The climate impact of the bread take-back agreement

A scenario-based assessment of decarbonization opportunities along the Swedish bread supply chain

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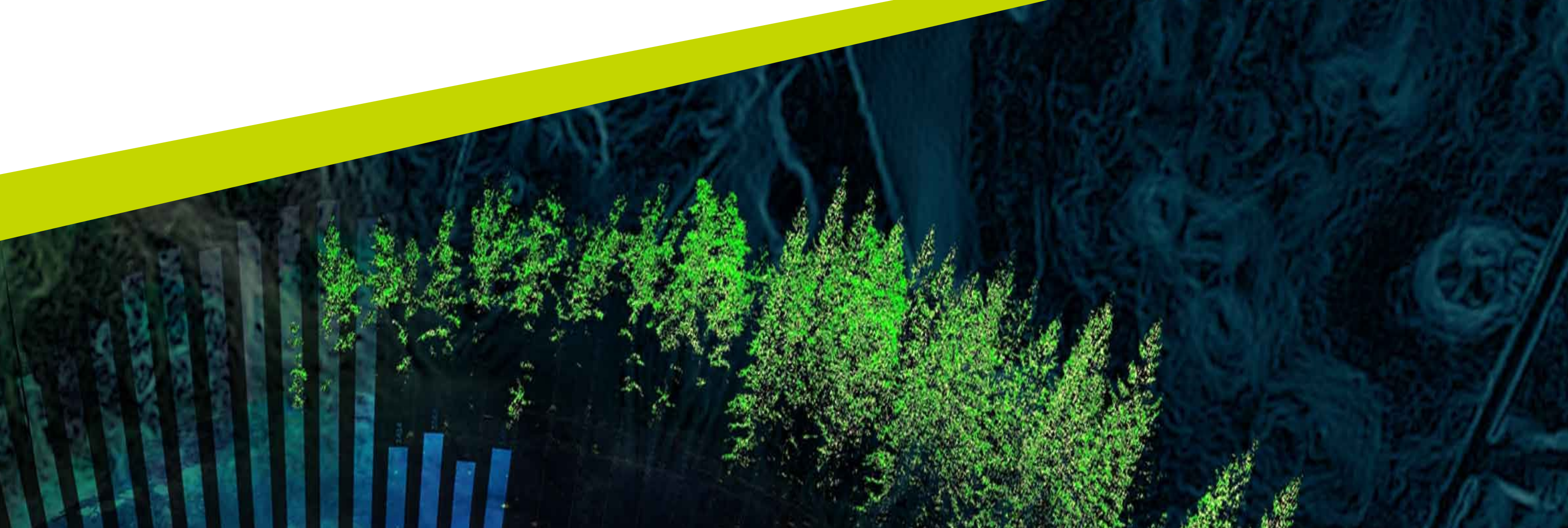

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Abstract

Food waste has economic, environmental, and social implications; the importance of reducing food waste is recognized in Sustainable Development Goal 12.3. The Swedish bread take-back agreement (TBA) has been identified as a risk factor for food waste generation at the supplier-retailer interface. The ideal business model for the bread supply chain remains debated, and the implications of the TBA on transport emissions present a research gap. This study compared the climate impact of the conventional take-back agreement for surplus bread in Sweden to a conceptual system with altered logistics and waste management. Life cycle assessment (LCA) with Global Warming Potential (GWP_{100}) as a single impact category was used to analyze alternative scenarios for the Swedish bread supply chain. The results showed that a shift from a TBA system to a non-TBA system in the city of Uppsala increased the climate impact marginally by 5%. Inversely, in other Swedish cities, the non-TBA scenarios clearly outperformed the TBA system, as transport back to the bakery caused 32% higher emissions and the poor re-valorization of bread held a 11% lower emission savings potential. The average GWP_{100} of all assessed cities is 28% lower for the non-TBA scenarios. The long-distance delivery of bread was identified as an impact hotspot, which points to the necessary decarbonization of the Swedish transport sector. The waste treatment stage offers leverage for emission savings, especially using bread for bioethanol, however, the latter is sensitive to transport distance. For Uppsala, the most prominent benefits come with collaborative approaches that prevent bread wastage in the first place and, at the same time, make use of the clean waste stream created by the TBA.

Keywords: food waste, bread take-back agreement (TBA), life cycle assessment, supply chain, transport emissions

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Abbreviations

AD	Anaerobic digestion
CF	Carbon Footprint
CH_4	Methane
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalents
DEFRA	Department for Environment, Food & Rural Affairs
EPR	Extended Producer Responsibility
EU	European Union
FAO	Food and Agricultural Organization
GHG	Greenhouse gases
GWP	Global Warming Potential
HGV	Heavy Goods Vehicle
IPCC	International Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life Cycle Assessment
N_2O	Nitrous Oxide
NTM	Network for Transport Measures
RC	Redistribution Center
RL	Reverse Logistics
SDG	Sustainable Development Goal
TBA	Take-back agreement
UK	United Kingdom

1. Introduction

Food waste reduction is recognized as a critical element of sustainable development and has received increased attention from actors such as policymakers, researchers, and media in recent years (IPCC 2019; Naturvårdsverket 2020; The Guardian 2021). It is estimated that one third of all food produced for human consumption is wasted (FAO 2011), and recent reports indicate that this figure is likely much higher (UNEP 2021). The wastage of food infers environmental and economic implications and creates ethical controversies (Garnett 2011). The International Panel on Climate Change (IPCC) found that food waste caused 8-10% of anthropogenic greenhouse gas (GHG) emissions during 2010-2016, which highlights its critical role in climate change responses (IPCC 2019). In the face of a growing world population, estimated to reach 10 billion in 2050, more efficient resource use and improved food security are crucial (FAO 2019).

Research advocates for reorganizing food supply chains to make them more sustainable (Kronborg Jensen et al. 2013; Münch et al. 2021). Food waste is generated at all stages of the supply chain (Canali et al. 2016; FAO 2019), and its measurement is crucial to establishing sustainable food systems (WWF-WRAP 2020). Food waste drivers are complex, and it is necessary to determine how business decisions affect food waste creation (Canali et al. 2016). One much debated business model is the bread take-back agreement (TBA) in Sweden.

The baking industry is among the food industries with the highest waste quantities in Sweden (Naturvårdsverket 2020) and bread is one of the most wasted food products in the European Union (Cicatiello et al. 2020). Approximately 90% of packaged bread sold in Sweden is delivered with a TBA (Brancoli et al. 2019). Such trading agreements require bakeries to remove unsold bread from the shelves, thus externalizing the risk and cost of the generated food waste (ibid.). Previous studies identified the TBA as a risk factor for food waste generation at the supplier-retailer interface (Ismatov 2015; Ghosh & Eriksson 2019), and it is suggested that a shift of responsibility for unsold bread to the retailers could incentivize food waste reduction (Brancoli et al. 2019). The TBA also dictates the transport logistics of bread, which would have to be re-organized if retailers took care of surplus bread. Transport contributes only to a limited extent to the food system's emissions

(Garnett 2011; Wakeland et al. 2012), but the Swedish transport sector relies on fossil fuels and generates one third of the national emissions (Xylia & Olsson 2021). The environmental implications of the logistics of the bread supply chain in Sweden have not yet been quantified, thus hindering a holistic evaluation of potential benefits and limitations related to the TBA system. Particularly the implications of the TBA on transport and the climate impact connected to it are research gaps.

1.1 Research aims and questions

The goal of this study is to quantify the climate impact of alternative bread supply chain scenarios in Sweden. Quantifying the climate impact of the current system subsequently enables an in-depth analysis of the impact of scenarios that either omit the TBA or apply changes to it. By that, this study aims to evaluate the implications of the TBA system, and to identify climate impact hotspots and opportunities to make the bread supply chain more sustainable. Therefore, the study aims to address the following research questions:

- How is bread transported from the bakery to end-of-life treatment?
- Can the cancellation of the TBA reduce the climate impact of the bread supply chain in Sweden?
- What role does transport play in the context of the TBA and the climate impact of the bread supply chain?

2. Problem Background

The following section covers the definition of food waste, how it can be managed, as well as the role of the retail stage and return practices for food waste creation. The bread take-back agreement in Sweden is also described, alongside its role in generating food waste.

2.1 The definition of food waste

Terms like ‘food loss’, ‘food waste’, and ‘surplus food’ describe waste connected to food (Teigiserova et al. 2020). However, various definitions exist, and debates about what should be considered food waste are ongoing (Naturvårdsverket 2020; Teigiserova et al. 2020).

The Food and Agricultural Organization of the United Nations (FAO) understands food loss and waste as ‘the decrease in quantity or quality of food along the food supply chain’ (p. xii). While food loss occurs from the harvest up to, but excluding the retail stage, food waste occurs at the retail and consumption stages. This definition is in line with Sustainable Development Goal (SDG) 12.3, which separates goals for food loss and food waste. Inedible parts and those used for alternative economic uses other than human consumption are not considered food loss or waste in this definition (FAO 2019). The EU FUSIONS project defines food waste as ‘(..) any food, and inedible parts of food, removed from the food supply chain to be recovered or disposed (including composted, crops ploughed in/not harvested, anaerobic digestion, bio-energy production, co-generation, incineration, disposal to sewer, landfill or discarded to sea)’ (Östergren et al. 2014:6), as it has been officially adopted by the European Commission (Joint Research Center 2020). Here, food used as animal feed is not considered as food waste; however, the additional processing necessary for bread compared to animal feed production must be noted (Brancoli et al. 2020). On the same note, Naturvårdsverket (2020) defines food waste as anything produced for human consumption but not eaten. Teigiserova et al. (2020) categorize surplus food as edible food that is fit for human consumption but, for some reason, is still discarded. Papargyropoulou et al. (2014:112) state that ‘food surplus is food produced beyond our nutritional needs, and waste is a product of food surplus’. Introducing the term surplus food into the waste hierarchy has been stressed as necessary for preventing wastage and reusing food for human consumption (Teigiserova et al. 2020).

These debated definitions affect comparability, quantification, and target monitoring (Naturvårdsverket 2020; Teigiserova et al. 2020). In this paper, food waste is defined as food produced for human consumption, but not used for this purpose, which aligns with the above definitions used by Brancoli et al. (2019) and Naturvårdsverket (2020).

2.2 The consequences of wasting food

Food waste causes a multitude of challenges both for humans and the planet. The IPCC estimates that the food system generates 21-37% of anthropogenic GHG emissions through agriculture, land use, transport, packaging, processing, retail, and consumption (IPCC 2019). The loss and wastage of food represent a waste of all emissions created at each stage of the supply chain, which are estimated to have caused 8-10% of anthropogenic GHG emissions globally during 2010-2016 (ibid.). Thus, the reduction of food waste is crucial for reaching emission reduction targets (Garnett 2011; WWF-WRAP 2020). Food waste also represents an avoidable use of natural resources such as land, water, and energy, and contributes to biodiversity loss (ibid.). Pressure on natural resources is likely to rise in the face of a growing world population (FAO 2019).

Food waste also infers estimated annual costs of 143 billion euros in the EU (Canali et al. 2014; WWF-WRAP 2020), affecting all actors along the supply chain (Mena et al. 2011). Avoidable food waste harms the income of producers and consumers (Papargyropoulou et al. 2014), but in the current system recovering or preventing food waste can often result in higher costs than wasting (Eriksson et al. 2017). Above that, food waste is also a moral and ethical issue, as it reflects the inequity of our food system. In 2017, 22% of the EU population lived in a household at risk of poverty and 7,4% in severely materially deprived situations, having limited access to suitable, healthy food. At the same time, 88 million tons of food are wasted in the EU every year (WWF-WRAP 2020). Ultimately, wasting still edible food impacts all three pillars of sustainable development: The environment, economy, and society (Eriksson et al. 2017).

2.3 How to reduce food waste

Reducing food wastage is a key lever in combating climate change, as highlighted in the last IPCC report (IPCC 2019). It can improve energy and resource-efficiency of food systems (Garnett 2011), lower GHG emissions along the supply chain (Wunder et al. 2020), reduce the pressure on natural resources, and help to meet increased demands (FAO 2013). More efficient supply chains also have social benefits as they can improve access by lowering prices, thus leading to better food security (Papargyropoulou et al. 2014; FAO 2019). Food waste reduction not only contributes to SDG 12 (Sustainable consumption and production) but can positively affect several SDGs such as SDG 2 (Zero hunger), SDG 13 (Climate action), and SDG 15 (Life on land) (FAO 2019).

SDG 12.3 sets the target to 'halve per capita global food waste at the retail and consumer levels and reduce food losses along production and supply chains, including post-harvest losses' by 2030 (United Nations 2015). The EU and its member states have committed to the SDGs (European Commission & European Parliament 2018). The Farm to Fork Strategy, at the heart of the EU Green Deal, lists the reduction of food loss and waste as a key action point and aims to set legally binding food waste reduction targets across the EU in 2023 (European Commission 2020). The Swedish government is committed to reducing food waste and reaching SDG 12.3 (Livsmedelsverket et al. 2018). Accordingly, an action plan for reduction has been set in place; however, no national food waste reduction target has been established so far (Naturvårdsverket 2020). The Waste Framework Directive defines waste prevention and management principles in the EU and established the waste hierarchy (European Commission & European Parliament 2008). The waste hierarchy defines an order of preference for waste management, with waste prevention as the preferred option and landfilling as the least preferred (European Commission & European Parliament 2018). The concept can be applied to the context of food waste to guide actions addressing it (*Figure 1*).

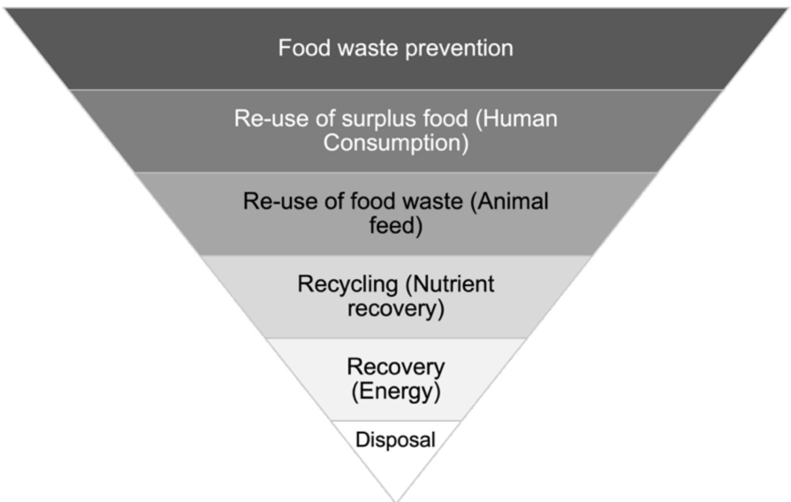

Figure 1: *Food waste applied to the waste hierarchy*
(Own illustration based on Joint Research Center 2020; European Commission 2008; Papargyropoulou et al. 2014)

The ranking shown in the hierarchy shows clearly that food waste prevention should be the primary goal, followed by re-use for human consumption (Joint Research Center 2020). Multiple studies also concluded that food waste prevention infers the highest environmental savings potential (Bernstad Saraiva Schott & Andersson 2015; Slorach et al. 2019). Waste management, on the other hand, can only recover a fraction of the resources invested in food production (Eriksson 2015b). Most generally, the potential for emission reduction is connected to the type of food treated. Bread has a high potential to reduce greenhouse gas emissions; it has a low carbon footprint and high energy and dry-matter content, making it a good substitution for fossil energy carriers (Vandermeersch et al. 2014; Eriksson et al. 2015).

Such preconditions for bread, and the above given framework, imply bread waste is a straightforward issue. However, it is recognized in the Waste Framework Directive that Life cycle assessment (LCA) can be used beyond the waste hierarchy to determine which treatment pathway is the most beneficial for individual situations (European Commission & European Parliament 2008).

2.4 LCA for environmental assessment

Life cycle assessment is one of the most used tools to assess environmental impacts (Ekvall et al. 2007) and a comprehensive, structured, and internationally accepted method (European Commission JRC 2010; Klöpffer et al. 2014). LCA maps the inputs and outputs along the entire life cycle of a product system, ideally from cradle-to-grave (Finkbeiner et al. 2006). It is standardized in ISO 14040/44, providing an iterative framework of four phases: Goal and scope definition, Life cycle inventory analysis, Life cycle impact assessment, and Interpretation (International Organization for Standardization n.d.) (*Figure 2*).

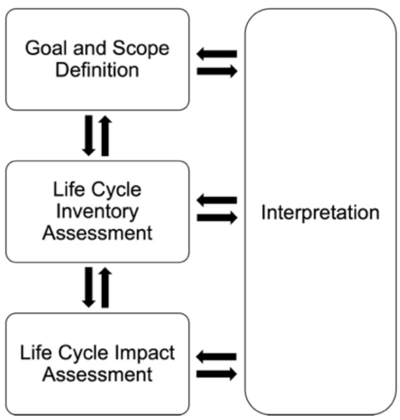

Figure 2: *LCA phases according to ISO (Klöpffer et al. 2014)*

LCA allows impact assessment for multiple impact categories, such as resource use, acidification, or global warming potential (Klöpffer et al. 2014). It thus provides a holistic overview of environmental impacts and helps to avoid burden shifting to other stages (Finkbeiner et al. 2006). This makes LCA a powerful decision-support tool for more sustainable production and consumption (European Commission JRC 2010). The growing concern for climate change and GHG emission reduction has led to an increased interest in the carbon footprint of products, a term today commonly used to describe the global warming impact category in life cycle assessment. The environmental impact of food products has become a particular topic of interest and numerous studies have focused on it (Espinoza-Orias et al. 2011). LCA has also been increasingly used in research on food waste (Scholz et al. 2015; Brancoli et al. 2017).

From an LCA perspective, bread has a low carbon footprint of approximately 0,6-1,2 kg CO_2e (Carbon dioxide equivalents) per kg of bread from cradle to grave (Andersson & Ohlsson 1999; Espinoza-Orias et al. 2011), compared to more resource-intense products, such as beef or cheese (Jensen & Arlbjørn 2014). In general, plant-based products have lower carbon emissions per kg than animal-based products, as livestock farming causes significant emissions from enteric fermentation, feed production, manure, and land-use change (Garnett et al. 2016). Bread is a staple product in many parts of the world and is consumed in large quantities every year (Jensen & Arlbjørn 2014; Axel et al. 2017; Vargas & Simsek 2021). Thus, despite its low climate impact, its environmental impact accumulates when bread is wasted in considerable volumes (Iakovlieva 2021).

2.5 Bread waste in Sweden

Over 750 million kg of bread and confectionery products, more than 70 kg per capita, were consumed in Sweden in 2018 (Jordbruksverket 2019). At the same time, Sweden wasted 1.3 million tons of food in 2018, equaling 133 kg per capita, with numbers steadily increasing since 2012 (Naturvårdsverket 2020). Bread is one of the most common household waste types in Sweden, and the baking industry is among the industries with the highest waste numbers (ibid.).

