less or no palm oil, installing solar panels and improve energy efficiency during the washing phase. Finally, to improve the environmental performance of reuse systems, it would be beneficial to have service providers of reusable tableware also in other parts of Estonia ensuring that industrial washing facilities are closer to events held outside of Tallinn.

The results of this study demonstrate the environmental benefits of adopting and promoting tableware reuse systems at public events in Estonia with attention paid on the reuse rate of the tableware and efficiency of the washing and transport phases.

KOKKUVÕTE

Igal aastal toodetakse maailmas aina suuremas koguses plasti. Euroopa Liidus (EL) toodetakse 40% plastist pakenditeks, sealhulgas ühekordselt kasutatavate toidunõude nagu tasside ja taldrikute jaoks. See on toonud kaasa plastjäätmete mahu suurenemise ja plastreostuse, eriti mereprügi, kasvu. Vastuseks nimetatud probleemidele võeti 2019. aastal vastu ühekordse plasti direktiiv, millega piirati ühekordselt kasutatavate lauanõude ELi turule pääsu. 2024. aasta 1. jaanuarist on Eestis keelatud ühekordsete plastnõude kasutamine avalikel üritustel. Mitmed ettevõtted pakuvad ühekordselt kasutatavate lauanõudele jätkusuutlikku alternatiivi korduvkasutatavate plastnõude rentimise teenuse näol. Teenus hõlmab nõude transporti üritusele ja tagasi ning nende pesemist. Arvukad uurimused on leidnud, et korduskasutusnõud avaldavad ühekordselt kasutatavatega võrreldes väiksemat keskkonnamõju. Korduskasutussüsteemi mõju ulatust mõjutavad aga mitmed tegurid nagu tootmise asukoht, tootmises kasutatav elektrienergia segu, toote eluea lõpu stsenaariumid, korduskasutussüsteemi disain ja kasutajate käitumine. Seetõttu on kasulik uurida korduskasutussüsteeme Eesti kohalikes tingimustes.

Uurimustöö eesmärk oli hinnata kolme Eesti ettevõtte plastnõude korduskasutussüsteemide keskkonnamõju võrreldes ühekordselt kasutatavate plastist alternatiividega. Uuritavateks nõudeks olid 400 ml joogitops ja praetaldrik. Analüüsiti 16 mõjukategooriat kõigis elutsükli etappides, kasutades keskkonnajalajälje (*Environmental Footprint*) määramise meetodit 3.1. Mõju kvantifitseerimiseks viidi läbi elutsükli hindamine, kasutades hällist hauani lähenemisviisi ja järgides Rahvusvahelise Standardiorganisatsiooni (ISO) standardeid 14040/14044 (ISO, 2006). Hindamisel kasutati OpenLCA tarkvara.

Uurimises viidi läbi viis tundlikkusanalüüsi, et mõista tegurite mõju tulemustele. Esimeses ja teises stsenaariumis uuriti, kuidas mõjutab nõude kasutuskordade arv korduskasutussüsteemi keskkonnamõju. Kolmandas stsenaariumis uuriti, kuidas sündmuse ja pesula vahelise vahemaa pikkus mõjutab süsteemi keskkonnatoimet. Neljandas ja viiendas stsenaariumis oli eesmärk mõista, millist mõju avaldab nõude kadu süsteemist.

Olelusringi hindamise tulemused näitavad, et korduskasutussüsteemidel on ühekordselt kasutatavatest topsidest ja taldrikutest väiksem keskkonnamõju pärast 25 kasutuskorda. Ainult ühe ettevõtte puhul muutub keskkonnamõju ühekordselt kasutatava süsteemiga võrreldes vähem oluliseks pärast 50 kasutuskorda.

Korduskasutussüsteem toimis kõige paremini, kui lauanõude kadu oli 0%. Kõige olulisemad mõjukategooriad olid kliimamuutused ja fossiilsete ressursside kasutamine.

