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Assessing the Potential Climate Impacts and Benefits of Waste Prevention and Management: A Case Study of Sweden

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Abstract: This study employs a life cycle perspective to analyze the carbon footprints of various waste streams, evaluating 52 cases across 26 types of household waste in Sweden, with a focus on waste prevention and management. It demonstrates that while recycling can reduce carbon emissions, prevention could significantly enhance these benefits, with savings ranging from −36.5 to −0.01 kg-CO_2-eq per kg of waste. Notably, Waste Electrical and Electronic Equipment (WEEE), textiles, tires, residual household, and plastic waste are the top five fractions most amenable to prevention on a per mass basis. Further analysis, considering waste volumes, shows that targeted recycling of materials like WEEE, metals, and paper could account for over 80% of potential carbon savings. However, the majority of potential climate impact is attributed to the energy recovery of unsorted (mixed) waste, contributing to more than 90% of total impacts. Redirecting all mixed waste to recycling could triple carbon savings, but focusing on prevention could potentially increase benefits twenty-sevenfold, particularly for waste like WEEE, food, and textiles. This research provides a valuable tool for identifying key areas in waste management to optimize climate benefits and enhance public awareness. However, it advises using local data for precise planning due to inherent uncertainties.

Keywords: LCA; carbon footprint; environmental assessment; household waste; CO_2; municipal waste; recycling

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1. Introduction

The challenge of combating climate change is prominently featured on today's political agenda, with many countries setting increasingly ambitious objectives, for example, the EU strives for zero greenhouse gas emissions by 2050 [1]. Notably, countries like Sweden have set even more aspiring targets, aiming for carbon neutrality as early as 2045 [2]. This urgency is paralleled by a growing emphasis on implementing the Circular Economy, which advocates for resource conservation through waste prevention and sustainable management practices. Integral to this initiative are ambitious waste management objectives, including a 65% material recovery goal for municipal waste by 2035 in EU member states and plans to phase out landfilling for recyclables and organic waste by 2030, as stipulated in the Waste Framework Directive [3].

These climate and circular objectives could potentially converge, as circular waste management strategies, which prioritize waste prevention and recycling, hold the promise of diminishing dependence on high-carbon-footprint virgin materials. Such strategies have the potential to yield "negative" climate emissions by reducing reliance on materials with high environmental footprints [4].

Finding sustainable solutions for circular resource management necessitates systematic analytical methods, such as Life Cycle Assessment (LCA), whose application in waste management still requires further research, Christensen et al. [5] reviewed the results of 350 journal articles, acknowledging the necessity for further application of LCA models in integrated waste management, as well as in policy and strategic development in waste management. A recent study by Bisinella, Schmidt [6] provides practical recommendations for LCA modeling of future waste management systems, emphasizing that climate change impact has the highest priority in the decades to come, due to its political focus. Although LCA is often used for waste management assessments, its coverage of various waste streams or waste prevention measures remains relatively rare. Studies often focus solely on specific waste streams, such as construction demolition waste [7,8], organic waste such as food or garden waste [9,10], and sometimes WEEE [11], rarely considering a comprehensive list of fractions in the same study with the same functional unit. Even for municipal waste, which is probably one of the most common application objects for LCA studies, evaluations mainly are limited to mixed municipal or household waste and the traditional 4–6 fractions or recyclables in the same study [12–15]; however, waste prevention is rarely considered as a strategy in evaluating the performance of waste management [16–19]. The rare studies that consider wider waste streams and waste prevention (e.g., [20]) usually lack concrete user-friendly indicators per mass unit and are limited to comparing different scenarios.

Our study estimates the climate impact of various household waste fractions through a comparative analysis of waste management solutions and waste prevention strategies, aiming to enhance understanding of the importance of prevention versus management for a broad range of waste types. We have developed a comprehensive set of climate impact factors in kg-CO_2-eq per kg of waste that may serve as a versatile screening tool, aiding decision-making, enhancing communication, and fostering awareness among households, organizations, and business. These factors were also applied at the national level in Sweden to demonstrate their potential applications and to better understand the impacts of specific waste volumes. This study stands out for its broad scope, encompassing a diverse range of waste fractions, and for its inclusion of waste prevention measures as an integral part of the decision-making process. Our research is grounded in a case study of Sweden, providing valuable insights into sustainable waste management practices.

2. Materials and Methods

2.1. Scope and Approach

This study evaluates the climate footprints of waste prevention strategies by comparing waste prevention with traditional waste management approaches for 26 waste fractions within Swedish municipal waste management systems.

The study used LCA, based on the attributional approach, which utilizes average data for production, materials, and energy carriers [21]. This approach was prioritized over the consequential approach [21,22] because the study aims to determine the average impact of a product/waste in an existing system rather than tracing the consequences of different decisions forward in time.

The traditional definition of waste prevention (e.g., outlined in the EU Waste Directive [3]) involves actions taken before materials or products become waste to reduce quantity, environmental impacts, or hazardous content. Strategies may include consumption reduction, product lifespan extension, reuse, and many others. In our study, we focused on absolute prevention, avoiding product purchases entirely. Textile reuse was chosen as a specific prevention strategy for just textiles due to its existing practice in the country. The WAMPS (Waste Management Planning System) software was used as a modeling tool. Developed by the Swedish Environmental Research Institute (IVL), WAMPS is a LCA model of waste management systems [15,23].

The climate impact is calculated as kg-CO_2-eq per 1 kg of 26 waste fractions of household waste collected in Sweden. The study assumes the national average waste composition and typical waste treatment technologies.

According to the terminology of Sundqvist and Palm [24], waste prevention scenarios are referred to as “upstream studies”, while traditional waste management scenarios are termed “downstream studies”. The upstream study focuses on products’ raw material extraction, production, and use phases, whereas the downstream study examines all post-use life cycle stages, such as waste collection, transportation, recycling, digestion, composting, incineration, or landfilling.

2.2. System Boundaries

To model waste prevention effects, the study accounts for carbon emissions from raw materials extraction to final waste management. Household transports to procure products are excluded due to the lack of data. The use phase of all packaging waste and plastic waste fractions is neglected due to negligible climate impacts compared to other life cycle stages. The same waste composition was used in the upstream and downstream studies, except for food waste. Food waste could be classified as avoidable (i.e., edible before disposal) and unavoidable (i.e., inedible food parts). In our model, food waste prevention was assumed to concern only the avoidable waste. This excluded the unavoidable food waste from the "upstream" study. All model assumptions are detailed in Appendices A–C.

We modeled waste management as providing two functions—waste treatment and the potential production of secondary materials through recycling. These were assumed to replace virgin material inputs and lower their original environmental loadings minus the loadings of waste conversions. Our model thus implied allocations of emissions between waste management services and raw material replacements. To avoid the subjectivity of allocation [15,25], we applied system boundary extensions using compensatory systems corresponding to life cycle extractions of virgin materials and energy carriers (see Figure 1).