Several studies confirm the relevance of bread in the food waste discourse. Brancoli et al. (2017) found that next to beef, bread contributes most to supermarkets' economic losses and environmental impacts in Sweden. In a later study, Brancoli et al. (2019) calculated an average economic cost of 240 million euros for bread waste in Sweden. As further concluded by Brancoli et al. (2019), bread waste at the retail level is of utmost importance for reducing food waste in the Swedish context, which is elaborated in the following section.

2.5.1 Return practices at the supplier-retailer interface

Some fundamental causes of waste along the food supply chain are quality standards, freshness, short shelf life, cost pressures, market conventions, and demand for variety (Ghosh & Eriksson 2019). Furthermore, the overfilling of shelves to attract customers, and the removal of items from shelves before their expiration date, are drivers of waste (Cicatiello et al. 2020; Rosenlund et al. 2020). Due to its short customer order and supply chain lead time, its perishability, and its short shelf life, bread has a high waste potential (Ghosh & Eriksson 2019).

Having been overlooked in the past (Mondello et al. 2017), research only recently focused more on food waste at the retail level (Canali et al. 2016; Rosenlund et al. 2020) and at the supplier-retailer interface (Mena et al. 2011; Eriksson et al. 2017; Herzberg et al. 2022). Recent findings suggest that 13% of food waste occurs at the retail stage (UNEP 2021). Even though retail food wastage in Sweden is estimated to be only 8%, it is the second-largest source of food waste after households (Naturvårdsverket 2020). Research suggests a considerable amount of unrecorded food waste at retail stores (Cicatiello et al. 2017), and a change in methodology led to a sharp increase in retail waste quantities in Sweden between 2016 (30,000 tons) and 2018 (100,000 tons) (Naturvårdsverket 2020). As also concluded by Naturvårdsverket (2020) waste at the retail level in Sweden had most likely been heavily underestimated before. Bread waste is generated at the retail level, but is recorded, handled, and paid for at the production level (Canali et al. 2014), and food waste is less noticeable when products are sent back to the supplier, as also pointed out by Rosenlund et al. (2020). Waste at the retail stage has a substantial impact because a lot of value creation happens before, accumulating energy and costs (Mena et al. 2011; Ghosh & Eriksson 2019). There has been increasing evidence that most food discarded at the retail stage is still fit for consumption (Cicatiello et al. 2020), which is especially true for bread and other baked goods (Cicatiello et al. 2017; Brancoli et al. 2019).

Recently, take back clauses have caught increasing attention in research (Ghosh & Eriksson 2019; Goryńska-Goldmann et al. 2020; Rosenlund et al. 2020) and policymaking (Livsmedelsverket et al. 2018). For instance, Canali et al. (2014) mention increased returns and pre-store waste due to supplier-retailer contracts as food waste drivers. Parfitt et al. (2010) determined that contractual penalties, poor demand forecasting, and product take-back clauses cause 10% of over-production and high waste levels in the UK food supply chain. Priefer et al. (2016) also list excess stock due to take-back systems and the cancellation of orders at the manufacturing stage among the main reasons for food waste. Cicatiello et al. (2020) identified take-back agreements as a hotspot for food waste generation. Moreover, a law has been enacted recently in the Czech Republic to prohibit the return of unsold produce to suppliers (Canali et al. 2014; Eriksson et al. 2017). The European Commission also recognized return policies as a possible spot to reduce food waste (European Commission 2018). Next to the role of return practices for bread wastage at the retail level, what happens to bread after the retail stage is important to consider.

2.5.2 Re-valorization of bread waste

The TBA system offers a clean flow of bread that is not mixed with other organic waste, which is usable for various re-valorization methods. This is a benefit compared to waste occurring at the household level, where it is discarded together with other waste for typical municipal waste treatment (Brancoli et al. 2020); in Sweden, that is most commonly incineration (46%) and anaerobic digestion (16%) (Avfall Sverige 2021). Neglected in many previous studies, Jensen & Arlbjørn (2014) found great carbon footprint reduction potential in the waste management stage. Brancoli et al. (2020) conducted a systematic study on the environmental savings potential offered by common valorization pathways for bread waste. Such are, in order of preference according to the results, prevention, ethanol production, usage in animal feed, beer production, donation, incineration, and anaerobic digestion (ibid.). The results mostly correspond with the waste hierarchy (European Commission & European Parliament 2008). All three large bread suppliers in Sweden use returned bread for ethanol production[1]. A study on retail waste management by Mondello et al. (2017) indicates that transport network organization can affect the environmental performance of waste management options. Brancoli et al. (2020) did not include the transport to the respective waste treatment facilities in their study but calculated a distance threshold to reflect how far bread can be transported until a treatment option loses its benefit compared to another. Because this threshold depends on the local availability of infrastructure for waste treatment, Brancoli et al. (2020) recommend assessing specific cases individually. For instance, while ethanol and feed production have a high savings potential but limited local availability and consequently require longer transportation distances, incineration and anaerobic digestion perform weaker in terms of environmental savings but are available locally (ibid.).

2.6 The bread take-back agreement

In Sweden, 90% of the pre-packaged bread market consists of bread suppliers operating with a TBA (Eriksson et al. 2017). The TBA allows the retailer to give back unsold bread and pay only for the amount sold, externalizing the risk and cost

[1] Bakery A, pers. comm. 2022-03-04. and 2021-11-25; Bakery B, pers. comm. 2021-11-04; Bakery C, pers. comm. 2022-01-28.

of the generated food waste (Brancoli et al. 2019). Previous studies identified the TBA for bread in Sweden as a risk factor for food waste generation at the supplier-retailer interface (Eriksson et al. 2017; Brancoli et al. 2019; Ghosh & Eriksson 2019). The return of unsold products with a refund is beneficial for the retailers as it forces suppliers to deal with transport, re-manufacturing, secondary markets, or disposal of bread (Ghosh & Eriksson 2019).

Brancoli et al. (2019) found that a major part of bread waste occurs at the supplier-retailer interface, and that 39% of waste consists of TBA products, making it the product with the highest waste levels at the retail stage. Eriksson et al. (2017) found significantly higher return levels for bread sold with a TBA than for that sold without such an agreement. Gosh & Eriksson (2019) further concluded that bread suppliers experience significantly higher rejection rates (~30%) when delivering with a TBA. Returned bread was also identified as one of the leading waste causes at Swedish bakeries (Iakovlieva 2021). When suppliers pick up unsold bread, retailers do not have much incentive to offer these products at a discount (Eriksson et al. 2017), which requires time and resources (Rosenlund et al. 2020). Eventually, it can be more economically profitable to waste food when its recovery is costly (Eriksson et al. 2017).

The Swedish action plan for food waste reduction recognizes the role of return practices and includes the 'mapping of business and logistics systems that create food waste, including systems for handling bread and returns' as a measure to reduce food waste (Livsmedelsverket et al. 2018:10). Beyond that, several studies confirm that return practices are a risk factor for bread wastage beyond the Swedish case, for instance in Austria (Lebersorger & Schneider 2014) and Poland (Goryńska-Goldmann et al. 2020).

Previous research found that most retailers see the TBA as a profitable arrangement, while bakeries are adverse to the agreement (Eriksson et al. 2017). For instance, one bread supplier stated that ideally, the retailer would take care of the bread, but a system change would require time and effort for various changes in the logistics system, prices, contracts, and drivers' jobs (ibid.). Trials that have been conducted by industry stakeholders have proven to reduce bread waste[2] (Company confidential 2021f). However, other waste drivers beyond the TBA are also mentioned frequently, such as the customers' expectations for freshness, the demand to fill shelves from the retailer side, and too much shelf space[3]. Most

[2] Store manager, pers. comm. 2022-03-01.

[3] Bakery A, pers. comm. 2022-03-04. and 2021-11-25; Bakery B, pers. comm. 2021-11-04; Bakery C, pers. comm. 2022-01-28.; Bakery D, pers. comm. 2021-11-16; Retailer A, pers. comm. 2021-11-30 and 2022-02-08; Retailer B, pers. comm. 2022-01-10; Retailer C, pers. comm. 2021-11-29; Retailer D, pers. comm. 2022-01-18; Retailer E, pers. comm. 2021-11-17.

profoundly, large assortments are a well-mentioned waste driver, as they make forecasting more difficult (ibid.). Overall, most recently, the topic has received greater polarization, and actors from both the retailer and bakery sides have been critical of the TBA (ibid.).

Several studies suggest that bread waste levels could go down if retailers had to take responsibility for unsold products (Lebersorger & Schneider 2014; Brancoli et al. 2019; Rosenlund et al. 2020). This could incentivize better demand forecasting and ordering, as well as waste reduction actions such as discounts for products close to their best-before date (ibid.). Food wastage for economic reasons could be reduced through risk-sharing along the supply chain (Herzberg et al. 2022). Brancoli et al. (2019) conclude that the bread take-back agreement is a priority area for food waste reduction in Sweden.

2.6.1 The theoretical background of the TBA

The bread take-back agreement is based on the Extended Producer Responsibility (EPR) concept (Eriksson et al. 2017). The Waste Framework Directive highlights EPR as a concept to ensure higher responsibility for producers and encourage the prevention, re-use, recycling, and recovery of waste (European Commission & European Parliament 2008). Avfall Sverige (2021) also recognizes producer responsibility to improve waste management and product development. Based on EPR, used or discarded products are sent back to the supplier, thus a reverse supply chain is operated (Eriksson et al. 2017). The reverse logistics (RL) concept introduces circularity into supply chains by establishing a backward flow of products, allowing the producer to recapture value through reprocessing or an appropriate disposal (Kronborg Jensen et al. 2013; Banihashemi et al. 2019; Münch et al. 2021). This can reduce waste, use resources more sustainably, and create a competitive advantage for companies (ibid.)

Particularly in food supply chains and at the supplier retailer interface, RL schemes are suspected not to improve sustainability (Eriksson et al. 2017). The high concentration of retailer power in Sweden is believed to affect the implications of the TBA (Brancoli et al. 2019). As shown in *Table 1*, 90% of the Swedish retail market is controlled by Coop, ICA, and Axfood (DLF et al. 2021); it is almost 95% when including recently acquired Bergendahls into Axfood (Axfood 2021). This level of market concentration is comparable to other Nordic countries, but much higher than Germany (60%), Spain, or France (both 50%) (Konkurrensverket 2018). Pågen, Fazer, and Polarbröd make up more than 80% of Sweden's bakery sector (*Table 1*), and most stores of the large retailers operate under a TBA with these bakeries (Brancoli et al. 2019).

Table 1: *Market shares of stakeholders in the Swedish bread industry*

Stakeholder	**Market share**
Retailer	
ICA	52,5%
Coop	18,1%
Axfood	18,9%
Lidl	5,3%
Bergendahls (now part of Axfood)	5,2%
Bakery	
Pågen	39%
Fazer	25%
Polarbröd	19%
Others	7%

2.6.2 The role of transport within the TBA

In the Nordic countries, 70% of transport emissions derive from road transport, of which a quarter origin from heavy road freight (Liimatainen et al. 2014a). Food transport in particular has previously been found to only represents a minor fraction of the total food supply chain emissions (Garnett 2011; Wakeland et al. 2012). In 2003, only 3,5% of the UK's total GHG emissions were linked to inland food transport (Garnett 2000). Generally, transport emissions depend on vehicle type and size, fuel type and consumption, traffic conditions, load, empty trips (Braam et al. 2001; Liimatainen et al. 2014b; a), and factors such as refrigeration (Garnett 2000). Fuel efficiency, and therefore also CO_2 emissions, vary with vehicle load (DEFRA 2021b). Even though increased loads require more fuel per distance, the required mileage is lower and energy efficiency higher, thus reducing CO_2 emissions (Liimatainen et al. 2014b). Notably, it is generally recognized that so-called 'food miles' are not a good indicator of a product's sustainability due to trade-offs with other life cycle stages such as production or storage (Wakeland et al. 2012; Garnett et al. 2016).

Swedish bakeries operate a sophisticated, circular logistics system, delivering bread all over Sweden (Eriksson et al. 2017). Operating the delivery and pick-up of bread simultaneously avoids 'the extra mile' (Company confidential 2021e). With its roots in the above-described concept of reverse logistics, such a circular supply chain creates a clean waste stream and avoids the empty backhaul of trucks (ibid.) Beyond that, efficient logistics reduce not only environmental but also economic costs for companies (Eriksson et al. 2017). Canceling the TBA will affect the logistics operated within the Swedish bread industry, particularly after the retail stage (ibid.); however, to what extent and with what implications for climate impact, is still unknown.

3. Life cycle assessment

Based on models of the bread supply chain in Sweden, LCA was used to assess the climate impact of the TBA. The results' sensitivity, particularly regarding transport, was evaluated by simulating alternative bread management scenarios. Maps created in ArcMap 10.7 provide visual support for the assessment and the results.

3.1 Goal and scope definition

This LCA aims to evaluate the climate impact of the TBA for bread in Sweden, aiming to provide in-depth information valuable as decision support for companies and policymakers. By quantifying the impact of transports related to the TBA, the present study addresses an identified research gap and a risk factor for food waste generation at the supplier-retailer interface. The assessment takes a consequential approach as it explores the consequences of a decision affecting the life cycle of a product system (Ekvall & Weidema 2004). The results can be of interest to researchers in the field of food waste and to companies in Sweden and abroad where a similar reverse supply chain is operated.

The functional unit (FU) is 1 kg of bread leaving the bakery. The geographical boundary for the assessment is Sweden, and the primary geographical reference is the city of Uppsala. The assessment only considers the three largest bakeries in Sweden, namely Pågen, Fazer, and Polarbröd.

The system boundary includes all mass flows from factory gate to grave, excluding the consumption stage; illustrated in a simplified form in *Figure 3*. This system diagram excludes the flow of packaging for simplification, but packaging is part of the reference flow; its consideration is explained further below. The waste treatment stage is calculated based on previous studies. This was most feasible due to the limited scope of this study and since the focus here lies on determining the changes in emissions stemming from changes in the way bread is traded, not its production and treatment.

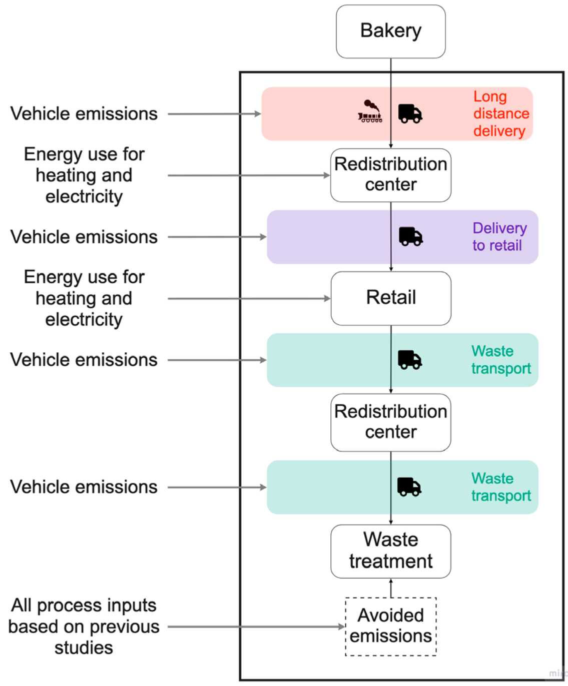

Figure 3: *General systems diagram for the assessed system*
The black line depicts the system boundary, the dashed line illustrates avoided emissions through system expansion, and the transport section are highlighted in colors, i.e., long-distance delivery [orange], delivery to retail [violet], waste transport [green].

This study focuses on bread sold under the take-back agreement, thus excluding store-baked bread and private label bread. All bread is assumed to be edible when discarded at the retail level. The study excludes energy use and emissions related to the construction, maintenance, and disposal of infrastructure (such as factories, power plants, and roads) and vehicles.

Multifunctionality was handled using system expansion, the suggested method for prospective LCA studies (Klöpffer et al. 2014). The scenarios are credited by accounting for the average emissions of the substituted products, i.e., through waste prevention and valorization. To achieve comparability, the reduction of bread waste was accounted for by the prevention of bread waste, assessed as a fraction of the baseline waste rate of 7,7% used in the scenario depicting the current TBA system.

It was necessary to combine multiple data sources to build the scenario models. Primary data was obtained from an internal, ongoing data collection via e-mail conversations and semi-formal interviews with retailers, bakeries, and other relevant industry stakeholders. This was combined with data from publicly available company information and reports, documents of public authorities, and scientific articles. Data for electricity and vehicles was collected from Ecoinvent 3.8, DEFRA (2021b) and the Network for Transport Measures (NTM) (n.d.).

3.2 Life cycle inventory (LCI) assessment

The second step in an LCA involves the compilation and quantification of all inputs and outputs along the life cycle of the analyzed system (Klöpffer et al. 2014). Two baseline scenarios were modeled for the assessment:

1. ***S*$_{TBA}$** is based on the conventional TBA system for bread in Sweden and depicts turnover time, bread return rate, and waste handling according to current practice.
2. ***S*$_{non\text{-}TBA}$** is a conceptual scenario in which bread is delivered without a TBA and which includes adapted turnover time, bread return rate, and waste handling.

The system operated by the three largest bread suppliers is mapped out both for ***S*$_{TBA}$** and ***S*$_{non\text{-}TBA}$** (*Figure 4)*. The scenarios were modeled based on best knowledge and available data. The differences in each company's operations were accounted for by using their market share, extrapolated to 100%.