Analüüsi tulemustest selgus, et keskkonnamõjud erinevad kahe süsteemi ja elutsükli etappide lõikes. Ühekordselt kasutatavate nõude puhul tekib 70% kogu elutsükli mõjudest tootmisetapis. Korduvkasutatavate nõude puhul tuleb suurim mõju kasutusetapist, mis jääb vahemikku 40–70%. Selles etapis on kasvuhoonegaaside heitkoguste suurim põhjustaja elektrienergia tarbimine pesemisfaasis, mis moodustab 25% kogu elutsükli mõjudest. Nõude transport pesula ja ürituste vahel on umbes 3% koguheitest, samas kui nõudepesuvahendi kasutamine moodustab 4%. Alles siis, kui transpordivahemaad kasutusfaasis algtasemega võrreldes peaaegu 20 korda suurenevad, ületab transport elektrienergia tarbimist.

Uuringu tulemused võimaldasid teha soovitusi süsteemide ökodisaini osas. Esiteks tuleb pöörata tähelepanu korduskasutusnõude materjalile ja valida väiksema kaaluga, vastupidavad ja elutsükli lõpus kergesti ümbertöödeldavad nõud. Kõige olulisem on tagada, et nõude tagastusmäär oleks üritustel võimalikult 100% lähedal. Seda aitab saavutada pandisüsteemi kehtestamine, kiire tagastusprotsess ning nõudel trükitud kujunduste vältimine. Oluline on suurendada ürituste külastajate teadlikkust korduskasutusnõudest ja julgustada neid nõusid kogumispunktidesse tagastama. Keskkonnamõjude vähendamist kasutusetapis toetavad ka vähese või üldse mitte palmiõli sisaldavate pesuvahendite kasutamine, päikesepaneelide paigaldamine taastuvenergia tootmiseks ja energiatõhususe parandamine pesemisfaasis. Korduskasutussüsteemi keskkonnatoimet parandaks ka see, kui korduskasutusnõude teenusepakkujaid asuks teisteski Eesti piirkondades, mis vähendaks nende ja Tallinnast väljaspool toimuvate ürituste vahelist vahemaad.

Selle uuringu tulemused näitavad lauanõude korduskasutussüsteemide kasutuselevõtu ja edendamise keskkonnakasu avalikel üritustel Eestis, pöörates tähelepanu lauanõude korduskasutusmäärale ning pesemise ja transpordi tõhususele.

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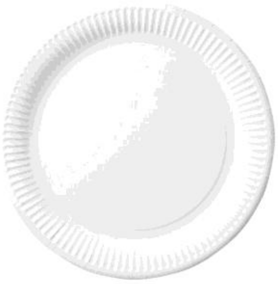
APPENDICES

Appendix 1 Sample photos of reusable tableware

Appendix 2 Sample photos of single-use tableware



Appendix 3 LCIA results of Scenario 1

Indicator	Company A cup	Company A plate	Company B cup	Company B plate	Company C cup	Company C plate	Single-use cup	Single-use plate	Unit
Acidification	5,04	10,20	4,04	6,23	2,65	4,32	2,67	7,76	mol H+-Eq
Climate change	1291,87	3274,55	1225,47	2000,20	687,47	1202,79	993,33	1773,60	kg CO2-Eq
Ecotoxicity: freshwater	4581,85	9435,79	4083,46	6045,95	3268,04	4509,16	2571,85	8113,90	CTUe
Energy resources: non-renewable	21067,90	58380,40	21427,00	37135,30	14182,00	24199,20	29570,80	29319,10	MJ, net calorific value
Eutrophication: freshwater	0,47	0,68	0,38	0,58	0,23	0,40	0,17	5,08	kg P-Eq
Eutrophication: marine	1,15	2,35	1,03	1,57	0,66	1,01	0,77	2,72	kg N-Eq
Eutrophication: terrestrial	10,24	20,86	8,76	13,51	5,35	8,83	6,75	23,32	mol N-Eq
Human toxicity: carcinogenic	6,25E-07	1,04E-06	5,43E-07	7,79E-07	2,52E-07	4,38E-07	2,14E-07	7,70E-07	CTUh
Human toxicity: non-carcinogenic	1,23E-05	1,50E-05	1,70E-05	2,97E-05	8,37E-06	1,24E-05	7,50E-06	1,89E-05	CTUh
Ionising radiation: human health	134,49	218,98	129,27	223,10	97,47	168,72	274,48	577,28	kBq U235-Eq
Land use	5160,49	7342,39	5296,31	8080,70	4451,76	6715,38	4448,92	129402,00	dimensionless
Material resources: metals/minerals	5,35E-02	9,18E-03	7,18E-03	1,08E-02	7,92E-03	1,09E-02	4,61E-03	7,24E-03	kg Sb-Eq
Ozone depletion	1,92E-03	3,17E-05	3,00E-05	5,31E-05	2,67E-05	4,16E-05	4,02E-05	3,93E-05	kg CFC-11-Eq
Particulate matter formation	4,32E-05	1,08E-04	3,61E-05	5,52E-05	2,82E-05	4,43E-05	2,71E-05	8,61E-05	disease incidence
Photochemical oxidant formation: human health	3,96	8,17	3,88	6,57	2,67	4,39	4,59	7,56	kg NMVOC-Eq
Water use	365,61	1392,30	325,40	516,14	331,61	475,61	270,54	975,40	m3 world Eq deprived