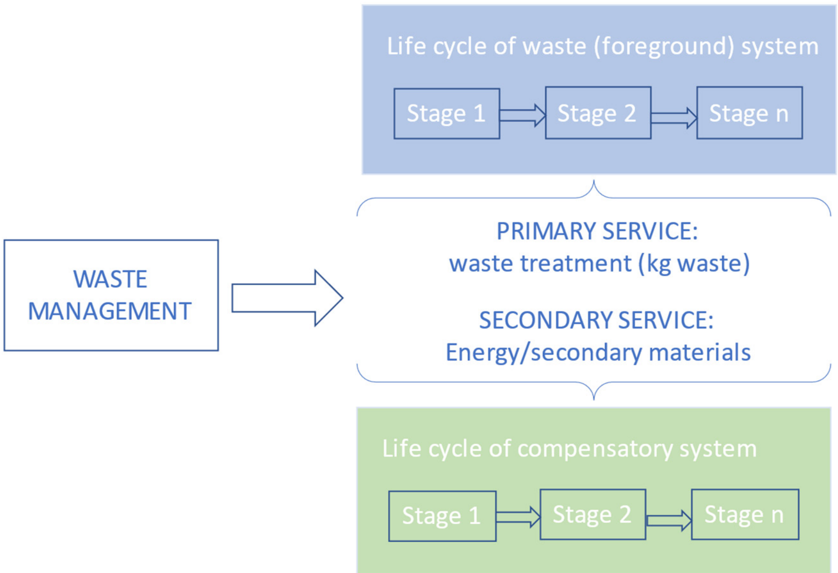


Figure 1. System boundary expansion (based on [15]).

To model the impacts for the upstream study, we equate them to the impacts of the product's life cycle as follows:

$$\mathrm{I}_{prev.} = -\mathrm{I}_{product}, \tag{1}$$

where $I_{prev.}$ represents the impact of waste prevention, which is the potential impact of avoided products and materials by recovered waste fraction, and $I_{product}$ represents the life cycle impacts of newly produced products from extraction to manufacturing, transport, distribution, use, and waste management.

For the downstream study, the impacts are modeled by equating them to the impacts of waste management minus impact credits for replaced products and energy, as described by Miliute and Staniškis [15], based on system expansion:

$$\mathrm{I}_w = I_{wm} - I_{pe}, \tag{2}$$

where I_w is the net impact of waste management; I_{wm} represents the direct impact of the waste treatment activities (collection, recycling, incineration, or landfilling); and $I_{pe,}$ represents the corresponding impacts of replaced products, i.e., energy and materials regenerated in waste treatment (e.g., recycled materials or recovered energy). An environmental gain is achieved when waste treatment operations generate a smaller climate footprint than the footprints of corresponding production of replaced products or energy, usually based on the use of virgin materials.

2.3. Data Sources

For the upstream study, we conducted a comprehensive literature review to gather data from available LCA studies (see Appendices B and C). To ensure data homogeneity across all waste fractions, we assessed whether the reference studies used marginal data (specific data for the background system reflecting the marginal changes induced by waste management) or average data. We prioritized average data or recalculated marginal data into averages based on typical prevailing technologies. Additionally, we examined the characterization factors used in the reference studies to ensure comparability. The characterization factors for climate change were normalized according to the IPCC-2013 method (IPCC 2016).

The reference studies often had data gaps regarding the use phase and transportation. For products where the use phase was deemed to have significant greenhouse gas (GHG) emissions (e.g., food storage, refrigeration, cooking, or the use of electric and electronic equipment (EEE)), we conducted our calculations or utilized data from other countries, adjusting the data to the Swedish electricity mix and its climate impacts. The impacts from the use phase of tires were disregarded due to the lack of reliable data.

In cases where transportation data were unavailable, we assumed that products were manufactured in Sweden, the EU, or China, based on the most plausible locations for different products. These estimates were derived from a literature search and expert opinions, assuming the most common modes of transport and templates for CO_2-eq. emissions per tonne per kilometer of transported products. Detailed information on data assumptions and data sources can be found in Appendices B and C.

2.4. Description of Case Study

Sweden and its waste management system were selected as the case study for our analysis. Sweden is recognized as one of the leading countries in diverting household waste away from landfills. As of 2019, only 0.8% of household waste was sent to landfills, with approximately half undergoing material recovery and the other half undergoing incineration with energy recovery [26]. Certain waste fractions, such as metal and glass packaging or food waste, exhibit significantly higher material recovery rates.

The collection and management of household waste in Sweden are organized based on waste flows, which vary among the 290 municipalities responsible for local waste management planning. Some waste categories, including packaging, paper waste, electronics, and tires, fall under extended producer responsibility (EPR), which is managed separately or in collaboration between industry and municipalities. Approximately 6000 unmanned recycling stations have been established for collecting packaging and paper waste through EPR systems. Curbside collection is also organized for EPR waste in some municipalities. Additionally, all municipalities have at least one central recycling center, totaling 580 centers nationwide, where households and smaller companies can dispose of sorted waste free of charge. These recycling centers accept bulky waste, garden waste, construction and demolition waste, used tires, textiles, white goods and electronics, as well as various hazardous waste fractions.

Approximately 82% of municipalities have systems for separate collection of household food waste as of 2019 [26]. By 2024, all municipalities are mandated to implement systems for the separate collection of all biowaste. This fraction typically undergoes anaerobic digestion with biogas production, where 90% of methane produced is used as vehicle fuel and the remaining 10% is utilized for electricity and heat production. The anaerobic digestion also produces a digestate that is used as fertilizer. Some municipalities also produce compost from food waste. All other mixed waste is exclusively processed in more than 30 operational waste-to-energy plants across Sweden, where it is incinerated [26]. To model the impacts of food waste prevention, we used a weighted combination of anaerobic digestion, industrial composting, and home composting.

Further details regarding assumptions and treatment methods for all modeled waste fractions are provided in Appendices A–C.

3. Results

The modeling results for both waste prevention and typical waste management scenarios are outlined in Table 1. Generally, waste prevention emerges as a more environmentally favorable option compared to other waste management strategies, including separate sorting for material recycling. While there are some climate credits gained from recycling, these credits vary across different waste categories.

As anticipated, waste prevention offers substantially greater carbon benefits compared to any waste treatment method. The variations in carbon savings between prevention and waste management range from −36.5 kg-CO_2eq for WEEE to −0.01 for construction materials, with WEEE, textiles, tires, and residual household and plastic waste standing out as the top five fractions most amenable to prevention. Since the outcomes are also affected by the prevalence of these fractions within the analyzed system, decision-makers may need to consider specific actions.

To exemplify the potential impacts of waste quantities and to gauge the disparity between waste prevention and waste management, we conducted supplementary analyses at the national level, utilizing actual household waste data from the year 2020. Our methodology involved (1) establishing a baseline scenario using 2020 data and formulating two theoretical scenarios in which residual and combustible mixed household waste would either (2) be separately collected and recycled, or (3) prevented instead of recycled. Detailed descriptions of the scenarios and results are presented in Appendix D and Table A4.