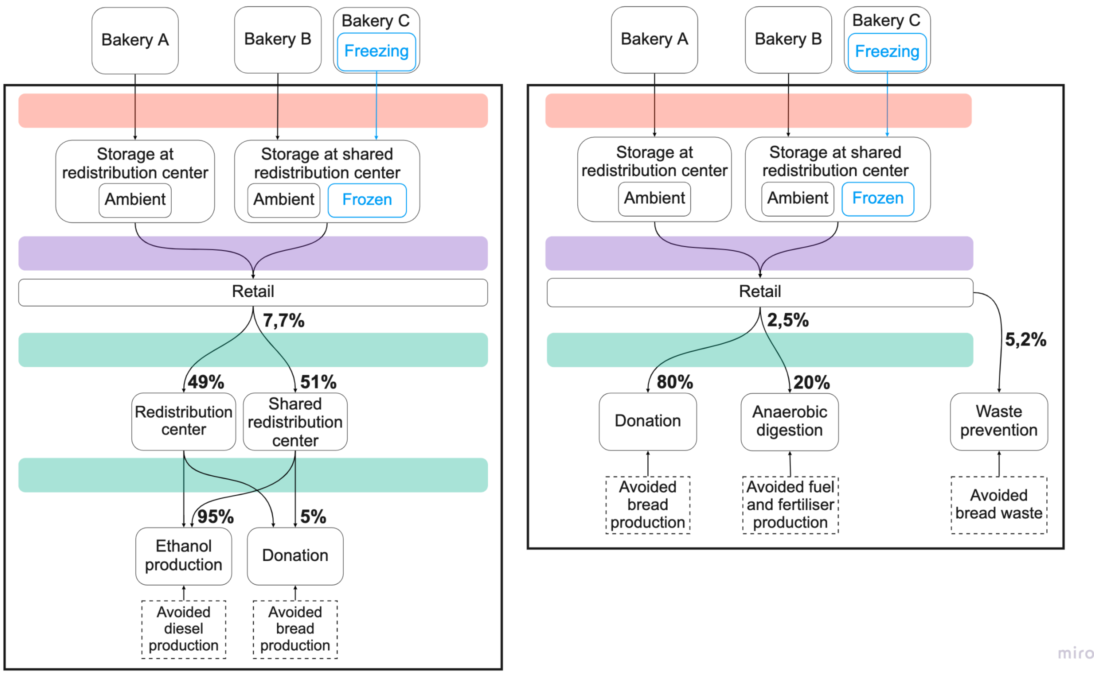

Figure 4: *Systems diagrams for **S**$_{TBA}$ and **S**$_{non\text{-}TBA}$*

Left-hand S$_{TBA}$; Right-hand: S$_{non\text{-}TBA}$. The black line depicts the system boundary, the dashed line illustrates avoided emissions through system expansion, and the transport section are highlighted in colors, i.e., long-distance delivery [orange], delivery to retail [violet], waste transport [green].

Long-distance delivery

The starting point of the system is the bakery, from where bread is transported to the local redistribution center, i.e., long-distance delivery (*Figure 4, [orange]).* This section was assumed to not be directly affected by changes related to the TBA[4]; $\boldsymbol{S}_{\boldsymbol{TBA}}$ and $\boldsymbol{S}_{\boldsymbol{non\text{-}TBA}}$ were therefore modeled identically.

The inputs for both scenarios are provided in *Table 2,* where transport distances are rounded, average values for each bread supplier, who all deploy several bakeries around Sweden. The complete calculations for all life cycle stages of $\boldsymbol{S}_{\boldsymbol{TBA}}$ and $\boldsymbol{S}_{\boldsymbol{non\text{-}TBA}}$ are provided in *Appendix 2.*

Table 2: *LCI for long-distance transport applicable to* $\boldsymbol{S}_{\boldsymbol{TBA}}$ *and* $\boldsymbol{S}_{\boldsymbol{non\text{-}TBA}}$*, expressed for 1kg of bread*

Input	Quantity	Unit	Source
Long-distance transport, bakery B, truck	116	km	Google Maps (Google n.d.)
Intermediate transport, bakery B, truck	6	km	
Long-distance transport, bakery B, train	571	km	
Long-distance transport, bakery C, truck, frozen	16	km	
Intermediate transport, bakery C, truck, frozen	4	km	
Long-distance transport, bakery C, train, frozen	647	km	
Long-distance transport, bakery A, truck	609	km	

Some bakeries deliver fresh bread, while other bakeries have opted for freezing the bread right after baking and letting it unfreeze on the way to the store (Company confidential 2021d). Some bakeries operate a supply chain consisting of a long-distance delivery via railway and a short-distance delivery via truck (Company confidential 2021d); others carry out all transport via lorry[5]. Where rail freight applied, railway transport was modeled for distances over 350km, trucks for distances below 350km and intermediate transport. Frozen transport was considered only for the bakery that freezes their bread after baking. For rail freight, the backhaul was excluded and assumed that trains transport other products back. One bakery employs a logistics company that owns both vehicles and redistribution centers, allowing the transport of other products, if not bread[6]; based on this, backhaul was excluded for road freight as well. The used emission factors either did not specify a load (Ecoinvent), or considered an average load (DEFRA), which is not specified but can be estimated ~16t, based on different sources (Network for Transport Measures n.d.; Valsasina n.d.); both options were assumed sufficient for this stage. Diesel or petrol fuel was modeled for all road freight, which portrays a conservative but realistic picture of the Swedish transport sector, which relies to 65% on diesel and gasoline (Swedish Energy Agency 2021). Beyond that, this was

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[4] Bakery B, pers. comm. 2021-11-04; Bakery A, pers. comm. 2022-03-04.

[5] Bakery A, pers. comm. 2022-03-04.

[6] Bakery A, pers. comm. 2022-03-04.

a necessary simplification based on the primary dataset used for vehicle emissions. The selection of vehicles, also based on estimated load, as well as the determination of emission factors, is further explained in *Appendix 1*.

Storage at the redistribution center

Storage at the redistribution center was modeled identical in both baseline scenarios (*Table 3*). Frozen storage was assumed according to the market share of the bakery that opts for freezing their bread. Energy consumption was calculated based on energy consumption data of one Swedish retailer (Company confidential 2021b). The proportional energy use for electricity and heating, and the fraction of electricity used for refrigeration, were derived from a study on food retail energy usage (Swedish Energy Agency 2010) and applied to the retailer's total energy use (see *Appendix 1.4)*. Those values were cross-checked and validated with a report by DEFRA (2008). A district heating system, the most common for commercial facilities in Sweden (Swedish Energy Agency 2010) and wood chips as an energy source, the most dominant source for district heating in Sweden (Swedish Energy Agency 2015) were assumed. The emissions were calculated based on the Swedish electricity mix.

Table 3: *LCI for storage at redistribution center applicable to* S_{TBA} *and* $S_{non\text{-}TBA}$*, expressed for 1kg of bread*

Input	Quantity	Unit	Source
Electricity for ambient storage	0,0025	kWh	DEFRA 2008; Swedish Energy Agency 2010; Company confidential 2021b
Electricity for frozen storage	0,0012	kWh	
Heating	0,0014	kWh	

Delivery to retail

The local delivery of bread from redistribution center to retail (*Figure 4, [violet])* was modeled with an exemplary route through Uppsala, as depicted in *Figure 5* and further explained with the respective locations in *Appendix 1.3*. This transport section is operated in a circular mode, i.e., delivering bread and picking up unsold bread simultaneously[7]. The driver has responsibility for transport and acts as a salesperson, forecasting and negotiating bread quantity and assortment for a specific zone (Eriksson et al. 2017). Salespeople serve around 3-5 stores in one go, assuring an adequate workload and salary[8]. Ismatov (2015) found that big stores can get up to two deliveries per day, six times a week, while other stores receive deliveries only once a week. One bakery stressed that they use small trucks as it is easier to acquire personnel when no driver's license for large trucks is required[9].

[7] Bakery A, pers. comm. 2022-03-04.
[8] Bakery A, pers. comm. 2022-03-04.
[9] Bakery A, pers. comm. 2022-03-04.

Both truck size and this bakery's determination to deliver fresh bread require them to deliver twice per day to most stores (ibid.).

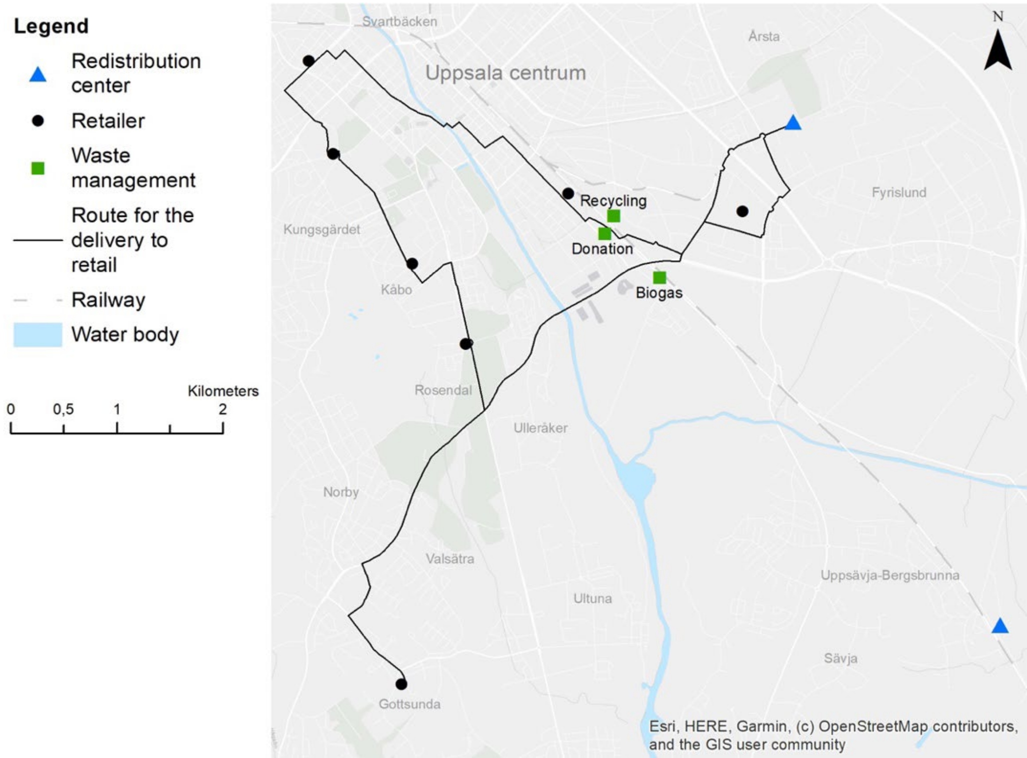

Figure 5: *Exemplary bread delivery route modeled for Uppsala (Uppsala Kommun 2018; Swedish Land Survey 2021; Google n.d.)*

To account for the company differences, the use of a smaller truck and a route with fewer stops along the delivery route were modeled for one bakery, and the use of a larger truck and more stops for the other bakeries. *Table 4* shows the input data for this stage; notably, it does not account for the difference in vehicles, which is further explained in *Appendix 1.1* and in *Appendix 2*.

Table 4: *LCI for delivery to retail applicable to* $\boldsymbol{S_{TBA}}$ *and* $\boldsymbol{S_{non\text{-}TBA}}$*, expressed for 1kg of bread*

Input	Quantity	Unit	Source
Delivery to retail, route type A	25	km	Google Maps (Google n.d.)
Delivery to retail, route type B	26	km	

Route types applicable to different bakeries, anonymized

The differences in delivery frequency per day and week were not included in the model due to uncertainty and variability of this number. The delivery to retail also includes the pick-up of return bread in $\boldsymbol{S_{non\text{-}TBA}}$, so the transport back to the redistribution center, as part of the waste transport, was included in this section.

The frozen bread is intended to de-freeze on the way to the store[10], so no frozen transport was assumed. A load of 50% or an average load was assumed, depending on what the dataset provided, as further explained *Appendix 1.2*.

Retail stage

Energy consumption at the retail stage is calculated similar as for the redistribution center. At the retail stage, 0,015m^2 of storage space is assumed for the FU. Turnover time was estimated based on data provided by stakeholders and previous studies on shelf life and the time frames for the removal of bread within the TBA (Ismatov 2015; Company confidential 2021e)[11]. Based on this, an average retail storage time of 4 days was assumed for $\boldsymbol{S_{TBA}}$. In $\boldsymbol{S_{non\text{-}TBA}}$, bread was assumed to be kept on the shelves for its entire shelf life due to waste prevention actions such as discounts. The ambient storage of bread is the standard; however, frozen storage is assumed in $\boldsymbol{S_{non\text{-}TBA}}$ based on the suggestion of an industry stakeholder[12] to use frozen storage to coordinate delivered volume, shelf life, and demand. Thus, $\boldsymbol{S_{non\text{-}TBA}}$ assumes 9 days of ambient storage; however, for 10% of bread delivered to defrost, 4 days of frozen storage and 5 days of ambient storage are assumed. The retail stage and all subsequent stages were modeled individually for the baseline scenarios; the respective LCI are provided in *Table 5* for $\boldsymbol{S_{TBA}}$ and in *Table 6* for $\boldsymbol{S_{non\text{-}TBA}}$, with the respective stages explained below.

***Table 5**: LCI for retail, waste transport, and treatment stages, applicable to $\boldsymbol{S_{TBA}}$, expressed for 1kg of bread*

Input	Quantity	Unit	Source
Storage, retail			
Electricity for ambient storage	0,0253	kWh	(DEFRA 2008; Swedish Energy Agency 2010; Company confidential 2021b)
Heating	0,0111	kWh	
Transport, waste[*]			
Bakery A	239	km	Google Maps (Google n.d.)
Bakery B	230	km	
Waste treatment[**]			
Bread waste			
Ethanol production	0,0732	kg	(Company confidential 2021c; d; e)
Donation	0,0039	kg	
Packaging waste[***]			
Incineration & Recycling	0,0015	kg	Own estimation based on Brancoli et al. 2020; Bakery A (2022), pers. comm. 2022-03-04

[10] Bakery C, pers. comm. 2022-01-28.

[11] Bakery A, pers. comm. 2022-03-04.

[12] Retailer A, pers comm. 2021-11-30; Bakery B, pers. comm. 2022-03-23.

[*] *Different vehicles depending on waste treatment type; average distance, for exact routes and distances, see Appendix 2*
[**]*Applied to a wasted fraction of 7,7% / 0,0785 kg incl. packaging*
[***]*For further details on packaging waste treatment, see Appendix 1.5*

Table 6*: LCI for retail, waste transport, and treatment stages, applicable to* $S_{non\text{-}TBA}$*, expressed for 1kg of bread*

Input	Quantity	Unit	Source
Storage, retail			
Electricity for ambient storage	0,0545	kWh	(DEFRA 2008; Swedish Energy Agency 2010; Company confidential 2021b)
Electricity for frozen storage	0,0046	kWh	
Heating	0,0251	kWh	
Transport, waste[*]			
Waste transport[**]	11	km	Google Maps (Google n.d.)
Waste prevention			
Bread waste prevention	0,0530	kg	Own estimation based on Bakery A (2022), pers. comm. 2022-03-04
Waste treatment[a]			
Bread waste			
Anaerobic digestion	0,0050	kg	(Brancoli et al. 2020; Company confidential 2021a)
Donation	0,0200	kg	(Company confidential 2021b; f)
Packaging waste[***]			
Incineration & Recycling	0,0015	kg	Own estimation based on Brancoli et al. 2020

[*]*Different vehicles depending on waste treatment type; average distance, for exact routes and distances for each bakery, see Appendix 2*
[**] *Applied to waste fraction of 2,5% / 0,0255 kg incl. packaging*
[***]*For further details on packaging waste treatment, see Appendix 1.5*

Waste transport

Waste transport depends on who handles the surplus bread, as this affects its destination after it leaves the shelves. In S_{TBA}, bread that has been transported back to the redistribution center is further transported for waste treatment (*Figure 4, left-hand [green]).* The storage of bread after the retail stage is outside of the scope of this study; since this commonly includes storage in a waste container outside of the redistribution center's building[13], it does not require any additional inputs. In contrast, $S_{non\text{-}TBA}$, bread is discarded at the retail stage and handled from there (*Figure 4, right-hand [green]).* The transport to waste treatment was calculated using the respective average distance to the treatment facility and the appropriate vehicle *(Appendix 1.1).*

[13] Bakery A, pers. comm. 2022-03-04.

Waste treatment

The bread return rate for S_{TBA} was assumed to be 7,7%, based on information provided by an industry stakeholder[14]. For $S_{non\text{-}TBA}$, a waste rate of 2,5% was assumed, this being a conservative estimation based on stakeholder opinion[15]. One bakery[16] directs all their bread waste occurring in Uppsala to ethanol production, the other bakeries do not mention other treatment types. Nevertheless, a small fraction of bread most likely is donated, as local charity organizations report that bread from those bakeries ends up in their food bags[17]. It was therefore assumed that in S_{TBA}, 95% of return bread is directed to bioethanol production and 5% is donated (*Figure 4, left-hand*). Bread is picked up from the redistribution centers in both use cases. For $S_{non\text{-}TBA}$ it was necessary to consider the way retailers would dispose of bread without a TBA in place. Store-baked bread and private label bread are usually disposed of with other organic waste and directed to anaerobic digestion (Company confidential 2021f). Beyond that, food donations are common among Swedish supermarkets, or food is sold via applications such as Too Good To Go (Company confidential 2021b; f). It was thus assumed that 80% of bread is donated to Matcentralen (Uppsala Stadsmission n.d.) and 20% is directed to anaerobic digestion in Uppsala (Uppsala Vatten n.d.) (*Figure 4, right-hand*). The climate benefit of valorizing bread waste was assessed by Brancoli et al. (2020); the results are provided in *Table 7* and were used to account for the avoided emissions.

Table 7: *Emission savings potential of bread waste valorization types (Brancoli et al. 2020)*

Valorization types	**kg CO_2e / 1 kg bread**
Prevention	-0,66
Donation[18]	-0,37
Ethanol production	-0,56
Feed production	-0,53
Incineration	-0,08
Anaerobic digestion	-0,02

Packaging

The packaging surrounding pre-packaged bread was assumed to consist of a low density polyethylene bag as outlined in previous studies (Williams & Wikström 2011). An average weight of 10 g of packaging per 500 g bread loaf was assumed based on the weighting of a product. The plastic clip was excluded for simplification and because observations showed a weight below what a common

[14] Bread industry stakeholder A, pers. comm. 2021-10-22.

[15] Retailer A, pers. comm. 2022-02-08.

[16] Bakery A, pers. comm. 2022-03-04.

[17] Niina Sundin, SLU, pers. comm. 2022-03-07.

[18] Pedro Brancoli, pers. comm. 2022-03-21.

kitchen scale can measure. Thus, the functional unit of 1 kg of bread requires an additional input of 20 g of packaging; the calculations are therefore based on a reference flow of 1,02 kg. Packaging separation and subsequent recycling in Uppsala (Returpappercentralen Uppsala n.d.) was modeled for $\boldsymbol{S_{TBA}}$, where the bakery handles bread waste. $\boldsymbol{S_{non\text{-}TBA}}$ assumed that packaging is not separated because supermarkets do not commonly invest in additional processing (Brancoli et al. 2020). Further details on packaging calculation are provided in *Appendix 1.5.*

Scenario alterations

The baseline scenarios, $\boldsymbol{S_{TBA}}$ and $\boldsymbol{S2_{non\text{-}TBA}}$, were altered to simulate additional, compromising scenarios:

3. $\boldsymbol{S_{coop}}$ was modeled to capture the potential impact of a TBA still in place, but with an increased commitment of all actors to reduce bread waste by cooperation and data sharing between retailers and bread suppliers.
4. $\boldsymbol{S_{co\text{-}log}}$ was modeled to simulate an integration of logistics as it was mentioned by stakeholders[19] as a possibility to improve the current system, and because some cooperation of this type already takes place (Company confidential 2021d).