Appendix 4 LCIA results of Scenario 2

Indicator	Company A cup	Company A plate	Company B cup	Company B plate	Company C cup	Company C plate	Single-use cup	Single-use plate	Unit
Acidification	9,94	18,31	7,57	11,49	6,16	7,85	11,40	36,76	mol H+-Eq
Climate change	2651,64	5793,79	2264,44	3638,10	1569,45	2138,72	3148,08	7615,70	kg CO2-Eq
Climate change: biogenic	2,59	6,07	2,41	3,43	2,70	3,27	4,74	13,60	kg CO2-Eq
Climate change: fossil	2636,96	5775,26	2250,14	3621,91	1556,16	2124,34	3141,60	7574,67	kg CO2-Eq
Climate change: land use and land use change	12,09	12,45	11,89	12,76	10,59	11,11	1,74	27,43	kg CO2-Eq
Ecotoxicity: freshwater	9233,29	16922,30	7692,21	11192,70	6984,23	8366,73	5894,44	18879,50	CTUe
Energy resources: non-renewable	41969,60	102343,00	38842,80	66397,60	29128,00	42528,70	74193,40	111368,00	MJ, net calorific value
Eutrophication: freshwater	0,91	1,27	0,72	1,07	0,58	0,74	0,36	10,25	kg P-Eq
Eutrophication: marine	2,27	4,23	1,93	2,89	1,48	1,84	4,19	14,71	kg N-Eq
Eutrophication: terrestrial	20,40	37,43	16,39	24,87	12,47	15,98	42,52	148,23	mol N-Eq
Human toxicity: carcinogenic	1,25E-06	1,93E-06	1,04E-06	1,47E-06	6,90E-07	8,20E-07	5,09E-07	1,82E-06	CTUh
Human toxicity: non-carcinogenic	2,49E-05	2,73E-05	3,08E-05	5,31E-05	1,87E-05	2,29E-05	1,73E-05	4,60E-05	CTUh
Ionising radiation: human health	253,658	394,979	235,090	399,875	201,707	297,743	553,208	1169,760	kBq U235-Eq
Land use	10925,500	12959,400	9719,360	14643,100	8721,090	11822,700	10523,700	264604,000	dimensionless
Material resources: metals/minerals	9,49E-02	1,73E-02	1,36E-02	2,01E-02	1,78E-02	2,06E-02	1,00E-02	1,72E-02	kg Sb-Eq
Ozone depletion	3,32E-03	5,60E-05	5,38E-05	9,43E-05	5,26E-05	7,42E-05	9,78E-05	1,39E-04	kg CFC-11-Eq
Particulate matter formation	9,32E-05	1,89E-04	6,76E-05	1,02E-04	6,06E-05	7,89E-05	2,05E-04	6,99E-04	disease incidence
Photochemical oxidant formation: human health	8,012	14,356	7,066	11,800	5,527	7,760	20,782	55,768	kg NMVOC-Eq
Water use	692,736	2442,650	599,351	932,911	658,230	873,571	566,910	2043,060	m3 world Eq deprived