When total waste volumes are considered, as in the baseline scenario, waste management practices yield notable carbon benefits (Figure 2). Specifically, WEEE, metals, and paper waste—all of which were separately collected and directed to recycling—along with wood waste designated for energy recovery, together account for more than 80% of the observed CO_2-eq benefits. In contrast, the positive climate impact, in terms of emissions, is largely due to residual household waste. Indeed, more than 90% of these emissions result from the energy recovery of mixed household waste and mixed combustible bulky waste (Table A4).

Separate collection and recycling of all residual waste, as opposed to energy recovery, could substantially enhance carbon benefits. The "Recycling Waste" Scenario 2 detailed in Appendix D suggests a potential tripling of these benefits compared to baseline. The contribution of waste fractions (Figure 3a) remains similar to the baseline (Figure 2). WEEE, wood, metal, and paper continue to be the most significant fractions, though plastic waste and textiles become increasingly prominent due to their considerable shares in residual fractions (Figure 3a). Despite clear benefits with increased recycling, however, prevention measures outpace recycling in terms of efficacy. As modeled in Scenario 3, preventing the same volume of waste that is recycled in Scenario 2 could amplify climate benefits by over twenty-sevenfold, as substantiated by Table A4. Notably, WEEE, food, and textiles are the primary contributors to these benefits, potentially yielding up to 80% of total negative emissions (Figure 3b).

While our results are subject to a range of uncertainties described in the following sections, the data unequivocally endorse waste prevention as a strategy. Its implementation at the national level is shown to yield significant climate advantages, both on a per mass unit basis and in aggregate.

Table 1. Climate impacts and benefits of different waste fractions induced by waste prevention or management, kg-CO_2-eq per kg of waste. The column labeled "Waste Prevention" contains the main results from Equation (1), and the column labeled "Waste Management" corresponds to the results from Equation (2). The climate impact factors listed in Table 1 are expressed per mass unit of waste and can be extrapolated to total carbon footprints if the overall volumes of each waste fraction are known.

Waste Fraction	Waste Prevention, kg-CO_2-eq	Waste Management, kg-CO_2-eq	Difference, kg-CO_2-eq	Rank	Dominant (Typical) WM (incl. Pre-Sorting)
WEEE	−38	−1.5	−36.5	1	Recycling (78%), incineration (13%), landfilling (9%)
Textiles (reuse)	−25	−7.1	−17.9	3	Reuse (80%), incineration (20%)
Textiles (recycling)	−25	−1.2	−23.8	2	Recycling (80%), incineration (20%)
Tires	−3.6	−0.1	−3.5	4	Recycling (40%); used for fuel (60%)
Residual household waste	−2.3	0.2	−2.5	5	Incineration (100%)
Metal packaging (20% Al; 80%Fe)	−2.2	−1.8	−0.4	16	Recycling (100%)
Food waste (anaerobic digestion)	−2.2	−0.1	−2.1	10	Anaerobic digestion (100%)
Food waste (industrial composting)	−2.2	0.03	−2.2	8	Industrial composting (100%)
Food waste (home composting)	−2.2	0.07	−2.3	7	Home composting (100%)
Plastic packaging	−2.1	−0.6	−1.5	11	Recycling (60%), incineration (40%)
Plastic (not packaging, bulky)	−2.1	−0.6	−1.5	11	Recycling (75%), incineration (25%)
Combustible bulky waste	−2	0.3	−2.3	6	Incineration (100%)
Example of bulky waste: sofa	−2.1	0.1	−2.2	9	Incineration (100%)
Metal scrap (bulky) (steel: 50% virgin, 50% recycled)	−1.9	−0.9	−1.0	14	Recycling (100%)
Flat glass	−1.2	−0.2	−1.0	13	Recycling (80%), incineration (20%)
Paper waste (non/packaging) (within EPR)	−1.1	−0.9	−0.2	20	Recycling (85%), incineration (15%)
Glass packaging	−0.8	−0.35	−0.5	15	Recycling (90%), incineration (10%)
Corrugated cardboard	−0.6	−0.3	−0.3	17	Recycling
Office paper	−0.4	−0.2	−0.2	21	Recycling (85%), incineration (15%)
Paper packaging	−0.5	−0.2	−0.3	17	Recycling (85%), incineration (15%)
Plasterboard	−0.3	−0.05	−0.3	19	Recycling (90%), landfilling as reject (10%)
Wood waste, not impregnated wood	−0.2	−0.4	0.2	22	Incineration (100%)
Construction materials	−0.01	0	−0.01	23	Use as road materials for landfills (100%)
Non-combustible/inert waste	−0.01	0.1	−0.4	16	Use as road materials for landfills (100%)
Garden waste	n/r	0.1			Open windrow composting (100%)

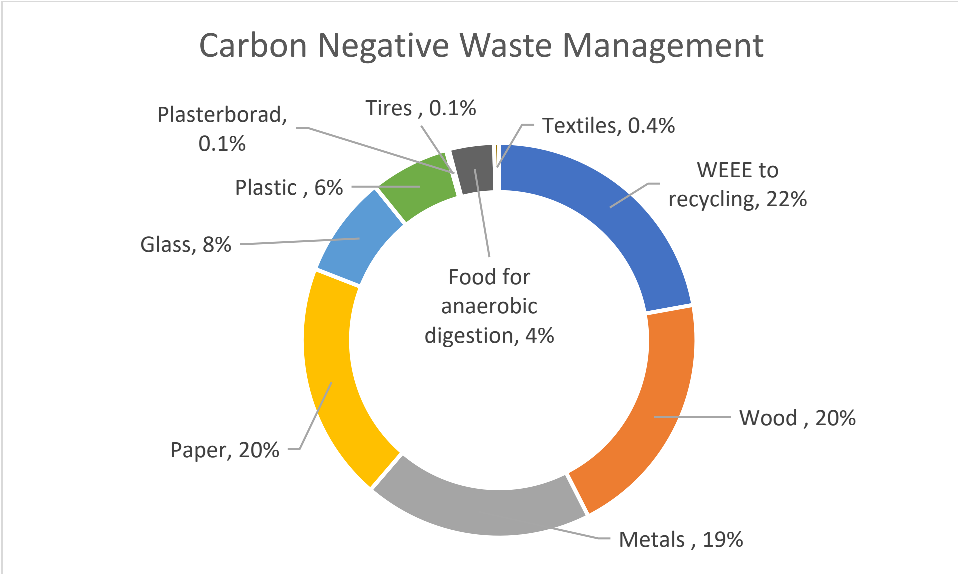
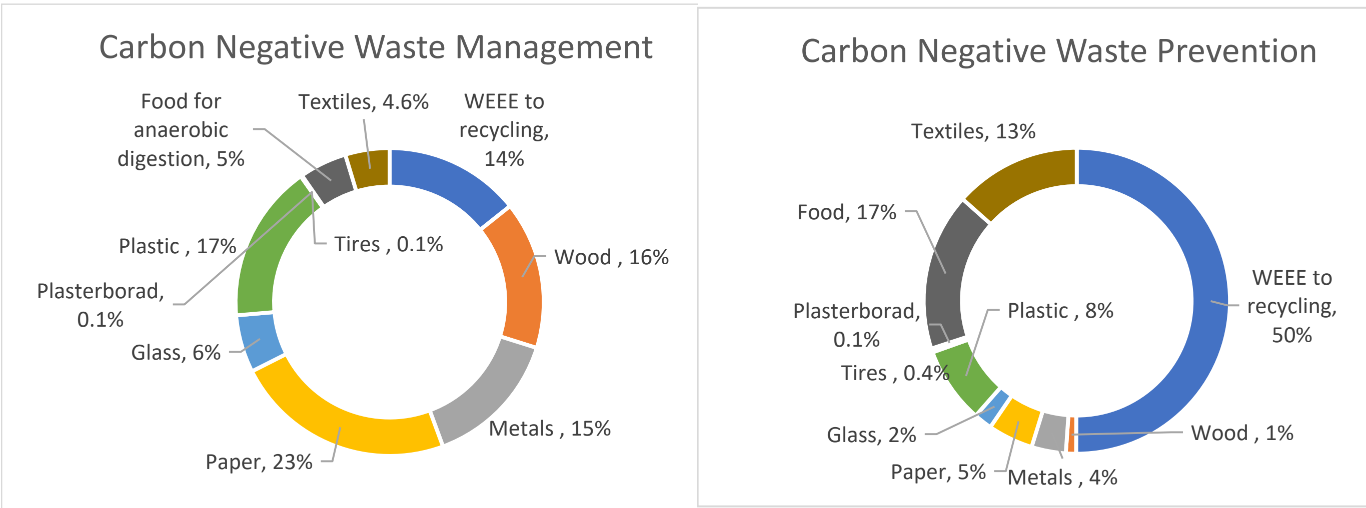