A longer retail storage than in $\boldsymbol{S_{TBA}}$ was assumed for $\boldsymbol{S_{coop}}$, but bread is still returned before its best-before date, as suppliers don't want to blemish their reputation by selling old bread[20]. Thus, $\boldsymbol{S_{coop}}$ assumed 7 days of ambient storage; however, for 10% of bread delivered to defrost, 3 days of frozen storage and 4 days of ambient storage were assumed. A return rate of 4%, and thus the prevention of 3,7% of bread waste was assumed. This aims to show a middle-ground between $\boldsymbol{S_{TBA}}$ and $\boldsymbol{S2_{non\text{-}TBA}}$ and to account for the minimum necessary waste rate of 4% to avoid empty shelves, as suggested by several stakeholders[21]. Based on this rate, waste treatment was modeled as in $\boldsymbol{S_{TBA}}$.

$\boldsymbol{S_{co\text{-}log}}$ assumed cooperation for the delivery to retail and waste transport stages. For the delivery to retail, a larger vehicle must be used to account for the larger volume of bread that must be transported. All waste was assumed to be handled from one redistribution center altogether. The distance for the donations pick-up was doubled to account for the fact that charity organizations presumably must go twice and do not own larger vehicles. Similarly, the truck remained unchanged for the waste pick-up by the ethanol producers, as they would presumably pick-up more often instead of employing a different truck. For the long-distance delivery, no changes

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[19] Logistics company, pers. comm. 2022-01-21; Retailer A, pers. comm. 2022-02-08.
[20] Bakery A, pers. comm. 2022-03-04.
[21] Bakery A, pers. comm. 2022-03-04; Bakery D, pers. comm. 2021-12-16

were modelled because cooperation was not assumed for this part, as the bakery locations are static. Turnover time, return rate, and waste treatment were modeled as in $\boldsymbol{S_{TBA}}$.

3.3 Life cycle impact assessment

This LCA included climate change as a single impact category, calculated using the IPCC CO_2 equivalent factors for Global Warming Potential (GWP_{100}). GWP_{100} includes CO_2, N_2O, and CH_4 emissions, where the global warming potential of N_2O and CH_4 emissions are set in relation to CO_2. GWP_{100} considers a middle-ground time horizon of 100-years and has a level of evidence accepted internationally (Myhre et al. 2013; Huijbregts et al. 2017). Global warming, and climate change as the consequence, are among the most urgent global challenges (Bebkiewicz et al. 2020) and global warming potential was found to be one of the most critical impact categories when researching transport; next to smog, human toxicity, et cetera (Jorgensen et al. 1996). This makes it valid to concentrate on GWP_{100} only and investigate this impact category in greater detail.

3.3.1 Sensitivity analysis

The results of $\boldsymbol{S_{TBA}}$ and $\boldsymbol{S_{non\text{-}TBA}}$ were tested for their sensitivity to distance changes in each transport section. It was also tested how far the waste can be transported in $\boldsymbol{S_{TBA}}$ until it is outperformed by $\boldsymbol{S_{non\text{-}TBA}}$. As explained above, $\boldsymbol{S_{non}}$-TBA does not assume the separation of packaging, however, as it could be beneficial to do so, thus the separation of packaging by the retailers in a non-TBA system was modeled in $\boldsymbol{S_{pack}}$. Furthermore, the usage of bread for pig feed in a non-TBA system was assessed in $\boldsymbol{S_{pig}}$, to evaluate an alternative, relatively more beneficial waste treatment options than what was assumed in $\boldsymbol{S_{non\text{-}TBA}}$. In $\boldsymbol{S_{pig}}$ 20% of bread is assumed to be directed to pig feed and 80% to be donated. An average transport distance of 25 km to pig farms was calculated based on a Google Maps search for pig farms around Uppsala within a radius of maximum 50 km. Lastly, $\boldsymbol{S_{coop}}$ and $\boldsymbol{S_{co\text{-}log}}$ were modeled in combination as $\boldsymbol{S_{max.collab}}$, to test a maximum level of cooperation while keeping the TBA system in place.

Data uncertainty

The robustness of LCA results depends on the quality of data inputs (Jensen & Arlbjørn 2014), and there were several datasets to choose emission factors from in the case of this assessment. Thus, a data uncertainty analysis was conducted for the transport sections in $\boldsymbol{S_{TBA}}$, $\boldsymbol{S_{non\text{-}TBA}}$, $\boldsymbol{S_{coop}}$, and $\boldsymbol{S_{co\text{-}log}}$, to achieve more transparency for the respective emissions. NTMCalc Advanced 4.0 was used to derive emission factors for all vehicles. *Table 8* compares these emission factors to the baseline dataset. In a second step, the effect of the driving environment was tested by adjusting the NTMCalc values for each section according to the most prevalent road used in each of them.

Table 8: *Comparison of vehicle emission factors from different data sets*

Baseline data		**NTMCalc Advanced 4.0**	
Vehicle	**kg CO_2e/ ton.km**	**Vehicle**	**kg CO_2e/ ton.km**
Freight, train	0,015$_a$	Mix, electric and diesel train	0,008
Freight, train with reefer, freezing	0,055$_c$	~	~
HGV, Articulated, >33t	0,080$_b$	Truck with trailer, 34-40t	0,078
HGV, Rigid, >17t (100% utilization)	0,120$_b$	Rigid truck, 20-26t (100% load)	0,065
HGV, Rigid, >17t (average utilization)	0,181$_b$	Rigid truck, 20-26t (66% load)	0,086
HGV, Rigid, >7,5-17t	0,340$_b$	Rigid truck, 7,5-12t	0,166
Lorry with refrigeration machine, freezing, 16-32t/>32t	0,461$_d$	~	~
HGV, Rigid, >3,5-7,5t	0,451$_b$	Rigid truck, <7,5t	0,195
Van, average, up to 3,5t, load capacity 1t	0,603$_b$	Van, load capacity 1,5t	0,473
Van, Class I, up to 1,305t	0,815$_b$	Pick-up, load capacity 0,6t	1,513

HGV = Heavy Goods Vehicles; When NTMCalc didn't offer values (~), the baseline values were used. The NTMCalc values in this table refer to average road conditions.
$_a$Own calculation, see Appendix 1.1; $_b$DEFRA 2021b; $_c$(Ecoinvent system generated n.d.b); $_d$(Ecoinvent system generated n.d.a)

Location uncertainties

One industry stakeholder explained that Uppsala and Stockholm are the only two regions in Sweden where one of the bakeries directs bread waste to ethanol production; for all other regions in Sweden, bread is transported back to the bakeries and re-valorized from there[22]. This implies that waste transport and treatment modeled for Uppsala depicts a special case that lacks representativeness for Sweden. Therefore, the model for S_{TBA} and $S_{non\text{-}TBA}$ was applied to Gävle, Göteborg, Jönköping, Helsingborg, Örebro, to evaluate the sensitivity of the results based on location, and to test if similar conclusions can be drawn for other regions in Sweden. *Figure 6* provides an overview of all cities covered in the assessment, as well as the bakeries of all three large Swedish bread suppliers and the two bioethanol plants, of which only the one in southern Sweden was included in the assessment.

[22] Bakery A, pers. comm. 2022-03-04.

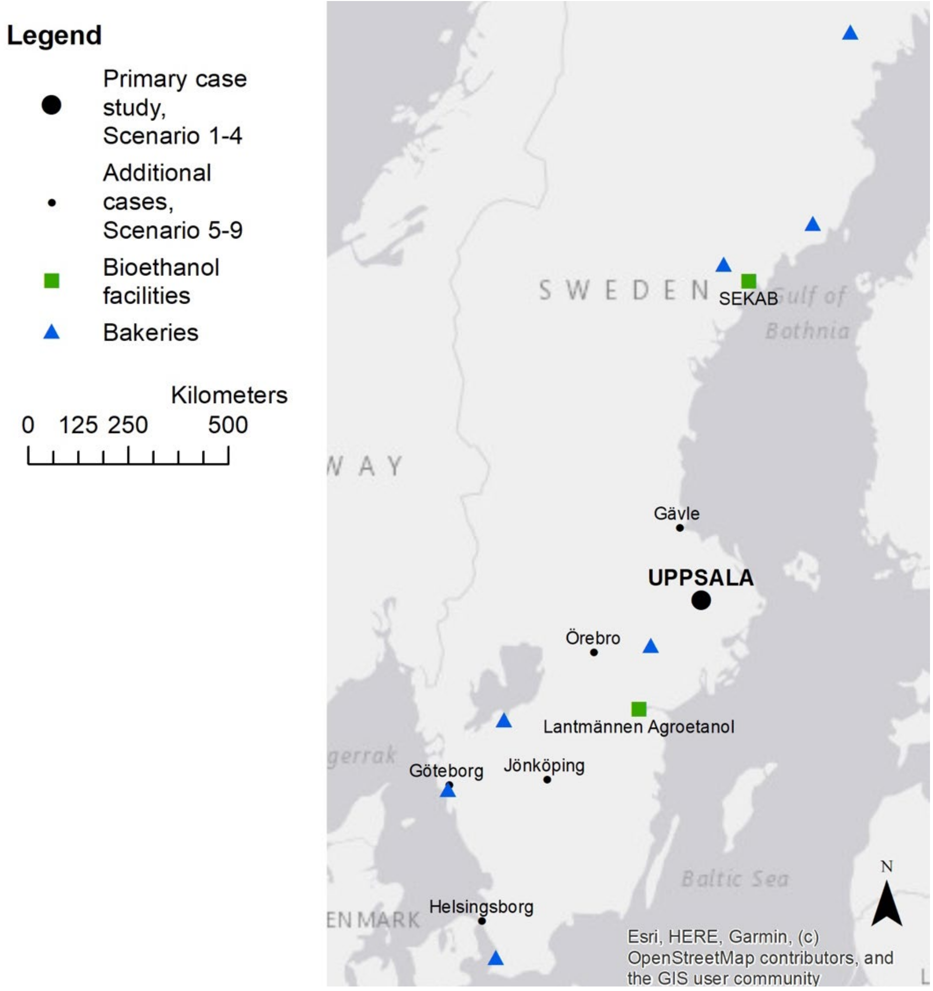

Figure 6: *Overview map of the Swedish bread industry*
(Google n.d.)
Scenario 1-4 referring to S_{TBA}, $S_{non\text{-}TBA}$, S_{coop}, $S_{co\text{-}log}$; *S5-9 referring to scenarios in additional cities*

Some parts of the system, i.e., storage emission and waste treatment in $S_{non\text{-}TBA}$, were modeled using the same inputs as in the baseline scenarios. *Figure 7* shows the systems diagram applicable for the TBA-scenarios; for the non-TBA-scenarios *Figure 4 (right-hand)* is applicable. Some aspects were adjusted for each city, such as routes, distances, and vehicles for all delivery steps. The most important difference, at the waste transport and treatment stages, is highlighted grey in *Figure 7*. In the TBA scenarios, of the bread transported back to the bakery, 35% and 65% was assumed to be used as pig feed and to be donated, respectively. Here it is important to mention that the bakeries to which this applies are in Göteborg and Malmö, of which the one that is closer to the assessed city was considered, or the average distance was taken, if both were at reasonable distance. In the case of Göteborg, some transport steps were excluded as the redistribution center was assumed to be located at the bakery.

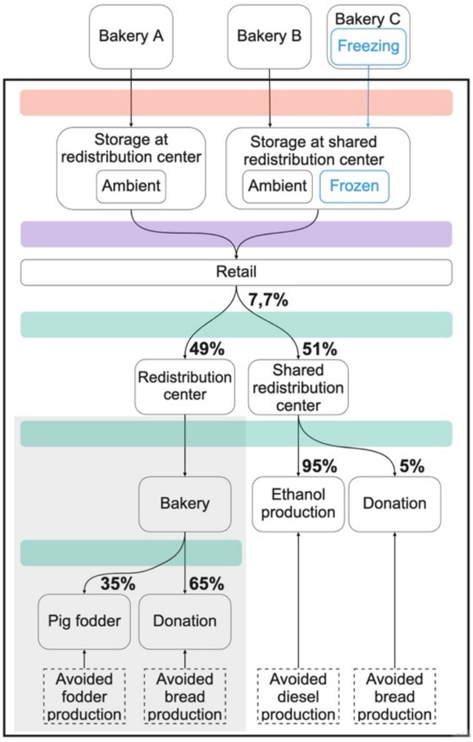

Figure 7: *Systems diagram for the TBA-scenario in additional cities*

In a second step, a scenario was modeled in which all bakeries direct their return bread to ethanol production, as it is commonly done in Uppsala. This was calculated for Gävle ($\boldsymbol{S}_{\boldsymbol{TBA+E,\ Gävle}}$) and Göteborg ($\boldsymbol{S}_{\boldsymbol{TBA+E,\ Göteborg}}$), which were deemed interesting to assess as one of them hosts a bakery and another one is relatively further away from the bioethanol plant than Uppsala.

Appendix 3 provides a summary of all modeled scenarios.

4. Results

The results show that a shift from the conventional $\boldsymbol{S_{TBA}}$ to a conceptual $\boldsymbol{S_{non\text{-}TBA}}$ scenario increases the climate impact in the case of Uppsala (*Figure 8*). With 5% or 0,7 g CO_2e per functional unit, the difference between the two baseline scenarios is marginal. Long-distance delivery and waste treatment were identified as impact hotspots in both $\boldsymbol{S_{TBA}}$ and $\boldsymbol{S_{non\text{-}TBA}}$, with 40% of the emissions stemming from long-distance delivery alone. Waste treatment has a higher emission savings potential in $\boldsymbol{S_{TBA}}$. The delivery to retail stage has the third-largest impact; the retail storage and waste transport stages have relatively lower impacts. Notably, the GWP_{100} of waste transport is 90% higher for $\boldsymbol{S_{TBA}}$, and that for retail storage is higher for $\boldsymbol{S_{non\text{-}TBA}}$.

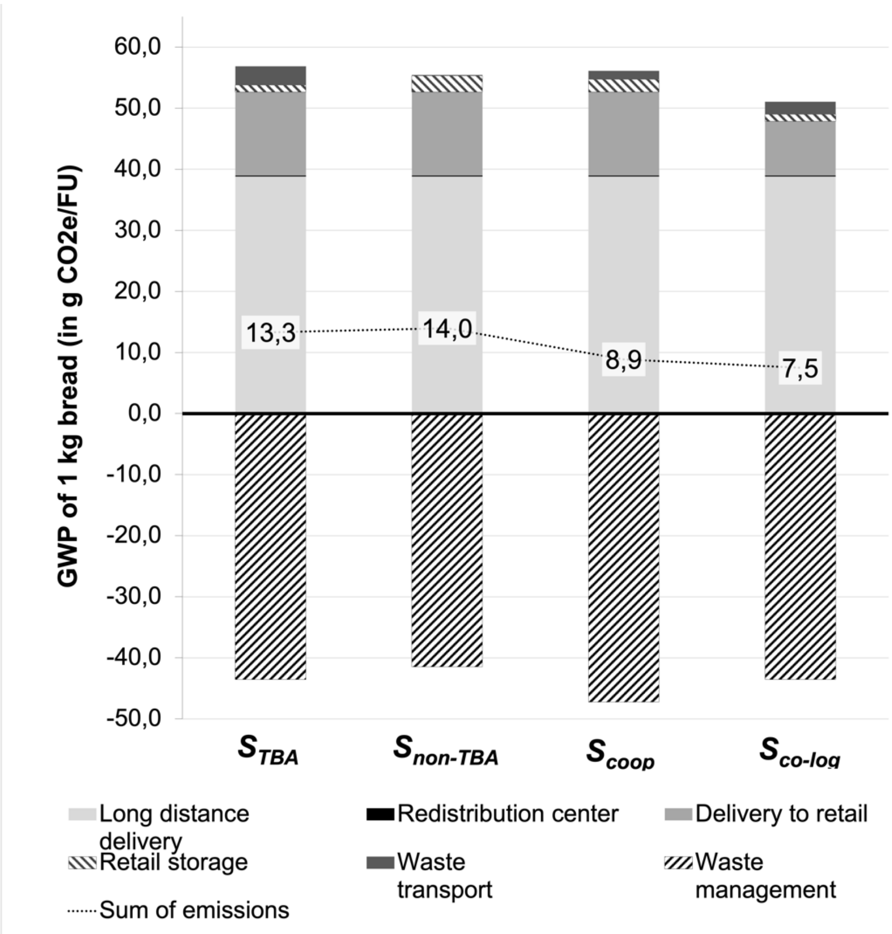

***Figure 8**: GWP_{100} of baseline scenarios and alterations, expressed per functional unit*

Packaging treatment makes up 3% and 1,2% of emission reduction potential of waste treatment in ***S***S_{TBA} and $S_{non\text{-}TBA}$, respectively. Both S_{coop} (joint waste prevention) and $S_{co\text{-}log}$ (logistics integration) have a lower GWP_{100} than S_{TBA} and $S_{non\text{-}TBA}$ *(Figure 8)*. $S_{co\text{-}log}$ has a considerably lower impact at the delivery to retail stage, while S_{coop} results in the highest emission savings potential from waste treatment.

$S_{max.collab}$ (combination of S_{coop} and $S_{co\text{-}log}$) (*Figure 9*), has a lower GWP_{100} than all before mentioned scenarios, with a reduction of GWP_{100} by 69% compared to S_{TBA}. Packaging separation at retail (S_{pack}) has a negligible effect on the result (*Figure 9*). If retailers used bread for pig feed (S_{pig}), the emission reduction potential could be increased by 8%, causing an overall benefit of $S_{non\text{-}TBA}$ over S_{TBA}.