Appendix 5 LCIA results of Scenario 3

Indicator	Company A cup	Company A plate	Company B cup	Company B plate	Company C cup	Company C plate	Single-use cup	Single-use plate	Unit
Acidification	8,42	7,32	2,65	4,37	2,35	3,29	1,48	4,50	mol H+-Eq
Climate change	2310,01	2581,07	887,20	1545,94	733,96	1096,13	516,24	1001,29	kg CO2-Eq
Climate change: biogenic	3,00	2,28	0,84	1,25	0,91	1,15	1,18	3,32	kg CO2-Eq
Climate change: fossil	2302,41	2575,25	883,21	1541,15	730,20	1091,88	514,65	991,21	kg CO2-Eq
Climate change: land use and land use change	4,60	3,53	3,16	3,55	2,85	3,10	0,41	6,76	kg CO2-Eq
Ecotoxicity: freshwater	7273,35	7129,81	2734,89	4398,48	2605,26	3497,96	1282,32	4123,29	CTUe
Energy resources: non-renewable	44468,10	44350,90	15264,70	27626,70	13026,40	20000,30	15020,30	16135,20	MJ, net calorific value
Eutrophication: freshwater	0,66	0,44	0,23	0,36	0,19	0,26	0,08	2,54	kg P-Eq
Eutrophication: marine	1,71	1,67	0,66	1,08	0,55	0,75	0,46	1,63	kg N-Eq
Eutrophication: terrestrial	16,41	15,39	5,84	9,64	4,98	7,03	4,16	14,65	mol N-Eq
Human toxicity: carcinogenic	7,02E-07	6,95E-07	3,27E-07	5,00E-07	2,38E-07	3,12E-07	1,07E-07	3,92E-07	CTUh
Human toxicity: non-carcinogenic	2,17E-05	1,36E-05	1,17E-05	2,12E-05	7,43E-06	1,03E-05	3,69E-06	9,64E-06	CTUh
Ionising radiation: human health	253,66	145,45	80,93	144,03	73,87	111,74	137,14	288,98	kBq U235-Eq
Land use	10635,50	7812,27	4108,23	6911,38	4207,76	6255,48	2157,16	64817,40	dimensionless
Material resources: metals/minerals	1,39E-01	7,84E-03	4,84E-03	7,90E-03	6,02E-03	7,73E-03	2,27E-03	3,68E-03	kg Sb-Eq
Ozone depletion	5,35E-03	3,18E-05	2,20E-05	4,04E-05	2,20E-05	3,30E-05	2,03E-05	2,13E-05	kg CFC-11-Eq
Particulate matter formation	8,73E-05	9,78E-05	3,01E-05	5,15E-05	3,14E-05	4,67E-05	1,69E-05	5,83E-05	disease incidence
Photochemical oxidant formation: human health	8,24	6,86	2,85	5,10	2,54	3,83	2,59	4,97	kg NMVOC-Eq
Water use	671,42	913,69	202,88	337,75	217,88	303,48	135,15	489,95	m3 world Eq deprived

Appendix 6 LCIA results of Scenario 4

Indicator	Company A cup	Company A plate	Company B cup	Company B plate	Company C cup	Company C plate	Single-use cup	Single-use plate	Unit
Acidification	3,53	8,27	2,84	4,76	2,37	3,36	1,33	3,88	mol H+-Eq
Climate change	907,15	2757,24	924,21	1619,41	668,98	999,72	497,17	886,80	kg CO2-Eq
Climate change: biogenic	1,18	3,14	1,12	1,64	1,16	1,48	1,18	3,32	kg CO2-Eq
Climate change: fossil	902,78	2750,65	919,94	1614,14	664,99	995,10	495,58	876,74	kg CO2-Eq
Climate change: land use and land use change	3,20	3,45	3,15	3,63	2,83	3,14	0,41	6,75	kg CO2-Eq
Ecotoxicity: freshwater	3101,94	7593,19	2797,77	4524,74	2405,63	3204,28	1289,36	4056,95	CTUe
Energy resources: non-renewable	16210,60	51680,50	17564,70	32203,90	13716,40	21442,40	14792,30	14659,60	MJ, net calorific value
Eutrophication: freshwater	0,31	0,49	0,26	0,43	0,22	0,32	0,08	2,54	kg P-Eq
Eutrophication: marine	0,78	1,89	0,72	1,20	0,55	0,77	0,39	1,36	kg N-Eq
Eutrophication: terrestrial	7,03	16,77	6,21	10,39	4,89	6,95	3,38	11,66	mol N-Eq
Human toxicity: carcinogenic	3,74E-07	7,28E-07	3,43E-07	5,36E-07	2,29E-07	3,03E-07	1,07E-07	3,85E-07	CTUh
Human toxicity: non-carcinogenic	8,74E-06	1,05E-05	1,41E-05	2,60E-05	6,61E-06	9,01E-06	3,76E-06	9,46E-06	CTUh
Ionising radiation: human health	102,84	178,58	104,59	191,65	96,69	152,26	137,28	288,64	kBq U235-Eq
Land use	3699,97	5519,84	4043,27	6596,39	3871,74	5651,81	2240,16	64701,00	dimensionless
Material resources: metals/minerals	4,99E-02	5,48E-03	4,83E-03	7,92E-03	5,50E-03	7,01E-03	2,30E-03	3,62E-03	kg Sb-Eq
Ozone depletion	1,88E-03	2,46E-05	2,55E-05	4,73E-05	2,30E-05	3,51E-05	2,01E-05	1,97E-05	kg CFC-11-Eq
Particulate matter formation	3,01E-05	9,04E-05	2,57E-05	4,23E-05	2,43E-05	3,50E-05	1,36E-05	4,31E-05	disease incidence
Photochemical oxidant formation: human health	3,00	6,91	3,11	5,61	2,47	3,76	2,30	3,78	kg NMVOC-Eq
Water use	278,23	1278,40	248,32	428,32	251,26	364,12	135,54	487,70	m3 world Eq deprived