Figure 2. Contribution of waste fractions to carbon benefits in baseline Scenario 1.



(**a**) (**b**)

Figure 3. Comparative contribution of waste fractions to carbon benefits based on waste strategy This figure illustrates the differing impacts of waste fractions on carbon benefits depending on the waste management strategy implemented: (**a**) maximum recycling and other carbon-negative waste management practices as outlined in Scenario 2; (**b**) maximum prevention as modeled in Scenario 3.

Below, we provide a brief overview and explanation of the estimates for selected key waste fractions.

3.1. *Waste Electrical and Electronic Equipment (WEEE)*

The most significant climate savings per unit of mass are observed from the prevention of WEEE. Certain electronic products, such as mobile phones, laptops, and screens, exhibit climate savings per weight of waste that are up to 10–20 times higher compared to other waste categories. However, the extent of savings within the WEEE category varies considerably (See Table 2).

Table 2. Results of the upstream study for electronics per kilogram and per unit. These results are utilized in modeling the factors for Equation (1).

Product Group	Waste Prevention, kg-CO_2-eq/kg	Waste Prevention, kg-CO_2-eq/Piece
Mobile phone	−415	−68
Laptop	−199	−252
Screen	−175	−989
Electric drill	−4.3	−10
Refrigerator	−7.5	−768

Climate gains from the prevention of Waste Electrical and Electronic Equipment (WEEE) vary depending on the product group. WEEE items containing larger quantities of integrated circuits (ICs) typically have significant climate impacts during the production stage, involving rare metals, ultra-pure chemicals, and clean-room environments. These factors result in large climate footprints, sometimes exceeding those of the use phase. Conversely, products like drills or refrigerators may have fewer ICs but are power-intensive during use. Research by Ercan et al. [27] indicates that approximately 80% of the life cycle greenhouse gas (GHG) emissions of mobile phones occur during production. In contrast, about 67% of the climate footprint of a manual vacuum cleaner is attributed to the use phase [28]. This suggests that the climate benefits of preventative strategies, such as re-use or remanufacturing, may vary among different WEEE groups. For some products, extending their lifetime may yield limited benefits due to technological advancements and improved energy efficiency during use.

Overall, waste prevention of WEEE can yield up to 25 times more climate benefits on average compared to their recycling (See Table 1), particularly for products with significant climate footprints in the production phase. Merely recycling materials at the end of life typically cannot fully offset the impacts from production and use phases. Recycling ICs remains technically challenging and economically unfeasible. Achieving economies of scale is difficult when dealing with large quantities of WEEE, especially considering that valuable rare metals are present in minute amounts. Typically, WEEE undergoes manual or automatic dismantling followed by shredding, separating bulky metals (e.g., iron, steel, and copper) and some precious metals (e.g., silver, gold, and palladium) for recycling, while other materials such as plastics are incinerated (See Table 3). Consequently, rare earth metals are largely lost during recycling processes, with only a portion of precious metals like gold and platinum being recovered.

Table 3. Material content and its management in WEEE: various small electronics, refrigerators, and freezers. Based on [29].

Materials	Various Small Electronics	Refrigerators and Freezers
Iron (material recycling)	40%	64.5%
Copper (material recycling)	5%	2.5%
Aluminum (material recycling)	4%	3%
Plastic (material recycling)	18%	9%
Glass		0.5%
Silver (material recycling)	0.012%	
Gold (material recycling)	0.001%	
Palladium (material recycling)	0.0005%	
Other metals (material recycling)	10%	
Other combustible materials (energy recovery)	10%	19%
Other non-recyclable or non-combustible materials (landfilling)	13%	1.5%
Total	100%	100%

3.2. Textiles

Textiles were the only waste fraction for which we examined not only absolute prevention but also reuse as a specific waste prevention category, in addition to recycling. Absolute prevention entails avoiding consumption and production. Reuse involves utilizing products, such as textiles, for their intended purpose without them becoming waste, typically involving collection and sorting operations. Textile recycling is primarily viewed as a recovery operation where waste materials are reprocessed into new textile products, involving waste collection and sorting operations.

Absolute prevention yields greater climate benefits than both reuse and recycling. However, reuse proves to be more beneficial than recycling, as it avoids upstream impacts associated with cotton cultivation, oil extraction for synthetic textiles, production, and distribution. The textile consumption in Sweden contributes approximately 4.2 Mt-CO_2-eq annual emissions, with the production phase accounting for roughly 80% of the total climate impact of textiles used in the country [30]. Within the production phase, the use of electricity constitutes the dominant contributor, comprising 43% of the total impact. Textiles consumed in Sweden are typically manufactured in countries such as China, Bangladesh, Turkey, or India, where the electricity mix has a high climate footprint. The use phase, encompassing washing and drying, contributes only 7% of the entire life cycle impact due to Sweden's greener electricity mix.