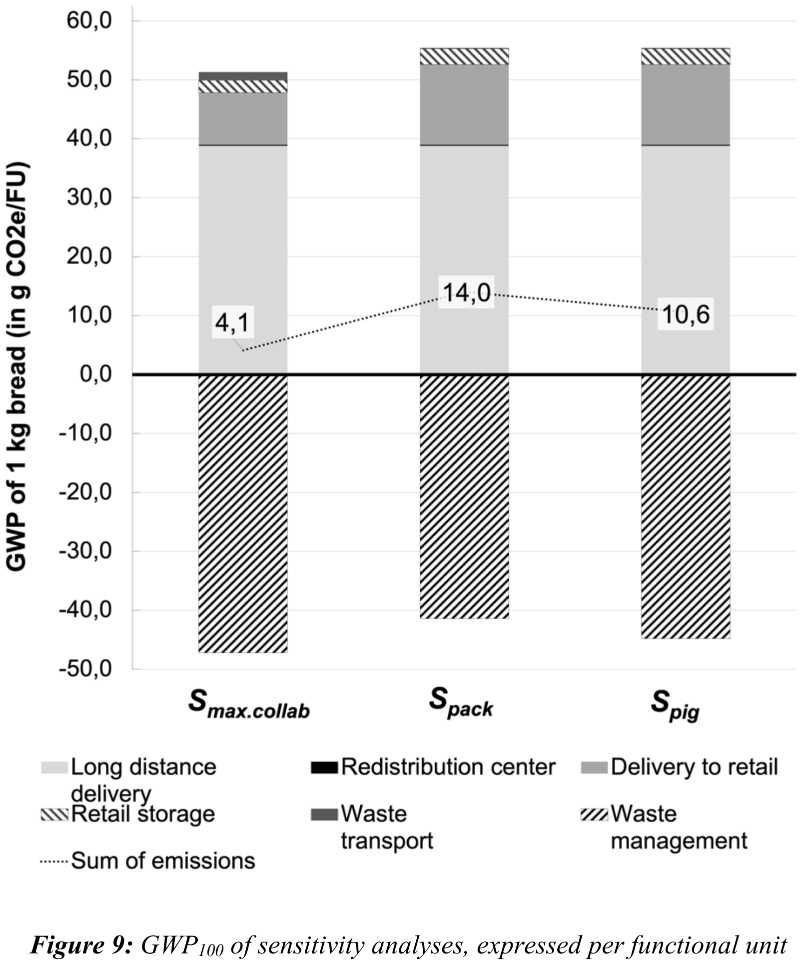

Figure 9: *GWP_{100} of sensitivity analyses, expressed per functional unit*

The sensitivity test identified a threshold above which $S_{non\text{-}TBA}$ outperforms S_{TBA}, which lies at approximately 285 km, taking the results for Uppsala as a reference. Above this distance, the benefits of ethanol production from bread waste are outweighed by the necessary transport. *Figure 10* illustrates which cities are within this threshold (*Ethanol threshold area 1*).

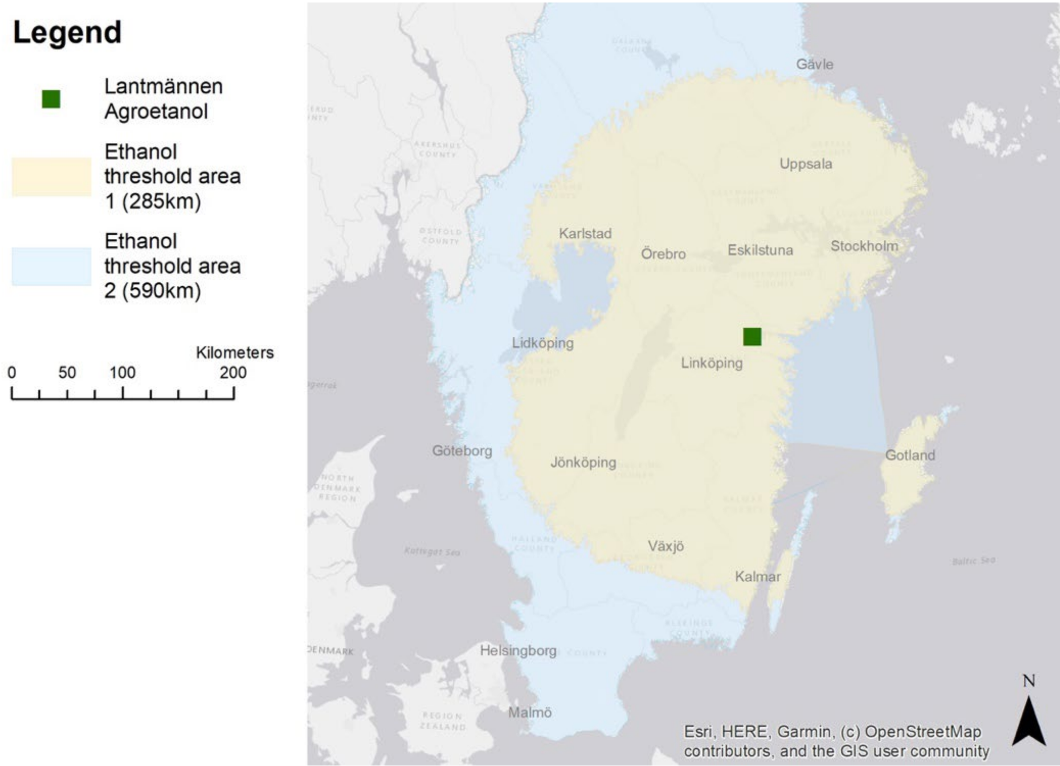

Figure 10: *Illustration of the ethanol threshold*
Ethanol threshold area 1 referring to primary dataset (DEFRA, Ecoinvent 3.8); Ethanol threshold area 2 referring to NTMCalc dataset (Swedish Land Survey 2020; Google n.d.).

The data uncertainty test for transport emissions, using the NTMCalc dataset, resulted in a lower GWP_{100} for all assessed scenarios (***S_{TBA}*, *$S_{non\text{-}TBA}$*, *S_{coop}*, *$S_{co\text{-}log}$***) (*Figure 11*). That is, on average, 66% and 92% lower, with the average and specific road factors, respectively. Nevertheless, the results confirm the order of preferability for the scenarios.

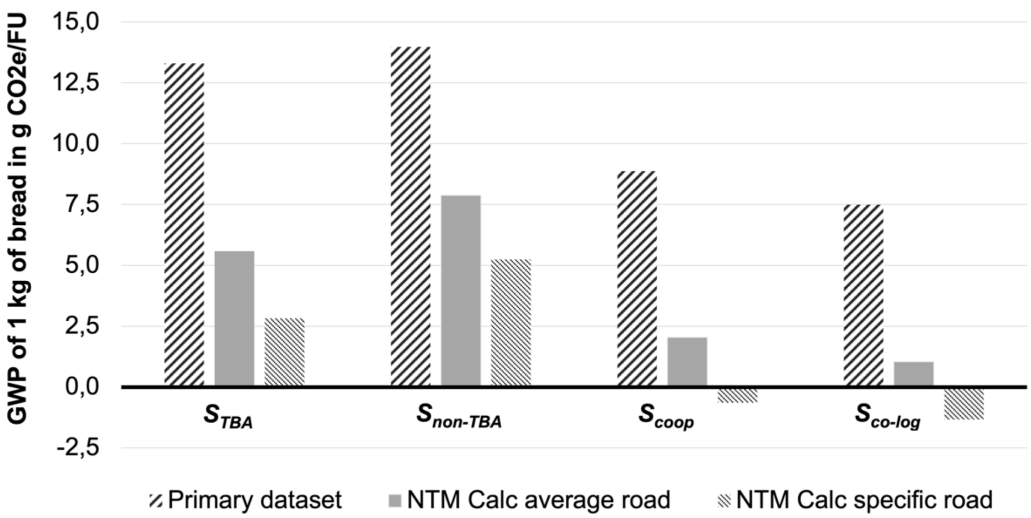

Figure 11: *GWP100 found in data uncertainty test*

The difference between the scenarios is relatively more pronounced with the NTMCalc dataset, for instance being 41% (average road) and 85% (specific road) between $\boldsymbol{S_{TBA}}$ and $\boldsymbol{S_{non\text{-}TBA}}$. With the NTMCalc factors, the ethanol threshold lies a 590 km, as depicted in *Figure 10* (*Ethanol threshold area 2*).

In all additionally assessed cities, the non-TBA scenarios outperform the TBA scenarios, and the same impact hotspots as in Uppsala were identified (*Figure 12*). All non-TBA scenarios result in considerably lower emissions, 50% lower on average, except for Helsingborg, where the difference is less pronounced. $\boldsymbol{S_{TBA+E,\ Gävle}}$ and $\boldsymbol{S_{TBA+E,\ Göteborg}}$ have a lower GWP_{100} than the standard TBA scenarios in Gävle and Göteborg. While $\boldsymbol{S_{TBA+E,\ Göteborg}}$ outperforms $\boldsymbol{S_{non\text{-}TBA,\ Göteborg}}$, in Gävle the non-TBA scenario remains the most beneficial.

The average GWP_{100} for all cities, including Uppsala, is 14,5 g CO_2e for the TBA scenarios and 10,4 g CO_2e for the non-TBA scenarios, which is a decrease by 28%.

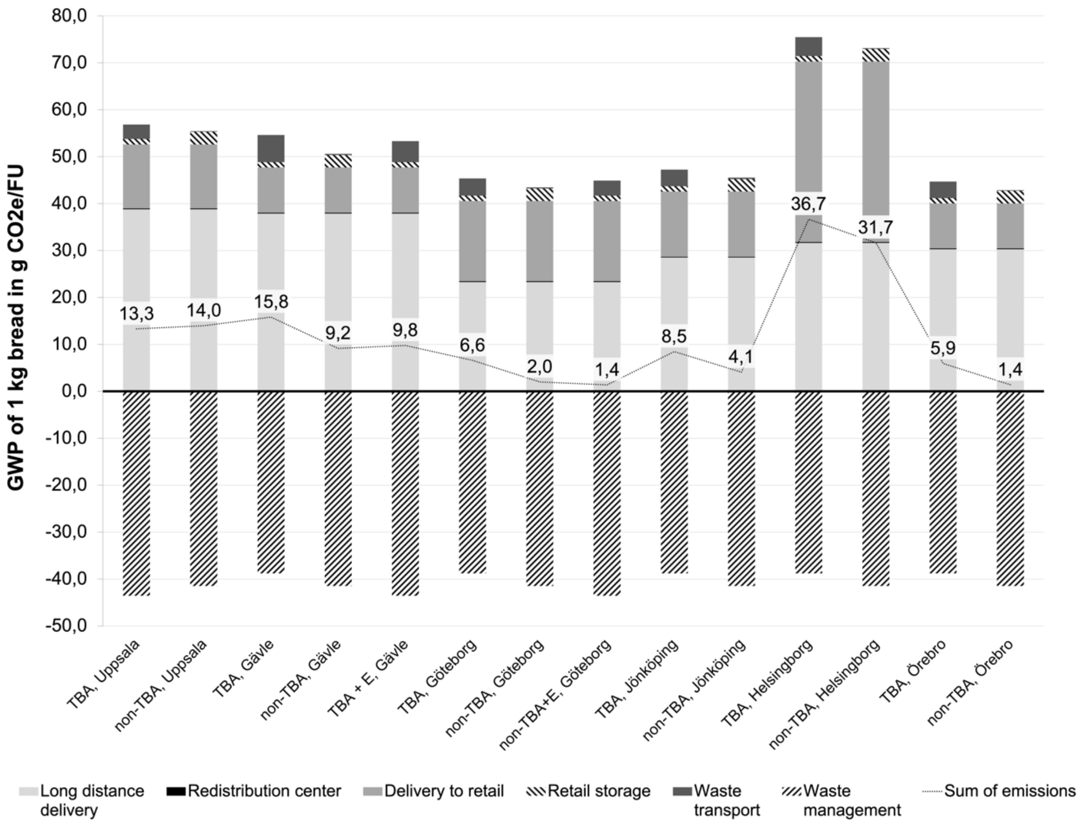

Figure 12: *GWP100 of the additionally assessed cities, expressed per functional unit*

5. Discussion

This section analyzes and discusses the results and contextualizes them with other studies. A validation of the results based on previous research is provided, and this study's limitations are outlined.

5.1 The climate impact of bread transport

Long-distance transport was identified as one of the most important climate impact hotspots, generating almost 40% of emissions in the case of Uppsala. This raises questions about whether the polarized debate on the TBA itself is reasonable and whether food transport chains need a general reformation. The transport sector is responsible for about one-third of Sweden's emissions, of which 90% come from road transport (Xylia & Olsson 2021), so a comprehensive evaluation the climate impact of bread wastage must address transport emissions. By excluding the previously identified impact hotspots for bread, namely the cultivation of raw materials, processing, and consumption stages, this study allowed for an in-depth evaluation of the transport contribution to climate impact. Transport emissions are expected to increase continuously (Liimatainen et al. 2014a), and the Swedish transport sector remains dependent on fossil fuels, relying 65% on petroleum products (Swedish Energy Agency 2021). The immense impact of long-distance transport points to the importance of local food production and, from a consumer perspective, choosing food according to local seasonality. Nevertheless, transport is a necessary step in the value chain of almost any food product (Wakeland et al. 2012), and even bread often requires sourcing raw material over large distances, as shown by Espinoza-Orias et al. (2011). The impact hotspot identified in this thesis must therefore be set in context. An assessment of larger scope could be of benefit; however, these stages are, if at all, only indirectly affected by the TBA; omitting it in this study can be considered reasonable.

The transport sector is arguably difficult to decarbonize due to its high dependency on fossil fuels (European Environment Agency 2021). However, there are several options to increase the energy efficiency of road freight transport beyond the widely implemented, simple and inexpensive options, such as choosing the truck type in line with the cargo (Liimatainen et al. 2014b). Part of the long-distance delivery in this study was modeled fully via truck, thus offering emission reduction potential through increased rail freight. Notably, the assumption that all long-distance transports below 350km are operated via truck affects the results and could also fail to meet the companies' actual operations. If rail freight had been assumed for all long-distance transport, irrespective of the distance, the climate impact of this stage would be 59% lower, in the case of Uppsala. The relatively larger impact of long-distance delivery in Gävle and Uppsala compared to the other cities (*Figure 12*) is most likely also connected to the type of vehicle used. Emissions increase the further away a city is located from those bakeries that solely use trucks for this transport section, which is the case for Uppsala and even more Gävle. This confirms the benefit of increased rail freight. In line with this finding, Liang et al. (2016) found that multi-modal approaches including rail transport can improve transport energy efficiency. However, to prove this feasible, cost-benefit and time aspects must also be assessed to determine whether they might outweigh environmental benefits.

$S_{co\text{-}log}$ and ***$S_{max.collab}$*** show that the integration of logistic operations, using larger and fewer vehicles, reduces transport emissions. The benefit of these scenarios over ***S_{TBA}***, ***$S_{non\text{-}TBA}$***, and ***S_{coop}*** is mainly due to the delivery to retail and waste transport stages, as those were assumed to be operated collaboratively. The environmental benefit of collaborative logistics, which also results in efficiency improvements, was also found by Eriksson (2015a) and Croci et al. (2021). However, logistics cooperation might not be entirely realistic, considering limited acceptability by industry stakeholders and the feasibility of integrating complex logistics operations, particularly for a commonly freshly consumed product, as also mentioned by stakeholders[23]. Innovations of a collaborative nature must also consider a company's priorities, as optimizing cost, time, and sustainability of transport could come with trade-offs. Eventually, ***$S_{max.collab}$*** indicates that maximized collaboration minimizes climate impact. The benefit of ***$S_{non\text{-}TBA}$*** and ***S_{coop}*** over ***S_{TBA}*** is much smaller when excluding waste treatment, but the benefit of ***$S_{co\text{-}log}$*** and ***$S_{max.collab}$*** remains large even then. This emphasizes the leverage of logistics integration even when waste treatment benefits cannot be made use of.

The delivery to retail stage has a relatively small share of the total emissions but is sensitive to changes in distance. Particularly Helsingborg, with the highest GWP_{100} of all scenarios assessed, where the redistribution center was assumed to be located quite far from the city, proves the impact of this stage. High emissions at this stage are primarily connected to the use of small vehicles and can further increase due to urban traffic conditions. This shows the limited conclusions one can draw from a study focusing on urban areas where redistribution centers are locally available.

[23] Retailer A, pers comm. 2021-11-30; Logistics company, pers. comm. 2022-01-21

While electric vehicles still lack sufficient range to transport heavy goods for considerable distances (Liimatainen et al. 2014b), this stage could benefit from electrification. Especially in urban areas, this could reduce air pollution, traffic congestion, noise, and accident risks. This aligns with the Swedish government mobilizing forces to electrify regional road freight (Ministry of Infrastructure 2021).

The assessment was based on conventional diesel fuel, even though the usage of biodiesel in Sweden has decreased sharply in the past year, now making up 20% of final energy use in the transport sector (Swedish Energy Agency 2021). This necessary simplification could cause overestimations, which should be accounted for in future studies. Notably, the relative benefit of biofuels over fossil diesel decreases when taking its impact on land-use change into account (Liimatainen et al. 2014b). Similarly, animal feed production from food waste was also found preferable to anaerobic digestion from a land-use perspective rather than solely looking at fossil resource impact categories (Vandermeersch et al. 2014). In this study, including a single impact category allowed to investigate different scenarios in detail but could also potentially increase the risk of burden-shifting, as also stressed by Jensen and Arlbjørn (2014). This underlines that GHG emissions alone are not a reliable indicator of the environmental impact of transportation. Future research should address this limitation by taking a more holistic approach and evaluating the TBA system including additional impact categories.

5.2 The essential role of waste transport

The waste transport stage is particularly affected by the TBA but was not identified as an emission hotspot. However, the difference in emissions between the TBA scenarios and non-TBA scenarios at this stage is over 90% for all assessed cities, indicating a leverage point for improvement. The result is particularly sensitive to changes in waste transport distance, where adjustments make $\boldsymbol{S_{non\text{-}TBA}}$ more beneficial in the case of Uppsala; notably, this sensitivity can also be explained by the marginal difference between those scenarios. Beyond that, changes in the transport distances affect the results along with each section's proportional share of total GWP_{100}.

The additionally assessed cities allow investigating the common waste transport operations for bread in Sweden, beyond the special case of Uppsala. The fact that the impact of waste transport in those cities was found to be 32% higher on average indicates that transporting bread back to the bakery is not necessarily beneficial. The identified threshold of 285 km (*Figure 10*) indicates which cities could make use of ethanol production from bread, which questions the decision to transport bread back to the bakery, at least in terms of climate impact. However, the limited usability of the threshold for other cities is essential to note, as it is only valid if no other parameters change. This is not the case in other cities where long-distance delivery might be longer and cause more emissions, which cannot be viewed disconnected from the waste transport and treatment stage. Notably, the preferability threshold for ethanol increases considerably with the NTMCalc dataset (*Figure 10*), indicating that the threshold might lie somewhere in between the two thresholds and that the benefit of ethanol production is even higher. With disregard for long-distance transport emissions, the threshold can be used as a criterion for managing bread waste. Overall, these results show that transport network organization can affect the environmental performance of waste management, and that distance is particularly critical for ethanol production. These findings are in line with previous studies (Mondello et al. 2017; Brancoli et al. 2020).

As Helsingborg and Göteborg are close to bakeries, one could assume that it is logical to take bread back to the bakery, given that the same treatment options are unavailable locally. However, $S_{TBA+E,\ Göteborg}$ proved the opposite and showed that ethanol production can be more beneficial even in cities with a bakery, at least in terms of climate impact. This reveals improvement potential with a high level of acceptability and practicability among bakeries, that already have ethanol production from bread widely implemented. The non-TBA scenario outperforms all other options in Gävle, including $S_{TBA+E,\ Gävle}$, which somewhat confirms the validity of the threshold. One could speculate that the TBA makes the least sense in those cities far away from bakery and ethanol facilities, at least with companies' current practices within the TBA. Overall, the TBA must be evaluated for each city individually.