Appendix 7 LCIA results of Scenario 5

Indicator	Company A cup	Company A plate	Company B cup	Company B plate	Company C cup	Company C plate	Single-use cup	Single-use plate	Unit
Acidification	2,06	3,70	1,64	2,37	1,38	1,69	1,33	3,88	mol H+-Eq
Climate change	526,19	1145,64	472,48	724,77	347,97	460,09	497,17	886,80	kg CO2-Eq
Climate change: biogenic	0,52	1,11	0,50	0,66	0,58	0,66	1,18	3,32	kg CO2-Eq
Climate change: fossil	522,75	1141,52	469,08	721,06	344,80	456,76	495,58	876,74	kg CO2-Eq
Climate change: land use and land use change	2,92	3,01	2,90	3,04	2,58	2,67	0,41	6,75	kg CO2-Eq
Ecotoxicity: freshwater	1912,46	3438,65	1691,25	2341,83	1624,65	1887,76	1289,36	4056,95	CTUe
Energy resources: non-renewable	7994,88	19372,50	7681,05	12575,50	6012,44	8498,60	14792,30	14659,60	MJ, net calorific value
Eutrophication: freshwater	0,20	0,27	0,16	0,23	0,13	0,15	0,08	2,54	kg P-Eq
Eutrophication: marine	0,48	0,86	0,42	0,60	0,34	0,40	0,39	1,36	kg N-Eq
Eutrophication: terrestrial	4,25	7,68	3,54	5,12	2,80	3,46	3,38	11,66	mol N-Eq
Human toxicity: carcinogenic	2,81E-07	4,23E-07	2,37E-07	3,21E-07	1,61E-07	1,86E-07	1,07E-07	3,85E-07	CTUh
Human toxicity: non-carcinogenic	4,96E-06	6,13E-06	6,04E-06	9,98E-06	4,37E-06	5,23E-06	3,76E-06	9,46E-06	CTUh
Ionising radiation: human health	51,23	78,78	46,88	76,26	39,09	55,38	137,28	288,64	kBq U235-Eq
Land use	2080,84	2838,73	2035,35	2904,58	1964,38	2593,50	2240,16	64701,00	dimensionless
Material resources: metals/minerals	1,67E-02	4,13E-03	3,01E-03	4,26E-03	4,24E-03	4,80E-03	2,30E-03	3,62E-03	kg Sb-Eq
Ozone depletion	5,58E-04	1,20E-05	1,04E-05	1,75E-05	1,12E-05	1,53E-05	2,01E-05	1,97E-05	kg CFC-11-Eq
Particulate matter formation	1,77E-05	3,83E-05	1,46E-05	2,10E-05	1,45E-05	1,86E-05	1,36E-05	4,31E-05	disease incidence
Photochemical oxidant formation: human health	1,53	2,87	1,42	2,26	1,19	1,62	2,30	3,78	kg NMVOC-Eq
Water use	139,95	442,12	124,31	182,59	143,67	183,29	135,54	487,70	m3 world Eq deprived