Furthermore, the climate benefits of textile reuse hinge on the product replacement rate, i.e., the extent to which consumers eschew purchasing new garments. In our study, we assumed a moderate 60% replacement rate, based on a customer survey conducted in Sweden by Farrant, Olsen, and Wangel (2010). Other reference studies have reported replacement rates ranging from 30% to 100% [31]. According to Sandin and Peters [31], many studies employ a 100% replacement rate without justification, potentially resulting in overestimated environmental benefits. Nevertheless, our estimates indicate that even at low substitution rates, reuse remains more advantageous compared to recycling (see Figure 4).

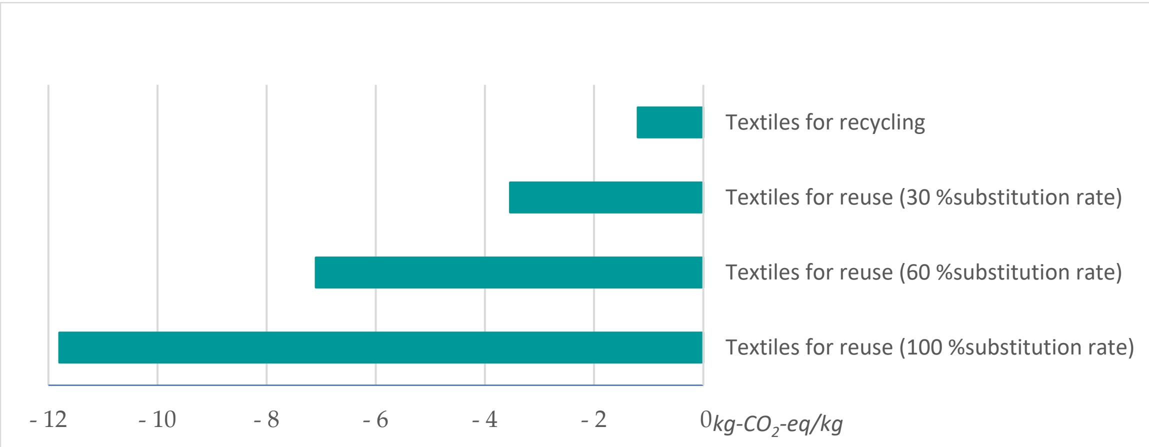


Figure 4. Climate benefits for circular solutions for textiles in relation to the different assumptions on the substitution rate.

3.3. Other Waste Streams

For other waste fractions, waste prevention yields carbon credits within a narrower range per mass unit—from −2 to −3 kg-CO_2-eq/kg of waste for food, plastic, residuals, and metals, to minus 0.01–1 kg-CO_2-eq for waste fractions containing paper, glass, and inert construction materials. Waste prevention, and partly recycling, can generate higher climate benefits when waste fractions contain higher shares of fossil carbon content (e.g., tires and plastics).

The prevention of tire waste ranks as the third largest climate benefit in our study, with over 3 kg-CO_2-eq/kg of waste. However, in our analysis, we neglected the use phase, treating tires as a separate product rather than as part of a car. If emissions from car use (direct fuel use and the fuel value chain) were allocated to tires, the climate footprint would increase by more than 1000 kg-CO_2-eq/kg of waste tires. In Sweden, separately collected waste tires are primarily incinerated in the cement industry, replacing coal or natural gas, which yields climate credits. Material recycling, such as filling layers for artificial turfs, also contributes to climate gains. In our study, we assumed that tire recycling replaces three filling materials in equal shares—expanded cork, EPDM, and TPE.

The prevention of metal packaging waste compared to bulk metal scrap results in slightly higher climate gains. This difference is mainly due to the higher share of aluminum (around 20%) in metal packaging, where replaced virgin aluminum yields significant climate savings. Overall, there is about a twofold difference in climate benefits between metal waste prevention and recycling. This discrepancy is much higher for other waste fractions, such as plastic, because relatively more homogeneous and pure metal fractions are available for recycling compared to plastics. For metals, we assumed a reject rate of only 5–10%. In the case of plastic waste, for instance, only around 40% of Swedish plastic packaging waste leaves the sorting facility for recycling, while other plastics undergo energy recovery [32].

The least carbon savings are achievable from the prevention and recycling of inert materials, such as plasterboards, mixed inert construction materials (crushed tiles and similar), or wood.

3.4. Recycling Versus Other Treatment Methods

The recycling of all separately collected waste fractions in this study yields climate benefits due to the simulated substitution of virgin materials and fossil energy. Another significant waste management practice for mixed household waste and bulky combustible waste in Sweden is incineration with energy recovery, which also brings some climate credits, although to a lesser extent compared to recycling of other fractions. This is because the heat and power produced replace relatively green energy mixes in both electricity and thermal energy. Sweden's electricity mix is dominated by nuclear and hydro power, and much of the thermal energy is produced from biomass, often including co-generation. Only the incineration of tires in the cement industry brings climate credits by replacing fossil energy carriers.

Regarding the climate effects of food waste treatment in Sweden, it involves a combination of two methods: anaerobic digestion (dominant) and composting in large-scale industrial facilities or (marginal) home-based composting. The prevalence of these methods varies among different municipalities. In our study, instead of using a national average, we simulated all three scenarios. The results indicate that only anaerobic digestion brings climate gains from the production of biogas and the substitution of fossil fuels. Meanwhile, virtually no climate savings can be expected from industrial and home-based composting, which instead generate climate impacts from methane and CO_2 emissions during composting and potentially weak substitution of artificial fertilizers. Typically, much of the produced compost does not effectively replace artificial fertilizers and is instead used as topsoil filler in closed landfills or for maintaining public spaces, such as municipal parks and other green areas [33].

4. Discussion

4.1. Uncertainties

The results of this study, like any other LCA studies, may be subject to data uncertainties and assumptions made throughout the analysis. For instance, in cases where specific data were unavailable, averages were assumed for parameters such as waste compositions across municipalities, material recovery rates, recycling losses, the degree of material downgrading, or the origin of manufactured products. Two main methods were used to address uncertainties: sensitivity analysis and data quality assessment based on expert judgment. When several data sources were equally reliable, sensitivity analyses were conducted. When different data sources were available, we relied on established data sources or Swedish-specific data sources. In cases where data were scarce or uncertain, expert judgment was used to make qualitative assessments of data reliability. Below, we present some examples of selected fractions where uncertainties potentially were the greatest.

Waste composition and the origin of substituted materials posed significant uncertainties, particularly for metals (see Figure 5). In Sweden, all waste metal packaging is collected in the same containers and later separated centrally into ferrous and non-ferrous fractions. According to the Swedish Extended Producer Responsibility (EPR) packaging organization, the national average metal packaging fraction consists of 80% steel and 20% aluminum [34]. However, the climate implications of prevention and recycling depend on the actual composition in specific cases. For example, the climate benefits from preventing aluminum waste are four times larger compared to those from preventing steel packaging waste (see Figure 5).