Referring to the reverse logistics concept, the results for Uppsala support the benefits promised by it, such as waste reduction and sustainable resource use. However, notably, the other assessed cities show that this depends on how the backflow of bread is re-valorized. Therefore, it is uncertain whether reverse logistics schemes always create more sustainable supply chains, as also concluded previously (Tibben-Lembke & Rogers 2001; Eriksson et al. 2017).

5.3 Bread waste treatment – a leverage point

It is important to view the waste hierarchy as a general guide for waste treatment prioritization, beyond which LCA should help to find the best treatment option for bread. A considerable influence of the bread waste management stage on the whole life cycle of bread was found, in line with the results presented by Eriksson et al.

(2015) and Jensen & Arlbjørn (2014). For instance, in $\boldsymbol{S_{TBA}}$, waste treatment outweighs the remaining supply chain, ultimately outperforming $\boldsymbol{S_{non\text{-}TBA}}$. The difference between the TBA and non-TBA scenarios is less pronounced when excluding waste treatment, emphasizing the impact of this stage. In the additionally assessed cities, particularly the way bread is treated there compromises the performance of the TBA, having a 11% lower emission savings potential at a stage that makes up a large proportion of total GWP_{100}. This immense disadvantage affects the average GWP_{100} in favor of a non-TBA system. Notably, the treatment of bread waste transported back to the bakery was modeled based on assumptions; changing these assumptions (35% pig feed, 65% donation) could change the result for the other cities and perhaps explain the reasoning behind this practice.

Notably, the identified thresholds for ethanol production refer to the current waste management infrastructure, and environmental savings are influenced by such (Brancoli et al. 2020). In Sweden, the availability of ethanol production is still limited, but the establishment of more facilities in the future could increase the benefit of this bread waste treatment option. Waste-based biofuels allow larger climate savings compared to crop-based biofuels. They also utilize a low-value waste stream (Hirschnitz-Garbers & Gosens 2015), which supports the circularity strategy adopted by the Swedish government (Ministry of Enterprise and Innovation & Ministry of the Environment 2020). The TBA offers a network to supply bread to ethanol production, which Hirschnitz-Garbers & Gosen (2015) found to be the most critical aspect for waste-based biofuels.

Several studies on food waste management found that the emission reduction potential is much higher when food is used as food than when used for energy production. This underlines the importance of tackling systematic causes of over-production and inefficient resource use, such as the TBA system. The estimated bread waste reduction and prevention rates in $\boldsymbol{S_{non\text{-}TBA}}$ and $\boldsymbol{S_{coop}}$ are uncertain, as the potential for waste reduction without with a TBA can only be theorized. Waste numbers also differ depending on a store's location and size and by that can infer the usage of different vehicles and transport frequency, and thus change emissions. It could be simulated how the $\boldsymbol{S_{non\text{-}TBA}}$ performed if bread return rates were not reduced from 7,7% to 2,5% as in this study; this would most likely reduce some benefits of waste prevention, eventually increasing the GWP_{100} of $\boldsymbol{S_{non\text{-}TBA}}$. It would also be valuable to evaluate whether the order volume would decrease without the TBA, as the responsibility for bread could make retailers reconsider their attitude towards full shelves.

Even though more efficient waste treatment arguably does not justify wasting food (Eriksson et al. 2017), the results in this assessment indicate that the waste rate might not matter as much as the waste treatment applied. The prevention of bread waste in $\boldsymbol{S_{non\text{-}TBA}}$ was outweighed by the poor bread waste treatment option assumed to be chosen by retailers. The ultimate question is how retailers would handle bread previously handled by bakeries. Of all unsold bread, the assumption was made that 20% is sent to anaerobic digestion and 80% is donated. Donation schemes exist in various forms in Sweden, and supermarkets commonly donate surplus food. However, the results suggest that the donation of bread is not the most favorable option from a strict climate impact perspective, as also found by previous studies (Eriksson et al. 2015; Brancoli et al. 2020), disregarding its social benefit. Donation centers were also found to experience a certain level of saturation (Ungerth 2021) and bread is among the most donated products in Sweden (Bergström et al. 2020). It is uncertain whether additional bread donations could be handled by charity, and if so, whether it would benefit them from a nutrition perspective; the increased donation of a product already donated in large volumes might not contribute effectively to the diet of donation recipients. It could also be debated whether bread can still be donated after spending a longer time at retail and whether this shifts bread waste to the household stage. Nevertheless, evidently most food waste at the retail level it is still fit for consumption (Cicatiello et al. 2017, 2020), emphasizing the need to study wastage at this stage and how surplus bread occurring there can be best managed, particularly focusing on questions relating to bread donations.

Eventually, the chosen treatment option significantly impacts the result because their respective emission factors differ considerably. For anaerobic digestion, the fuel considered avoided is an additional factor affecting its preferability (Bernstad Saraiva Schott & Andersson 2015). However, anaerobic digestion likely requires the least effort and thus also the lowest cost from a retail perspective. Nevertheless, if more bread was directed to anaerobic digestion, the emission savings potential and the preferability of the non-TBA scenarios would decrease. The assumptions made about waste treatment at retail is likely one of the main sources of uncertainty in this study, especially because this stage has such a large impact on the total GWP_{100} and because the difference between $\boldsymbol{S_{TBA}}$ and $\boldsymbol{S_{non\text{-}TBA}}$ is so small. Moreover, losses occurring during the donation process were not included in the model, which could decrease the benefit of this treatment option considerably.

$\boldsymbol{S_{pig}}$ was found to be beneficial, even though a relatively large transport distance and small vehicle was assumed for this treatment option. The same benefit, even to a larger extent, is expectable if retail directed bread to ethanol production. So far, supermarkets have not used bread waste as animal feed (Brancoli et al. 2017); however, some retailers acknowledge leverage in the treatment of bread and the usage for animal feed in particular [24]. One bakery stated that they stopped using bread for pig feed[25], which makes sense considering the economic preferability of ethanol production (Ungerth 2021). Most pig farms in Sweden are in the southern regions (Jordbruksverket 2020), setting a geographical limit similar to that of ethanol. In addition, legislative requirements to ensure the safety of feed and animal health, and suitable channels to sell bread for feed production are barriers to directing bread to pig feed (Eriksson et al. 2015; Brancoli et al. 2020). It could be valuable to evaluate if there is a distance threshold beyond which either of these options becomes more beneficial, considering the better local availability of pig farms but the higher emission savings potential of ethanol. This could support choosing the ideal treatment based on the location of a retail store.

Eriksson et al. (2015) also calculated the emission reduction potential of bread and found a much higher emission factor for bread donation and a lower emission factor for animal feed. The difference in the factor for donations could be explained by the inclusion of losses at the charity and household stages by Brancoli et al. (2020), which applies to more than 50% of surplus bread. It is recognized that the chosen values eventually dictate a specific hierarchy of preference. The factors of Brancoli et al. (2020) exclude transport, which is ideal for this study's aim to explore the transport section separately; only the value for donation includes some transport, but the impact was found to be small and not affect the overall conclusion regarding the preferability of scenarios. The difference between $\boldsymbol{S_{TBA}}$ and $\boldsymbol{S_{non\text{-}TBA}}$ is slight, so choosing different emission factors for waste treatment can affect the conclusions drawn. Eventually, waste treatment LCAs are affected by conditions such as the energy supply mix (Brancoli et al. 2020) and the substituted product (Eriksson et al. 2015). Brancoli et al. (2020) found a high sensitivity to the substituted energy type for anaerobic digestion, while ethanol and pig feed were less sensitive, underlining the favorability of these treatment options.

The ideal bread waste treatment option also depends on how food waste is defined. For instance, one bakery does not categorize bread that is revalorized as waste, as it still fulfills a purpose[26]. If food waste is all food produced for human consumption but used otherwise, all scenarios in which surplus bread is prevented are preferable. A certain waste rate seems necessary to keep the system going and avoid empty shelves, which was also confirmed in stakeholder interviews[27]. Retailer market

[24] Retailer D, pers. comm. 2022-03-02.
[25] Bakery B, pers. comm. 2021-11-04.
[26] Bakery A, pers. comm. 2022-03-04.
[27] Bakery A, pers. comm. 2022-03-04; Bakery D, pers. comm. 2021-11-16; Retailer D, pers. comm. 2022-01-18 and 2022-03-02.

power also plays into this, as bakeries, caught up in unfair trading practices, overorder bread to stay competitive[28]. While the weaker players carry the full responsibility for returned products, which can lead to inappropriate disposal (Eriksson et al. 2017), influential players have lower incentives to correctly manage stock and reduce waste (Canali et al. 2016), particularly when they do not bare the cost for the latter (Eriksson et al. 2017). The EU also recognizes that imbalances in bargaining power occur commonly in food supply chains (European Commission & European Parliament 2019) and advises member states to prohibit the free return of unsold products unless unambiguously agreed upon by the parties. Effectively, such practices remain legal, although they can manifest unfairly even when agreed upon (ibid.).

Previous studies on waste treatment relating to the TBA (Brancoli et al. 2020) excluded bread packaging, so even with its small impact, it was decided not to omit it here. To achieve a lower rejection rate in pre-treatment for anaerobic digestion, Brancoli et al. (2017) recommend separating bread from packaging, but $\boldsymbol{S_{pack}}$ does not confirm a benefit in doing so. However, the small fraction of packaging has an overall marginal effect, and the result is affected by the emission factors for incineration and anaerobic digestion (*Table 7*). Eventually, if separation held a sustainability benefit, its cost-benefit aspect still needs to be assessed.

5.4 Storage emissions – a possible trade-off?

The impact of retail storage is relatively low compared to the contribution of the transport and waste treatment stages, but still relatively higher for $\boldsymbol{S_{non\text{-}TBA}}$, $\boldsymbol{S_{coop}}$ and $\boldsymbol{S_{max.collab}}$ compared to $\boldsymbol{S_{TBA}}$. More energy is necessary at the retail stage if bread is stored longer, which exemplifies the possible trade-offs of reducing bread waste. Notably, storage time at the retail stage is based on estimations and would most likely vary with location, turnover, and size of a store. The storage time at the redistribution center is a conservative estimation but was also found to cause low emissions. The ultimate question is whether the benefit of reducing bread waste can outweigh additional storage input, resulting in increased emissions. This is hard to answer as bread waste reduction rates are uncertain. In contrast, reducing storage time could create rebound effects through increased transport emissions from just-in-time delivery. An increase in storage emissions could be justified by social benefits, that are, for instance, offering discounts on bread nearing its expiration date. However, some stakeholders[29] disapprove of discounts for bread, as it could

[28] Bakery C, pers. comm. 2022-01-28; Retailer A, pers comm. 2021-11-30; Retailer B, pers. comm. 2022-01-10; Retailer D, pers. comm. 2022-01-18; Logistics company, pers. comm. 2022-01-21

[29] Bakery A, pers. comm. 2022-03-04; Bakery C, pers. comm. 2022-01-28; Retailer C, pers. comm. 2021-11-29 and 2022-03-01.

harm their reputation for freshness, and reduced prices could also infer a burden shift to the household level. Extended retail storage must also be seen in the context of consumer expectations of freshness, and storage costs. The usage of the Swedish electricity mix to calculate storage emissions could be a shortcoming, as some retailers report the usage of renewable energy sources for their buildings (Company confidential 2021a; b). However, the low emissions based on this conservatively chosen electricity mix confirm the small impact of this stage.

5.5 The benefits of collaborative approaches

While the difference between S_{TBA} and $S_{non\text{-}TBA}$ is slight, it is considerable (42% on average) between S_{TBA} and the collaborative scenarios. Even though less pronounced than for co-logistics, the benefit of collaboration is also evident from S_{coop}. The joint consensus among stakeholders suggests that better forecasting is necessary and that sharing point-of-sale (POS) data could further improve the latter[30], allowing for maximized profits, improved customer satisfaction, and reduced waste levels. Other bread waste prevention options that would require cooperation are discounts for older bread, a reduced assortment, and targeted consumer information. Such changes could generally be motivated by cost-sharing or legislative pressure, as for instance the Czech Republic's 2018 amendment to the Food Act, requiring supermarkets to donate unsold, consumable food (Radio Prague International 2018). However, the somewhat heterogeneous opinions about the TBA indicate that a system change based on cooperation among industry players might be difficult to achieve. Inconsistency in stakeholder attitude could also imply inadequate communication between management and local store managers regarding food waste reduction goals, as also partially implied in one interview with a Swedish bakery.[31]

Although assortment is commonly seen as a waste driver, legislation might hinder collaborations regarding bread assortment (Nyheter & Wikén 2022). This exemplifies the difficulty in balancing environmental and economic aspects with legislation and policy recommendations, and somewhat discourages future incentives for collaborative approaches. Future research would benefit from including the economic outcome of the scenarios analyzed in this study, to ensure acceptance for these innovations among bread industry stakeholders. To support this, policymakers must address situations where wasting food is cheaper than prevention.

[30] Bakery A, pers. comm. 2022-03-04. and 2021-11-25; Bakery B, pers. comm. 2021-11-04; Bakery C, pers. comm. 2022-01-28.; Bakery D, pers. comm. 2021-11-16; Retailer B, pers. comm. 2022-01-10; Retailer C, pers. comm. 2021-11-29; Retailer D, pers. comm. 2022-01-18.

[31] Bakery A, pers. comm. 2021-11-25.

5.6 Validation of results

The results of this study are difficult to compare with previous research, as only a few studies touch on the supplier-retailer interface of the bread supply chain in Sweden (Brancoli et al. 2019), particularly the environmental impact of its' logistics. Brancoli et al. (2020) did not include transport in their analysis of bread waste revalorization options but also calculated a threshold for ethanol production. Their result of 730 km, including return trip, is in the same magnitude as the 285 km (primary dataset) and 590 km (NTMCalc) calculated here, as their threshold does not include return trips. The higher value identified by Brancoli et al. (2020) could also be explained by the fact that they compared ethanol production to incineration, which performs poorer than what ethanol was compared to in this study, namely a combination of anaerobic digestion and donation. However, Brancoli et al. (2020) excluded all life cycle stages before the waste management stage, so their threshold could be more generalizable than the one identified, and applicable to any city in Sweden.

Transport was not identified as an impact hotspot in other LCA on bread, contributing only 5% to its carbon footprint in the study by Espinoza-Orias et al. (2011), compared to the 40% that long-distance transport contributes here. Factors that affect this proportion are the life cycle stages included in the LCA as well as the resulting total emissions. This is exemplified by previous studies: Espinoza-Orias et al. (2011) found a CF of 1472 g CO_2e per kg of bread; Jensen & Arlbjørn (2014) found a CF of 731 g CO_2e per kg of bread, of which 9,7%, thus a slightly higher proportion, are caused by transport. Jensen and Arlbjørn (2014) compared several LCA and found that transport makes up between 2,4% and 35,4% of the bread life cycle emissions, excluding waste management and consumption. Most LCA on bread include the sourcing of raw materials and the processing, which cause a large part of emissions, thus reducing the proportional impact of transport. By excluding these stages, the large impact assigned to transport in the present study, particularly long-distance, is reasonable.

The average GWP_{100} of the transport stages, considering both the TBA and non-TBA scenarios in all cities, was calculated at 51g CO_2e per functional unit. Similar values for the absolute impact of bread transport were found in both previously mentioned studies, i.e., 70 g CO_2e (Jensen & Arlbjørn 2014) and 29 g CO_2e (Espinoza-Orias et al. 2011) for 1 kg of bread. The slight difference in results could be due to several reasons. Jensen and Arlbjørn (2014) assumed a distance of 175 km from depot to retail, much higher than what was assumed here, likely increasing the emissions, as this section typically uses small vehicles. Espinoza-Orias et al. (2011) assumed a distance of 50 km from bakery to retail, which is hardly comparable to the Swedish case and much lower than the distance modeled here, and also explains their low result for transport emissions. Espinoza-Orias et al. (2011) actually included raw material transport from Canada, but transport emissions are not increased considerably by that. This exemplifies the impact of local trucking compared to the comparably small impact of sea freight per kg transported. Jensen & Arlbjørn (2014) found a net savings potential of -325 g CO_2e per kg of bread for waste management, somewhat in line with the identified -554 g per kg of bread for $\boldsymbol{S_{TBA}}$, since Jensen & Arlbjørn assumed that bread is incinerated.

5.7 Evaluation of data uncertainties

The variety of datasets that offer emission factors, particularly for transport emissions, can add uncertainty to LCA results, which this thesis accounted for by testing a second dataset. The differing climate impact when using NTMCalc, most notable at the delivery to retail and waste transport stages, can be explained primarily by the difference in emission factors. Generally, the methodology of including certain aspects into a characterization factor is a source of data uncertainty. In this regard, NTMCalc could be considered more transparent, as it allows for adaptive parameter settings, including road type, fuel type with the proportion of biodiesel, load capacity, and load weight. Ecoinvent and DEFRA provide fewer options for adjustments. For example, NTMCalc allows adapting vehicle load in percent of the load capacity, while Ecoinvent provides emission factors simply with an average load. DEFRA (2021b) offers several percentage loads (0%, 50%, 100%, average) for larger vehicles and an average load for smaller vehicles. However, previse parameter settings can be hampered by a lack of data, especially since data on bread waste at the supplier retailer interface is rare (Brancoli et al. 2019) and not easily accessible due to the controversy and secrecy around food waste (Herzberg et al. 2022). For instance, attempting to model the delivery to retail stage more precisely with consideration of load differences, the precise parameters in NTMCalc were helpful to a limited extent as vehicle capacity and utilization had to be assumed based on limited available data.

The driving environment, i.e., road type, affects the energy intensity of transport, as fuel consumption is higher in irregular traffic conditions (Liimatainen et al. 2014b). An 18t truck consumes 33% more fuel in delivery mode (urban conditions) than in highway mode (ibid.). Looking at the NTMCalc dataset, the emission factors are higher for urban roads and lower for motorways than for the average road; thus, using the latter could cause an under-estimation and over-estimation in the respective case. Ecoinvent does not give information on the driving environment used for the emission factor. At the same time, DEFRA (2021a) explains that the emission factors refer to the typical usage pattern of each truck class. For instance, articulated HGVs (Heavy Goods Vehicles) typically drive with higher fuel efficiency and speed, in free-flowing traffic conditions, and emit relatively less than rigid HGVs, which operate at lower speed and fuel efficiency, and usually in congested traffic conditions (ibid.). This implies that the road type is included in the emission factor in a generalized way, limiting the adaption to specific cases and assigning a lower impact to the large vehicle based on both size and its usage, for example. However, as the vehicles are mainly used in their common driving environment in the modeled scenarios, using DEFRA might be more representable than using the average road values of NTMCalc.

The inclusion of more parameters using NTMCalc, based on best knowledge and several assumptions, resulted in lower climate impact than using Ecoinvent and DEFRA (*Figure 11*). The reason for the lower result could be that the other data sources are more general and perhaps conservative to avoid underestimations. At the same time, the level of preciseness in NTMCalc could increase the margin of error if parameters are set ill-advised. Overall, the lower results with the additional dataset tested suggest that the primary results are likely not an underestimation. One exception could be the smallest vehicle, where the emission factors differ considerably. This points to another possible reason for the difference in emission factors; different vehicle categories used in the datasets complicate a straightforward comparison of the vehicles. The impact of dataset selection on the results is clear, and it could be beneficial to use only one dataset, if possible. It would be ambitious to conclude which one is the better choice for this study or for Sweden in general, but data availability is certainly a decisive factor in this regard.

5.8 Limitations

This study had to use secondary data, average values, and estimations, which can impact the reliability of the results, especially because consequential LCA should commonly use marginal data (Ekvall & Weidema 2004). Moreover, some simplifications necessary for modeling may not account for all company operations. Several sensitivity analyses were conducted to account for uncertainties and to increase the robustness of the results. However, LCA results must be used with precaution as they show a potential environmental impact, not a precise value prediction of the impacts (Klöpffer et al. 2014). Future research quantifying innovations in the bread supply chain requires a large-scale trial of alternative scenarios, including the measurement of bread waste volumes, to improve data availability and allow for more precise LCA input data.

Some methodological choices had to be made that could affect the results. Differences in long-distance transport in a non-TBA system are uncertain and thus, the models assumed the same operations in both the TBA and non-TBA system.

This is a potential shortcoming in this study, as this transport section makes up a large fraction of total emissions. However, the locations of the bakeries are fixed, which makes changes in long-distance delivery arguably more challenging to achieve than what happens after the bread is already delivered. Eventually, a focus on modeling the differences in waste transport as a less static part was thus reasonable. However, indirect effects of the TBA on the long-distance delivery part are possible and it could be speculated that delivery frequency would increase without a TBA, to maintain freshness and high assortment on the shelves, without removing older bread loaves. Higher delivery frequency with smaller volumes could infer the usage of smaller trucks, thus increasing emissions. Beyond this, one bakery[32] delivers fresh bread twice a day, which leaves room for judgment on whether one or two delivery trips should be modeled for completing the delivery of the functional unit. Including the weekly or monthly delivery volumes into the model would enable an assessment of delivery frequency, bread removal, and re-fill rates in different bread supply chain scenarios. This would allow an in-depth investigation of the TBA's effect on long-distance transport and possible rebound effects.

The effects of cancelling the TBA on the delivery to retail stage were found challenging to address. The different systems can only be modeled with different vehicle load utilization, depending on whether unsold bread is picked up during the delivery trip or not. However, the change in load is minimal, and the primary data set did not allow for modeling small percentage changes in load. Nevertheless, if load was modeled more precisely, this could add emissions at this stage to $\boldsymbol{S_{non\text{-}TBA}}$ and infer a changed result in favor of the TBA.

The results were also found to be sensitive to the way backhaul is modeled. It could be included in long-distance transport and waste transport, which both either contribute largely to the total emissions or are most profoundly affected by the TBA. However, the modeling of return trips was not trivial, which is also connected to the database used. The Ecoinvent database provides emission factors that include the backhaul of trucks, while DEFRA and NTMCalc do not. The usage of one or the other database can lead to different results, especially considering that the emission factors from Ecoinvent including backhaul are not much higher than the others. The limited data availability complicates this, too. Assumptions on backhauling and empty returns also depend on who is operating it. A large company would perhaps maximize efficiency, using the backhaul for another product, excluding it from the system boundary, while small actors would likely allow lower utilization rates. According to DEFRA (2021a), the impact of vehicle load on emissions increases with vehicle size. Thus, excluding backhauls from long-

[32] Bakery A, pers. comm. 2022-03-04.

distance transport is a shortcoming and might lead to under-estimations, particularly because previous research suggests that a low vehicle utilization cannot be ruled out. For instance, Croci et al. (2021) state that the average load factor was below 50% of half of the commercial vehicles in Europe, similar to Braam et al. (2001) who found that usually only 40-70% of vehicle capacity is used. However, Liimatainen et al. (2014a) found that among the Nordic countries, Sweden has a low level of empty-running transport due to its three major economic regions in the south which balance transport flows. This leaves room to assume vehicle utilization could be less favorable in northern Sweden and points to the shortcoming of this study's focus on the southern part of Sweden.

With its central position in Sweden and beneficial waste treatment, Uppsala presented a somewhat ideal scenario, making it hard to generalize from. The selection of additionally assessed cities considered the fifteen largest cities in Sweden and aimed to cover southern Sweden as the main economic region. The cities were also selected aiming to test varying distances to bakeries and the southern ethanol facility, allowing to draw conclusions about different aspects of the supply chain. Eventually, the multiple case studies still limit general conclusions, as the results suggest evaluating the TBA individually for each city. Further research could extend the assessment for northern Sweden, including the second bioethanol plant. This would allow an evaluation of alternative bread supply chain scenarios in a less connected and economically integrated area of the country, which could yield longer distances in most of the transport sections. Focusing on the three largest bakeries in Sweden is also a shortcoming of this study, perhaps providing distorted results by extrapolating the market share, and disregarding treatment options feasible for small-scale operations.

6. Conclusion

Food waste reduction benefits the environment, economy, and society and is crucial for sustainable development. The Swedish bread take-back agreement has been identified as a food waste risk factor, but changes to the business model are debated. This study compared the climate impact of the conventional take-back agreement for surplus bread in Sweden to a conceptual system with altered logistics and waste management, consequently reducing a gap within the research on the TBA system and contributing to a more sustainable bread supply chain in Sweden.

The results showed that a shift to a non-TBA system in the city of Uppsala increased the climate impact marginally. Inversely, in other Swedish cities, the non-TBA scenarios outperformed the TBA system, as transporting bread back to the bakery and the connected re-valorization were found to be disadvantageous from a climate impact perspective. As long-distance transport and bread waste treatment were identified as critical climate impact hotspots, this study highlights the need to decarbonize the Swedish transport sector and the importance of high-value reuse of bread waste, such as ethanol production. Whether or not the TBA exists, supermarkets should consider alternatives to the standard bread waste treatment. The results stress the link between waste transport and treatment under consideration of the local waste treatment infrastructure. Overall, the focus should not necessarily be to remove the TBA in its entirety; instead, the solution could be a collaborative approach that prevents bread wastage in the first place, perhaps by legislative pressure, and at the same time makes use of the clean waste stream generated by the TBA. The most acceptable and practicable solution from the bread industry stakeholders' point of view remains to be identified. Eventually, this study emphasizes that sustainable development is only achievable through joint commitment and action.

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Internal material

All footnotes refer to unpublished sources based on personal communication. All company information and their names are anonymized, please get in touch with the author for further clarification.

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Popular science summary

Food waste reduction is a critical element of sustainable development, as the wastage of food infers environmental and economic implications and creates ethical controversies. Reducing food waste-related emissions is vital, and more efficient resource use and improved food security are crucial considering the growing world population. Among various drivers for food wastage, certain business practices can cause food wastage; one example of this is the bread take-back agreement (TBA) in Sweden. The TBA allows the retailer to give back unsold bread and pay only for the amount sold, externalizing the risk and cost of the generated food waste. Previous research suggests that bread waste levels could go down if retailers had to take responsibility for unsold products. The TBA dictates the complex transport logistics of bread operated by Swedish bakeries, and affects waste treatment, offering a clean flow of bread, usable for various re-valorization methods.

A holistic evaluation of the potential benefits and limitations of the TBA system is necessary, particularly regarding its implications on transport. This study quantified the climate impact of alternative bread supply chain scenarios, to identify climate impact hotspots and opportunities to achieve a more sustainable bread supply chain in Sweden. The method used in this study was Life cycle assessment (LCA), one of the most used tools to assess environmental impacts. By assessing all inputs and outputs along the entire life cycle of a product system, LCA provides a holistic overview of impacts and helps to avoid burden shifting to other stages. This LCA used Global Warming Potential as a single impact category, a functional unit of 1kg of bread, and factory gate to grave as a system boundary. A shift from the conventional TBA system to a non-TBA system increased the climate impact by 5% in the case of Uppsala. In other Swedish cities, a non-TBA system outperforms the TBA system with 28% lower emissions, due to the disadvantageous transport back to the bakery and the poor re-valorization in the TBA scenarios. As long-distance transport and bread waste treatment were identified as climate impact hotspots, this study emphasizes the need to decarbonize the Swedish transport sector and use high-value bread waste treatment, such as ethanol production. The focus should not necessarily be to remove the TBA in its entirety, instead, the solution could be a collaborative approach that prevents bread wastage in the first place, perhaps by legislative pressure, and at the same time makes use of the clean waste stream generated by the TBA.

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Appendix 1

Appendix 1 provides background information about the models, calculations, how emission factors were derived, and how assumptions were made.

Appendix 1.1

Vehicle type selection

For long-distance delivery a large articulated truck **(HGV, Articulated >33t)** was chosen based on the interview with one bakery; this could differ for other bakeries with smaller market shares, but was still deemed representable and any other assumption too uncertain. The selection of vehicles for the delivery to retail, the following data on bread deliveries was used, provided in an interview[33].

Store	**Bread loaves delivered/week**
Medium store*	700
Large store*	2000
Large store*	7000
Estimations	**Quantities**
Daily delivery/store in loaves	100-1000
Average delivery/day/store in loaves	500
Average delivery/day/store in kg	250
Average truck load, delivery route type A	1000
Average truck load, delivery route type B	1750

**Anonymized*

One bakery explained that they use truck with a load capacity of 1t and total vehicle weight of 2,5t, thus a van was chosen here **(Van, average, up to 3,5t).** However, they state that the vehicle size must be compensated by taking more than one drive to deliver all bread; based on this, for the other bakeries, a larger truck **(HGV Rigid, >3,5-7,5 tons)** was chosen, as they cooperate and must thus transport approximately the same amount of bread. For $S_{co\text{-}log}$, a larger vehicle **(HGV, Rigid, (>7,5-17 tons)** was chosen due to co-logistics operated.

[33] Bakery A, pers. comm. 2022-03-04.

To determine the appropriate vehicle for the waste transport, the following estimations were made based on the data above provided data:

250 kg of bread/day/store (average), return rate 7,7%
Estimated number of stores in Uppsala region: 72
Estimated total returns at regional hub/day in kg: 1386 (One bakery)
Estimated total returns at regional hub/week in kg: 9702
*50% for waste rate of 3,5% (**S_{coop}**): 4851*

Bread is picked up weekly or bi-weekly by the ethanol producer. According to Ecoinvent, 'the average freight load factor of a 16-32 metric ton lorry is 5.79 tonnes, with a gross vehicle weight (GVW) of 15.79 tonnes' (Valsasina n.d.). Based on this, the vehicle was chosen **(HGV, Rigid >17t**), assuming that it can go over the average load and pick up 95% of the 9702 kg of bread waste per week. For ***$S_{co\text{-}log}$*** the same vehicle is used, but with a higher load factor. Charity organizations are assumed to use a small van **(Van, Class I, up to 1.305 tons)**, as only 5% of returned bread volume is directed to this treatment option, which is da daily return of 69,3 kg (of one bakery). This vehicle is also used in ***$S_{co\text{-}log}$***, as the charity organization likely does not have a range of vehicles to choose from and might just go twice of with low utilization if necessary. For the same reason, this vehicle is also used in ***$S_{non\text{-}TBA}$***. Bread waste from the retailers directed to biogas production was assumed to be picked up in a common municipal waste collection lorry.

Rail freight

The ecoinvent database provides an emission factor for rail freight in Europe, but refers to a train operating with diesel, electricity, or coal. However, nearly all passenger trains are electric powered in Sweden (SJ 2020), and according to Green Cargo, the largest actor in rail freight in Sweden, with 60% of market share (Regeringskansliet n.d.) more than 96% of rail shipments were operated with electric locomotives (Green Cargo AB 2021). It was decided to calculate an emission factor for rail freight in Sweden, using the electricity usage per ton.km provided by Green Cargo.

Rail freight emissions, Green Carbo AB			
Input factor	**Quantity**	**Unit**	**Source**
kWh usage electric rail traffic/ton.km	0,0360	kWh/ ton.km	Green Cargo 2021
market for electricity, medium voltage, SE	0,0442	kg CO_2e/ kWh	Ecoinvent[34]
GWP100/ton.km electric rail traffic	**0,0016**	kg CO_2e/ ton.km	Own calculation

[34] Treyer n.d.b.

Rail freight emissions, general			
Input factor	**Value**	**Unit**	**Source**
market for transport, freight train, Europe without Switzerland (operates with diesel, electricity, or hard coal)	**0,0458**	kg CO_2e/ ton.km	Ecoinvent[35]

Calculation: Rail freight emissions, Swedish average			
Fraction of market	**Value**	**Unit**	**Source**
70% of rail freight electricity powered	0,0016	kg CO_2e	Assumption based on Regeringskansliet n.d.
30% of rail freight powered according to the European mix	0,0458	kg CO_2e	
Emission factor for rail freight, Sweden	**0,0149**	kg CO_2e	Own calculation

Appendix 1.2

Vehicle load

The load was modelled in a simplified way by assuming a total of 100kg of bread delivered to ten stores, with a return rate of 7,7% and 0%. The load volume was averaged out, and based on this it was determined most realistic to calculate with a load of 50% for the delivery to retail transport, and where not possible due to emission factor availability, with an average load factor.

Route		S_{TBA}	$S_{non\text{-}TBA}$
From	**To**	**~ weight of bread (kg)**	
Redistribution center	1	70,00	70
1	2	60,08	60
2	3	50,16	50
3	4	40,24	40
4	5	30,31	30
5	6	20,39	20
6	7	10,47	10
7	Redistribution center	0,55	0
		~35	~35

Appendix 1.3

Delivery to retail route type A

Shop name	Address	Station
Hemköp	Kansliskrivargatan 1, 752 37 Uppsala	1
ICA Nära Hörnan Kåbo	Artillerigatan 16, 752 37 Uppsala	2
Coop Konsum Luthagen	Ringgatan 31, 752 17 Uppsala	3
Willys Kungsgatan	Kungsgatan 95, 753 18 Uppsala	4

[35] Ecoinvent system generated n.d.c

Delivery to retail route type B (as illustrated in Figure 5)

Shop name	Address	Station
City Gross Boländerna	Stångjärnsgatan 10, 753 23 Uppsala	1
Willys Kungsgatan	Kungsgatan 95, 753 18 Uppsala	2
Coop Konsum Luthagen	Ringgatan 31, 752 17 Uppsala	3
ICA Folkes Livs	Rackarbergsgatan 8, 752 32 Uppsala	4
ICA Nära Hörnan Kåbo	Artillerigatan 16, 752 37 Uppsala	5
Lidl Gottsunda	Valthornsvägen 1a, 756 50 Uppsala	6
Hemköp Rosendal	Kansliskrivargatan 1, 752 37 Uppsala	7

Full round trip type A: 24,9 km
Full round trip type B: 26,2 km

Average distances from retailers to treatment options in km

The calculations are based on the average distances calculated for delivery route B, as those are based on a larger sample of supermarkets and serve as a good reference.

Station	Biogas plant (Uppsala Vatten)	Donation central (Stadmission Uppsala)	Recyling center (Returpappercentralen)
1	2,6	2,0	1,6
2	1,6	0,6	1,1
3	5,0	3,9	4,6
4	4,8	3,6	4,9
5	5,2	4,5	5,4
6	6,9	6,2	7,1
7	3,9	3,2	4,1
	~4,3	~3,4	~4,1

Appendix 1.4

Energy consumption at redistribution center and retail store

Category	Final energy use in food retail (Swedish Energy Agency 2010)		Applied to Swedish retailer (Company confidential 2021b)	
	kWh/m2/ year	Fraction	kWh/m2/ year	kWh/m2/ hour
Total energy use	399,2	100%	348,0	0,040
Total electricity use	321,4	81%	280,2	0,032
Refrigeration	144,5	36%	125,3	0,014
Heating	77,8	19%	66,1	0,008

Electricity use, ambient storage, Swedish retailer*	154,9	0,018

* *= Total electricity use – refrigeration*

Cross-check: According to DEFRA (2008), more than 70% of energy is used for electricity, of which 30-60% is used for refrigeration. These values are in line with the percentages of the Swedish Energy Agency that were applied to the retailer. The energy use was calculated as following:

$$\mathbf{kWh/FU} = \mathbf{(yearly\ energy\ consumption\ (kWh) \div 365 \div 24) \times storage\ area\ (m2) \times turnover\ time\ (h)}$$

Appendix 1.5

Packaging waste treatment, emission factors

For the treatment of plastic packaging, a plastic bag made from polyethylene, both the incineration as well as the recycling of material must be considered.

Determination of plastic recycling emission factor

On a very basic level, plastic film production can be divided into the production of plastic granulates, also called resin, and their processing into plastic film. This is relevant to distinguish because the recycling of plastic packaging prevents the production of virgin polyethylene, but not the processing into film. While the prevention of virgin material production prevents emissions, the processing still causes emissions (U.S. Environmental Protection Agency 2016).

Activity	Quantity		Unit	Source/Emission factor
Raw material production	(+)	2,4766	kg CO_2e /kg	market for polyethylene, low density, granulate, GLO[36]
Processing	(+)	0,5404	kg CO_2e /kg	market for extrusion, plastic film, GLO[37]
Plastic film production	(+)	1,9362	kg CO_2e /kg	Own calculation
Avoided emissions	**(-)**	**-1,3958**	**kg CO_2e/kg**	**Own calculation**

Cross-check:

Plastic film production	(+)	3,0170	kg CO_2e /kg	market for packaging film, low density polyethylene, GLO[38]

Determination of plastic incineration emission factor

Plastic incineration requires energy to burn the material, but the process itself also provides electric and thermal energy as plastic is an energy intensive material. Therefore, both the emissions from the incineration and the avoided emissions from the produced energy must be considered.

[36] (Bourgault n.d.b)

[37] (Ecoinvent system generated n.d.a)

[38] (Bourgault n.d.a)

Activity	Value / Unit		Source/Emission factor
Incineration of waste polyethylene	3,025	kg CO_2e /kg	treatment of waste polyethylene, municipal incineration, RoW[39]
Net electric energy production	5,500	MJ/kg	
Net thermal energy production	10,690	MJ/kg	
Net electric energy production	1,542	kWh/kg	Own calculation
Net thermal energy production	2,969	kWh/kg	Own calculation
Emission factor, electricity prod.	0,044	kg CO_2e /kWh	market for electricity, medium voltage, SE[40]
Emission factor, heat prod.	0,003	kg CO_2e /kWh	heat and power co-generation, wood chips[41]
Avoided emissions, electric energy production	-0,068	kg CO_2e /kWh	Own calculation
Avoided emissions, thermal energy production	-0,008	kg CO_2e /kWh	Own calculation
Avoided emissions, sum	-0,076	kg CO_2e /kWh	Own calculation
Incineration of waste polyehtylene, per kg disposed	**2,949**	**kg CO2e/kg**	**Own calculation**

Determination of final plastic packaging treatment emission factors

68% of packaging in Sweden recycled, but the recycling rate for plastic is lower than for glass or paper (Avfall Sverige 2021). Packaging that is not recycled is incinerated for energy recovery (ibid.) It can be assumed that a higher recycling rate is achievable on a company level than on household level, as companies can operate with economies of scale and more automated, regulated processes; the recycling rate for industry was 90% and for households 60%. The emission factors were calculated as shown below.