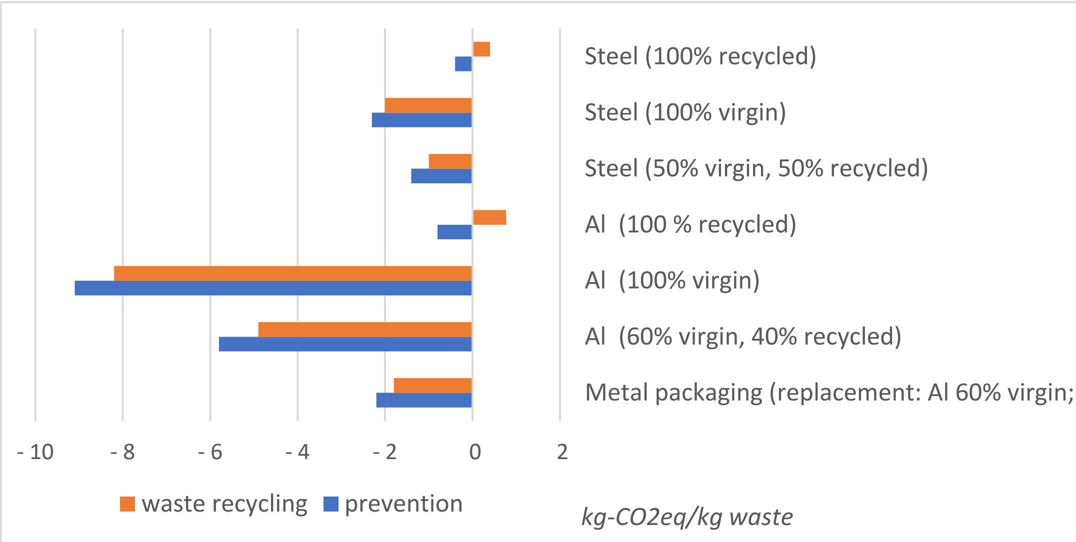


Figure 5. Climate benefits for metal packaging in relation to the different assumptions on the substituted materials.

The results for metals are also sensitive to assumptions regarding the share of recycled materials in avoided or replaced materials. Virgin aluminum and steel have significantly higher climate impacts compared to recycled materials. However, total climate gains from waste prevention and recycling are almost linearly proportional to the share of recycled material in packaging. In our baseline scenario, we assumed moderate percentages for recycled materials: 40% recycled and 60% virgin aluminum, and 50% recycled and 50% virgin materials for steel (see Figure 5). If recycling were to substitute materials with 100% recycled content, the downstream model would result in climate burdens instead of gains due to the climate impacts in collection, sorting, and recycling chains. However, this scenario is unrealistic since virgin materials are always added in relatively high proportions to existing input material stocks.

The results for textiles are also sensitive to assumptions regarding the substitution of new textiles. The outcomes of the sensitivity analysis were previously discussed and presented in Section 3.2. We opted for a moderate substitution rate for the main results. Nevertheless, our findings indicate that even with lower substitution rates, reuse remains more advantageous compared to recycling (see Figure 4).

Results for WEEE in both the upstream and downstream studies are subject to three significant data-related uncertainties: variation in carbon footprints, diversity in composition of materials, and material recovery efficiency.

There is a notable variation in carbon footprints within and across different WEEE product groups. For example, carbon life cycle intensities for different smartphone models ranged between −16 and −110 kg-CO_2-eq per product [28]. This variability is influenced by factors such as average life spans, manufacturing technologies, product complexity, and geographic locations (energy mixes). We compared case-specific data to average data available from LCA studies, prioritizing more Sweden-relevant studies for cases with large differences.

Material composition in WEEE varies at both the product and product group levels, adding to the uncertainty. Data about the composition of WEEE were aggregated based on several product categories, but considering WEEE composition at a product level would provide more certainty. Unfortunately, such granular data were not available at the time of the study.

Material recovery efficiencies from WEEE are also diverse, further contributing to uncertainties in the results. Our assessment of the climate effects of WEEE recycling relied on the material composition of different EEE categories rather than specific statistics on local management characteristics, such as recycling efficiencies.

The lack of data regarding the composition of other waste fractions, such as plastic, food, and bulky waste, also reduces result certainty. Instead of relying solely on the Swedish average, we utilized specific data from one or a few case studies for these waste fractions. However, more recent compositional studies for waste would enhance the reliability of estimates. Data on life cycle carbon intensities for certain product categories, such as electronic waste and food waste, are particularly important for improving the accuracy of future assessments.

4.2. Support for Decision-Making and Potential Limitations

The results of this study serve as a valuable tool for identifying hotspots or areas with the highest climate benefits within the local waste management system. They can effectively communicate the importance of waste prevention and the benefits of household waste sorting to raise public awareness. However, caution should be exercised in using the study for concrete local waste management planning due to the discussed uncertainties. Utilizing local-specific data is recommended for more accurate planning.

The study focuses solely on one environmental aspect, climate change, while other local environmental impacts may also be significant. For instance, the presence of hazardous substances in WEEE highlights the importance of eco-toxicity considerations if waste is not properly managed. Similarly, factors such as toxic contamination and water use are critical for textiles, while microplastic contamination is relevant for plastic and tire waste.

It is worth noting that the study only considers prevention as an absolute decrease in consumption, and in the majority of cases excludes other prevention types such as reuse, remanufacturing, and eco-design. While the upstream study provides insights into which fractions could benefit most from extended lifetime measures, caution should be exercised in quantifying these due to high uncertainties. Additionally, qualitative prevention measures focusing on reducing hazardousness and related environmental impacts should be considered in future studies to enhance waste management system planning.

Moreover, an absolute reduction in primary consumption may not always lead to decreased climate emissions in practice due to potential rebound effects. The study highlights the importance of behavioral economics in understanding such effects. Future research should delve into rebound effects of waste prevention measures, an area that currently lacks sufficient exploration, especially concerning waste prevention measures' effects on household consumption.

Additionally, an absolute reduction in primary consumption may not always result in decreased climate emissions in practice due to the potential rebound effect [35]. Issues related to behavioral economics play a crucial role here. For example, reused products are often cheaper, leading consumers to save money that could be redirected towards additional consumption of other goods and services, known as the second-order rebound effect [35].

Similarly, approximately 30% of food waste in Sweden is avoidable and could be relatively easily prevented [36]. However, if households plan their meals more efficiently and waste less food, they may save money that could be spent on activities with higher climate impacts, such as trips. Therefore, the climate benefits ultimately depend on how households reallocate potential savings and the extent to which rebound effects are influenced by price elasticities of additional goods and services. Studies on the rebound effects of household consumption, particularly regarding the effects of waste prevention measures, are still relatively limited.

5. Conclusions

This study offers 52 carbon footprint estimates for waste prevention and dominant waste management measures across 26 household waste fractions in Swedish municipalities. The paper underscores the significant carbon benefits of waste prevention through reduced consumption compared to any waste treatment method, including recycling.

The variations in carbon savings between prevention and waste management range from −36.5 kg-CO_2eq for WEEE to −0.01 for construction materials per kg of household waste, with WEEE, textiles, tires, and residual household and plastic waste standing out as the top five fractions most amenable to prevention.

The results of applying these estimates to total household waste volumes in the Swedish context indicate that waste management holds potential for carbon savings. Specifically, WEEE, metals, and paper waste, separately collected and directed to recycling, along with wood waste designated for energy recovery, account for over 80% of observed CO_2-eq benefits. Conversely, the positive climate impact, in terms of emissions, largely stems from residual household waste, with over 90% of emissions resulting from the energy recovery of mixed household waste and combustible bulky waste.

Modeling scenarios indicate that redirecting all residual mixed household waste to separate collection and recycling, rather than energy recovery, could triple total waste management benefits. However, preventing the same amount of waste could amplify climate benefits by over twenty-sevenfold. Notably, WEEE, food, and textiles are primary contributors to these benefits, potentially yielding up to 80% of total negative emissions.

The climate gains from recycling vary considerably among different waste material fractions depending on primary material substitution rates and the origin of primary materials. Incineration and landfilling are consistently less favorable options from a climate perspective compared to separate sorting and material recycling across all waste categories. Although based on the Swedish waste management context with a relatively green national energy mix, the study's findings apply to other countries, where higher carbon footprints of background energy systems may lead to even greater climate gains from waste prevention or recycling.

The study provides decision-makers with a valuable tool for communication and quick screening of waste management systems for identifying hotspots or areas with the highest climate benefits within the local waste management system. It can effectively communicate the importance of waste prevention and the benefits of household waste sorting to raise public awareness. However, caution should be exercised in using the study for concrete local waste management planning due to the discussed uncertainties. Utilizing local-specific data is recommended for more accurate planning.

Future research should focus on data quality for life cycle carbon footprints of complex products like electronic equipment, and conducting more detailed compositional studies, particularly at the product level, for waste streams like food, textiles, and bulky waste is essential. While this study focuses solely on carbon footprints, it is crucial to acknowledge the significance of other environmental impact categories such as eutrophication, acidification, photochemical oxidation, and toxicity, among others. Future research should also broaden the scope of environmental indicators beyond climate change, consider additional prevention measures, and delve into the rebound effects of waste prevention measures.

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Appendix A

Table A1. Modeled substances and compositions in the "upstream" and "downstream" studies.

Waste Fraction	Modeled Substances/Composition	
	Upstream Study	Downstream Study
Food waste	Avoidable food waste composition: Meat 10% (pork 5%, beef 2.5% poultry meat 2.5%), bread 15%, dairy products 3% (cheese 2.4%, milk, filet, yogurt 0.3%, cream 0.3%), vegetables and fruits 37% (carrot 4.1%, onion 4.1%, tomato 4.1%, cucumber 4.1%, lettuce 4.1%, broccoli 4.1%, apple 4.1%, orange 4.1%, melon 4.1%), processed food 27% (pasta 9%, rice 9%, potatoes 9%), other 8% [37].	Generic food waste composition (Swedish average) based on generic data in WAMPS
Residual (mixed household waste)	Waste shares: food 29.1%, green 2.8%, paper waste 2.8%; packaging: paper 9.8%, plastic 13.5%, glass 2.5%, metal 1.6%, textiles 3.6%, other non-combustible 5%, other combustible 25%, hazardous waste 0.1%, batteries 0.05%, WEEE 0.35%. Source: Avfall Web, 2016.	
Paper packaging	100% paper packaging	
Plastic packaging	50% hard PE, 50% soft PE (assumption).	
Metal packaging	20% aluminum, 80% steel [34].	
Glass	100% glass packaging	90% glass packaging, 10% combustible material (as a reject)
Wastepaper	100% newspaper	
WEEE	ICT products 12%, fridges/freezers 20%, other 30%, diverse electronics 38%. Source: [38]	Household appliances: large 46%, small 5%, ICT and office 12%; home equipment (TV, A/V) 23%. power tools 3%, toys, leisure and sports 1%. other: 15% [38].
Office paper	100% office paper	100% paper packaging.
Corrugated cardboard	100% corrugated cardboard.	100% paper packaging.
Textiles	Cotton 37.4%, polyester 57.4%, other (viscose) 5.4% [39].	
Metal scrap	100% steel	
Plasterboard	100% plasterboard (assumption)	
Flat glass	Window glass without frame: 80%; wood: 15% PVC frame: 5% (assumption).	
Plastic (not packaging)	PP 28%; HDPE 7%; LDPE 6%; PET 3%; PVC 7%. Other 49%—aggregated weighted average of the above [40].	50/50% of HDPE/LDPE.
Tires	100% tires.	100% tires.
Wood waste, not impregnated wood	100% wood material	100% wood material
Combustible bulky waste	20% PE plastic, 20% mixed paper, 20% garden waste, 20% wood, 10% other combustible, 10% non-combustible (assumption).	
Construction materials and non-combustible, inert waste	Assumption: 100% inert material (50% concrete, 50% soil/sand).	

Appendix B

Table A2. Main assumptions and specific calculations of the "upstream" study.

Waste Fraction	Data Sources, Including Main Assumptions
Mixed municipal waste	Weighted values based on household waste composition in Sweden and prevention values for different materials as defined/calculated below.
Food waste prevention	Based on composition of "avoidable" waste and weighed values for food waste products based on [41], includes the life cycle from cradle to retail, manufacturing in Sweden and abroad, based on market 2011–2015, but excludes use phase [41] The use phase has been additionally calculated based on an average household with 1800 kWh/y (freezer and refrigerator) and 800 kWh cooking equipment [42]. Annual food consumption 267 kg/pers or 720 kg/household. Electricity for cooking −2.5 kWh/kg food. Climate intensity of electricity in Sweden: 0.11 kg-CO_2-eq/kWh [43].
Paper packaging	Calculations/data sources: average value of 10 different paper products (source: [44], additional calculation of transports based on Thinkstep (2018). Assumptions: Manufacturing in Sweden; transport within Sweden (500 km); use phase is considered insignificant, therefore excluded.
Plastic packaging	Calculations/data sources: based on material production from PlasticsEurope (2014) (Coinvent database) with 10% additional GWP from manufacturing (assumption) and transports within Europe (based on Thinkstep (2018). Assumptions: manufacturing in Europe, transports within Europe (1500 km), assumption that manufacturing corresponds 10% of impact of material production; use is considered insignificant and excluded.
Metal packaging	Calculations/data sources: based on the European metal market data [45–47], incl. transports within Europe, based on Thinkstep (2018). Assumptions: transports within Europe (1500 km); use phase is considered insignificant, therefore excluded.
Glass packaging	Calculations/data sources: Impacts of container glass from cradle to consumer of glass packaging in Europe (mixed products, EU production, recycling rate 7%, all transports included, inventory database Ecoinvent v3.5. Assumptions: use phase is considered insignificant and excluded.
Waste paper	Calculation/assumptions: average of 4 products (newspapers and magazines) manufactured and used in Sweden [48–50]. Assumptions: use phase considered insignificant and excluded.
WEEE	Calculation/data sources: weighted values on composition) of 5 WEEE products is for refrigerators [51], smart phones [27] laptops computers—3 average products [52,53], screens [53]; electric drill [54]. Use phase electricity consumption based on Swedish electricity [43].
Textiles	Calculation/data sources: based on total GHG emissions from Swedish consumption of textiles [30] with adjustments (we excluded 14% of CO_2 eq. for consumer transport. Data on the total amount of textiles consumed in Sweden—[55] and population size—[56].
Office paper	Calculations and sources: Assumed Swedish production (cradle to the gate) including transportation within Sweden (internal IVL's data and Thinkstep (2018) Assumptions: use phase considered insignificant and neglected.
Corrugated cardboard	Manufacturing cradle to the gate in Europe [57] including transportation of average product based on Thinkstep (2018). Assumptions: use phase is considered insignificant and neglected.
Metal (bulky)	Low alloy cradle to gate steel production in Europe (Ecoinvent database); average transportation within Europe (Thinkstep (2018): the use phase is considered insignificant, therefore excluded.
Plasterboard	Based on the data from the main Swedish producer (Gypro) [44]; use phase is considered insignificant and neglected.
Flat glass	Flat glass production in Europe (Gabi database); for PVC frame production—[44]; transportation of average product based on Thinkstep (2018); use phase is considered insignificant and neglected.