Treatment	Quantity	Unit	Source
Recycling of polyehtylene, per kg disposed	-1,3958	kg CO_2e	Own calculation, see above
Incineration of polyehtylene, per kg disposed	2,9486	kg CO_2e	Own calculation, see above
Recycling rate household level	0,6	%	Own estimation based on Avfall Sverige 2021
Recycling rate industry level	0,9	%	Own estimation
Plastic treatment, household	**0,3419**	**kg CO_2e**	**Own calculation**
Plastic treatment, industry	**-0,9614**	**kg CO_2e**	**Own calculation**

[39] Doka n.d.

[40] Treyer n.d.b.

[41] Treyer n.d.a.

Appendix 2

Appendix 2 provides the detailed life cycle assessment calculations for S_{TBA} *and* $S_{non\text{-}TBA}$. *For detailed calculations for the other scenarios, please get in touch with the author.*

All distances measured in Google maps based on the shortest street route. Routes were determined based on assumptions, company reports, and information provided in an interview with one bakery. Vehicles were selected based on company reports, an interview with one bakery, and assumptions. The waste management stage is based on a total weight of 1,02 kg, including bread (1 kg) and packaging (0,02 kg).

For detailed explanations for vehicle selection see Appendix 1.1, for load estimation see 1.2, for delivery to retail route see 1.3, for energy consumption for storage see 1.4, for packaging emission details see 1.4.

Long-distance delivery *(STBA & Snon-TBA)*

From	To	Fraction of bread taking this way (%)*	Distance (km)	Vehicle	Load	Fuel	Comment on backhaul	kg CO2e/ ton.km	Source for emission factor	kg CO2e/ different routes of each supplier	sum kg CO2e/ route	average kg CO2e/ supplier
Bakery	Redistribution center	31%	118	HGV, Articulated >33t	average	diesel	Backhaul not considered as we don't know if they go back empty	0,080	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	3,0E-03	3,0E-03	4,8E-03
Bakery	Redistribution center		335	HGV, Articulated >33t	average	diesel	Backhaul not considered as we don't know if they go back empty	0,080	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	8,5E-03	8,5E-03	
Bakery	Umeå train station		7,5	HGV, Articulated >33t	average	diesel	Backhaul not considered due to small distance	0,080	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	1,9E-04	3,0E-03	
Umeå train station	Uppsala train station		571	Freight train	not specified	electric/ diesel	Assumed train doesn't go back empty	0,015	See Appendix 1.1	2,7E-03		
Uppsala train station	Redistribution center		3,6	HGV, Articulated >33t	average	diesel	Backhaul not considered due to small distance	0,080	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	9,1E-05		
Bakery	Älvsbyn station	20%	1,4	Lorry, freezing (16-32t/>32t)	not specified	diesel	Backhaul included in emission factor but not significant due to small distance	0,461	market for transport, freight, lorry with refrigeration machine, freezing, GLO, LCIA version, IPCC 2013 climate change GWP100a, Ecoinvent 3.8	1,3E-04	9,8E-03	9,6E-03
Älvsbyn station	Uppsala main station		833	Freight train, freezing	not specified	electric/ diesel/ coal	Emission factor considers empty trips indirectly over the lifetime of the vehicle, thus overlooked because it can be assumed the vehicle does not go back empty.	0,055	market for transport, freight, train with reefer, freezing, GLO, LCIA version, IPCC 2013 climate change GWP100a, Ecoinvent 3.8	9,3E-03		
Uppsala main station	Redistribution center		3,6	Lorry, freezing (16-32t/>32t)	not specified	diesel	Backhaul included in emission factor but overlooked due to small distance	0,461	market for transport, freight, lorry with refrigeration machine, freezing, GLO, LCIA version, IPCC 2013 climate change GWP100a, Ecoinvent 3.8	3,4E-04		
Bakery	Örnsköldsvik station		41,2	Lorry, freezing (16-32t/>32t)	not specified	diesel	Backhaul included in emission factor but overlooked in this single case, might infer a more conservative result, but this factor was still the ideal option	0,461	market for transport, freight, lorry with refrigeration machine, freezing, GLO, LCIA version, IPCC 2013 climate change GWP100a, Ecoinvent 3.8	3,9E-03	9,4E-03	
Örnsköldsvik station	Uppsala main station		461	Freight train, freezing	not specified	electric/ diesel/ coal	Emission factor considers empty trips indirectly over the lifetime of the vehicle, thus overlooked because it can be assumed the vehicle does not go back empty.	0,055	market for transport, freight, train with reefer, freezing, GLO, LCIA version, IPCC 2013 climate change GWP100a, Ecoinvent 3.8	5,2E-03		
Uppsala main station	Redistribution center		3,6	Lorry, freezing (16-32t/>32t)	not specified	diesel	Backhaul included in emission factor but overlooked due to small distance	0,461	market for transport, freight, lorry with refrigeration machine, freezing, GLO, LCIA version, IPCC 2013 climate change GWP100a, Ecoinvent 3.8	3,4E-04		
Bakery	Redistribution center	49%	540	HGV, Articulated >33t	average	diesel	Backhaul not considered as other products are transported back	0,080	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	2,2E-02	2,2E-02	2,4E-02
Bakery	Redistribution center		678	HGV, Articulated >33t	average	diesel	Backhaul not considered as other products are transported back	0,080	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	2,7E-02	2,7E-02	

3,9E-02
or: *0,0388*

* *Based on market share, rounded up to 100%*

Storage emissions

STBA

Input	Fraction of bread taking this way (%)*	Turnover time (days)	Turnover time (h)	Storage area (m2)	Energy consumption (kWh/m2/hour)	Energy consumption (kWh)	kg CO2e/ kWh	Source for emission factor	kg CO2e
Redistribution center									
Electricity, frozen storage	20%	Redistribution, max. 1 day	12	0,003	0,0320	0,0012	0,044	market for electricity, medium voltage, SE, IPCC climate change GWP 100a, Ecoinvent 3.8	5,1E-05
Electricity, ambient storage	80%	Redistribution, max. 1 day	12	0,012	0,0176	0,0025	0,044	market for electricity, medium voltage, SE, IPCC climate change GWP 100a, Ecoinvent 3.8	1,1E-04
Heating	100%	Redistribution, max. 1 day	12	0,015	0,0077	0,0014	0,003	heat and power co-generation, wood chips, 6667 kW, state-of-the-art 2014, SE, IPCC 2013 climate change, GWP 100a, Ecoinvent 3.8	3,7E-06
									1,7E-04
								or:	*0,0002*
Retail stage									
Electricity, ambient storage	100%	4	96	0,0150	0,0176	0,0253	0,044	market for electricity, medium voltage, SE, IPCC climate change GWP 100a, Ecoinvent 3.8	1,1E-03
Heating	100%	4	96	0,0150	0,0077	0,0111	0,003	heat and power co-generation, wood chips, 6667 kW, state-of-the-art 2014, SE, IPCC 2013 climate change, GWP 100a, Ecoinvent 3.8	2,9E-05
									1,1E-03
								or:	*0,0011*

Snon-TBA

Input	Fraction of bread taking this way (%)*	Turnover time (days)	Turnover time (h)	Storage area (m2)	Energy consumption (kWh/m2/hour)	Energy consumption in (kWh)	kg CO2e/ kWh	Source for emission factor	kg CO2e
Redistribution center									
Electricity, frozen storage	20%	Redistribution, max. 1 day	12	0,0030	0,0320	0,0012	0,044	market for electricity, medium voltage, SE, IPCC climate change GWP 100a, Ecoinvent 3.8	5,1E-05
Electricity, ambient storage	80%	Redistribution, max. 1 day	12	0,0120	0,0176	0,0025	0,044	market for electricity, medium voltage, SE, IPCC climate change GWP 100a, Ecoinvent 3.8	1,1E-04
Heating	100%	Redistribution, max. 1 day	12	0,0150	0,0077	0,0014	0,003	heat and power co-generation, wood chips, 6667 kW, state-of-the-art 2014, SE, IPCC 2013 climate change, GWP 100a, Ecoinvent 3.8	3,7E-06
									1,7E-04
								or:	*0,0002*
Retail stage									
Electricity, frozen storage	10%	4	96	0,0015	0,0320	0,0046	0,044	market for electricity, medium voltage, SE, IPCC climate change GWP 100a, Ecoinvent 3.8	2,0E-04
Electricity, ambient storage	10%	5	120	0,0015	0,0176	0,0032	0,044	market for electricity, medium voltage, SE, IPCC climate change GWP 100a, Ecoinvent 3.8	1,4E-04
Electricity, ambient storage	90%	9	216	0,0135	0,0176	0,0513	0,044	market for electricity, medium voltage, SE, IPCC climate change GWP 100a, Ecoinvent 3.8	2,3E-03
Heating	100%	9	216	0,0150	0,0077	0,0251	0,003	heat and power co-generation, wood chips, 6667 kW, state-of-the-art 2014, SE, IPCC 2013 climate change, GWP 100a, Ecoinvent 3.8	6,6E-05
									2,7E-03
								or:	*0,0027*

** Based on market share (S1, S2) and several own estimations (S2)*

Delivery to retail *(STBA & Snon-TBA)*

Route	Fraction of bread taking this way (%)*	Distance (km)	Vehicle	Load	Fuel	kg CO2e/ ton.km	Source for emission factor	kg CO2e
Full round trip type A	49%	24,9	Van, average (up to 3,5t)	average	diesel	0,603	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	7,5E-03
Full round trip type B	51%	26,2	HGV Rigid (>3,5-7,5 tonnes)	50%	diesel	0,451	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	6,1E-03
								1,4E-02
							or:	*0,0136*

** Based on market share, rounded up to 100%*

No backhaul modeled because the delivery functions in a circular mode.

Waste transport

STBA

From	To	Distance (km)	Vehicle	Load	Fuel	Comment on backhaul	kg CO2e/ ton.km	Source for emission factor	fraction of bread taking this way	wasted fraction in kg	kg CO2e/ waste treatment option	kg CO2e/ bakery
RC 1 Uppsala	Donation central	13,2	Van, Class I (up to 1.305 tonnes)	average	diesel	a	0,815	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	2%	1,9E-03	2,1E-05	1,5E-03
RC 1 Uppsala	Lantmännen Agroetanol	226	HGV, Rigid >17t	average	diesel	b	0,181	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	47%	3,7E-02	1,5E-03	
RC 2 Uppsala	Donation central	5,6	Van, Class I (up to 1.305 tonnes)	average	diesel	a	0,815	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	3%	2,0E-03	9,1E-06	1,6E-03
RC 2 Uppsala	Lantmännen Agroetanol	224	HGV, Rigid >17t	average	diesel	b	0,181	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	48%	3,8E-02	1,5E-03	
												3,1E-03
											or:	*0,0031*

Snon-TBA

From	To	Distance (km)	Vehicle	Load	Fuel	Comment on backhaul	kg CO2e/ ton.km	Source for emission factor	fraction of bread taking this way	wasted fraction in kg	kg CO2e
Retailer	Donation central	6,8	Van, Class I (up to 1.305 tonnes)	average	diesel	a	0,815	UK Government GHG Conversion Factors for Company Reporting (2021), Freighting goods	80%	2,0E-02	1,1E-04
Retailer	Biogasanläggning Uppsala	4,3	Municipal waste collection lorry	not specified	diesel	c	1,280	**market for municipal waste collection service by 21 metric ton lorry, GLO, Version LCIA, IPCC 2013 climate change GWP100a, Ecoinvent 3.8**	20%	5,1E-03	2,8E-05
											1,4E-04
										or:	*0,0001*

RC = Redistribution center

a Backhaul included by doubling distance. No value for empty trip available, but according to DEFRA, the smaller the vehicle the less load capcacity affects overall emissions

b Backhaul excluded to be able to assess threshold for single trip and because we assume company does not go back

c Average distance based on delivery route A, no packaging separation, emission factor does not mention backhaul which was accepted as it drives a circle

Waste management

STBA

Bread

Bread valorisation	Fraction of bread waste (%)	Fraction of bread waste (kg)	Comment	Emission factor per kg bread	Source for emissions factor	kg CO2e
Ethanol production	95%	0,0732	Most of the bread goes to ethanol production as all bakeries state this as their main treatment option, and because it is known that they get paid for it.	-0,557	Brancoli et al. 2020	-4,1E-02
Donation	5%	0,0039	But local charity reports that bread ends up there, but likely this is only a small part as they don't get paid for donations.	-0,370	Brancoli et al. 2020	-1,4E-03

-4,2E-02

Packaging

Plastic valorisation	Fraction of plastic waste (%)	Fraction of plastic waste (kg)	Comment	Emission factor per kg plastic	Source emission factor	kg CO2e
Ethanol production	95%	0,0015	Packaging is separated from bread before it us further valorized.	0,342	See Appendix 1.5	2,6E-05
Donation	5%	0,00008	When donated, packaging ends up in households and consumer is responsible for waste separation.	-0,961	See Appendix 1.5	-1,4E-03

-1,4E-03

-4,4E-02

or: *-0,0435*

Snon-TBA

Bread

Bread valorisation	Fraction of bread waste (%)	Fraction of bread waste (kg)	Comment	Emission factor per kg bread	Source for emissions factor	kg CO2e
Donation	80%	0,0200	Assuming that supermarkets can donate a larger fraction than suppliers as they are closer to local charity and according to recent reports and interviews don't direct it to ethanol production.	-0,370	Brancoli et al. 2020	-7,4E-03
Anaerobic digestion	20%	0,0050	Assuming that only a small part cannot be prevented or donated and must be discarded with food waste. Here plastic and bread is considered together, as retail does not separate it from the bread.	-0,020 -0,080	44% is lost at pre-treatment and incinerated; substitutes diesel (Brancoli et al. 2020).	-5,6E-05 -1,8E-04

-7,6E-03

Packaging

Plastic valorisation	Fraction of plastic waste (%)	Fraction of plastic waste (kg)	Comment	Emission factor per kg plastic	Source for emissions factor	kg CO2e
Donation	80%	0,0004	When donated, packaging ends up in households and consumer is responsible for waste separation.	0,342	See Appendix 1.5	1,4E-04
Anaerobic digestion	~	~	No plastic has to be treated when it is directed to anaerobic digestion, as it is not separated by the retailer and thus must be removed in pre-treatment. The plastic part is considered above together with bread.	~	Brancoli et al. 2020, 2017	~

1,4E-04

Waste prevention

Prevention type	Fraction of total FU	Fraction total FU (kg)	Comment	Emission factor per kg	Source for emissions factor	kg CO2e
Bread waste prevention	5,2%	0,0520	Based on a total weight (bread) of 1 kg and areturn rate of 2,5%.	-0,660	Brancoli et al. 2020	-3,4E-02
Plastic waste treatment	5,20%	0,0010	Based on a total weight (packaging) of 0,02 kg. The prevention of wastage bread does not prevent the production of packaging, as the reduction of bread volume happens at retail only, when packaging is already manufactured. Packaging of prevented bread waste is sold and thus discarded at the household stage, where separation is up to the consumer.	0,342	See Appendix 1.5	3,6E-04
						-3,4E-02
						-4,1E-02
					or:	*-0,0415*

Appendix 3

Appendix 3 provides an overview of all assessed scenarios.

Type of scenario	Abbreviation	Description
Baseline scenarios	$\boldsymbol{S_{TBA}}$	Scenario for current system with TBA
	$\boldsymbol{S_{non\text{-}TBA}}$	Conceptual scenario without a TBA
Parameter sensitivity tests	$\boldsymbol{S_{pig}}$	$\boldsymbol{S_{non\text{-}TBA}}$ with alternative bread waste treatment (pig feed)
	$\boldsymbol{S_{pack}}$	$\boldsymbol{S_{non\text{-}TBA}}$ with packaging separation by retailers
Scenario alterations	$\boldsymbol{S_{coop}}$	$\boldsymbol{S_{TBA}}$ with increased cooperation
	$\boldsymbol{S_{co\text{-}log}}$	$\boldsymbol{S_{TBA}}$ with co-logistics
	$\boldsymbol{S_{max.collab}}$	Combines $\boldsymbol{S_{coop}}$ and $\boldsymbol{S_{co\text{-}log}}$
Assessment of additional Swedish cities	$\boldsymbol{S_{TBA,\ Gävle}}$	$\boldsymbol{S_{TBA}}$ applied to the city of Gävle
	$\boldsymbol{S_{non\text{-}TBA,\ Gävle}}$	$\boldsymbol{S_{non\text{-}TBA}}$ applied to the city of Gävle
	$\boldsymbol{S_{TBA,\ Göteborg}}$	$\boldsymbol{S_{TBA}}$ applied to the city of Göteborg
	$\boldsymbol{S_{non\text{-}TBA,\ Göteborg}}$	$\boldsymbol{S_{non\text{-}TBA}}$ applied to the city of Göteborg
	$\boldsymbol{S_{TBA,\ Jönköping}}$	$\boldsymbol{S_{TBA}}$ applied to the city of Jönköping
	$\boldsymbol{S_{non\text{-}TBA,\ Jönköping}}$	$\boldsymbol{S_{non\text{-}TBA}}$ applied to the city of Jönköping
	$\boldsymbol{S_{TBA,\ Helsingborg}}$	$\boldsymbol{S_{TBA}}$ applied to the city of Helsingborg
	$\boldsymbol{S_{non\text{-}TBA,\ Helsingborg}}$	$\boldsymbol{S_{non\text{-}TBA}}$ applied to the city of Helsingborg
	$\boldsymbol{S_{TBA,\ Örebro}}$	$\boldsymbol{S_{TBA}}$ applied to the city of Örebro
	$\boldsymbol{S_{non\text{-}TBA,\ Örebro}}$	$\boldsymbol{S_{non\text{-}TBA}}$ applied to the city of Örebro
Parameter sensitivity tests, additional cities	$\boldsymbol{S_{TBA+E,\ Gävle}}$	$\boldsymbol{S_{TBA}}$ applied to the city of Gävle with alternative bread waste management (ethanol)
	$\boldsymbol{S_{TBA+E,\ Göteborg}}$	$\boldsymbol{S_{TBA}}$ applied to the city of Göteborg with alternative bread waste management (ethanol)

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