Table A2. *Cont.*

Waste Fraction	Data Sources, Including Main Assumptions
Plastic (not packaging)	Material production in Europe based on PlasticsEurope (2014); manufacture of packaging—10% of primary production (assumption); transportation average product within Europe (Thinkstep (2018)); use phase is considered insignificant and neglected.
Tires	50% production in Europe [58] and 50% in China [59] Transportation of average product based on Thinkstep (2018). Assumptions: average weight 9.5 kg; the use phase was excluded.
Wood waste, not impregnated wood	Fiber-board production data—[44]; use phase is considered insignificant and neglected.
Combustible bulky waste	Average of the different materials and their impact. *Assumptions* 20% PE plastic, 20% mixed paper, 20% garden waste, 20% wood, 10% other combustible, 10% non-combustible
Construction and non-combustible, inert waste	Assumption: impacts of mixed sand/soil and concrete

Appendix C

Table A3. Main assumptions and specific calculations for the" downstream" study.

Waste Fraction	Data Sources, Including Main Assumptions
Mixed municipal waste	WAMPS (v.2019) modeling. Main input data and assumptions: Produced energy (in co-generation plants): 85% district heating, 15% electricity; Replaced products/energy: district heating is biofuel and electricity production replaces the Swedish electricity mix.
Food waste to anaerobic digestion	WAMPS (v.2019) modeling. Main input data and assumptions: All 100% are directed to anaerobic digestion; produced energy is used: biogas for vehicles (90%) and district heating (10%); digestate/bio-fertilizer products replace the equivalent nitrogen and phosphorous compounds in mineral fertilizers; transports within Sweden.
Food waste to ind./home composting (same for paper packaging)	WAMPS (v.2019) modeling. Main input data and assumptions: All 100% are directed to central composting plant; the produced compost replaces the equivalent nitrogen and phosphorous compounds in mineral fertilizers; transports within Sweden.
Plastic packaging	WAMPS (v.2019) modeling. Main input data and assumptions: All 100% are sent to material sorting of which 25% are rejects (go to energy recovery), and 75% undergo material recycling; the collection transports within Sweden, 50% of centrally pre-sorted materials are assumed to be transported to Germany for further recycling.
Metal packaging	Mixed data from literature on emissions from primary and secondary material production in Europe and Sweden [45–47,60].
Glass packaging	Based on data from the Nordic region [61].
Waste paper	WAMPS (v.2019) modeling. Main input data and assumptions: All 100% are sent to material sorting, 15% are rejects (energy recovery), and 85% undergo material recycling with transports within Sweden.
WEEE	WAMPS (v.2019) modeling for the handling of the reject by incineration and landfilling. Main input data and assumptions: WEEE treatment (of 100% separately collected, after dismantling and shredding) with 75% recycling and 13% incineration with energy recovery. CO_2-eq data is from the recycling is based on maximum possible savings due to material content [62].
Textiles (recycling)	WAMPS (v.2019) modeling for the handling of rejects by incineration. Main input data and assumptions: separately collected waste textiles—80% recycling and 20% (reject) by energy recovery. CO_2-eq data is from recycling (avg. of different recycling technologies) [63,64].
Textiles (reuse)	Main input data and assumptions: separately collected waste textiles—80% reuse (reject) by energy recovery. Modeling based on [63,64] and WAMPS (v.2019)

Table A3. *Cont.*

Waste Fraction	Data Sources, Including Main Assumptions
Office paper and corrugated cardboard	WAMPS (v.2019) modeling. Main input data and assumptions: All 100% are sent to material pre-sorting of which 15% is reject (energy recovery), and 85% (material recycling; same emissions from recycling process as with paper packaging, but assumed to replace office paper produced in Sweden.
Metal scrap (bulky)	Emissions from primary and secondary material production in Europe and Sweden (Eco-invent 3.1) and WAMPS (v.2019) modeling for the rejects.
Plasterboard	Data on virgin plasterboard based on the LCA study [65] on plasterboard but adjusted according to the Swedish electricity production and transports. Waste operations based on interview [66] on energy use, transports and the reject modeled with WAMPS. Main input data and assumptions: 90% recycling and 10% landfilling (rejects).
Flat glass	WAMPS (v.2019) modeling and Hillman, Damgaard [61]. Main input data and assumptions: 80% glass (for recycling), same as glass packaging, 15% (wood) and 5% plastic, to energy recovery.
Plastic (not packaging)	WAMPS (v.2019) modeling. Main input data and assumptions: 100% is sent to material sorting of which 25% is reject (goes to energy recovery), and 75% undergoes material recycling. Collection transports within Sweden, 50% of centrally pre-sorted materials are assumed to be transported to be transported to Germany for further recycling.
Tires	WAMPS (v.2019) modeling of energy recovery; recycling-literature based [67]. Main input data and assumptions: Treatment of separately collected tires [68]: 40% recycling (10% granulated tires used as fill layers in artificial turf and replace fossil-based alternatives *, 30% other use (e.g., used as elements of playground or road marking; and replaces wooden materials); 60% for energy recovery: 30% more cement factories (replaces coal); 30% to conventional waste incineration. * assumed to replace three alternative filling materials: expanded cork, EPDM and TPE14 (average of all three used in calculations).
Non-impregnated wood and combustible bulk waste	WAMPS (v.2019) modeling. Main input data and assumptions: All 100% are incinerated with energy recovery: 85/15% heating/electricity; energy replacement:—biofuels for heating and the Swedish electricity mix for the produced electricity.
Construction materials	WAMPS (v.2019) modeling. Main input data and assumptions: 100% used as construction materials and replaced sand, soil, gravel, stone or similar.
Non-combustible/inert waste	WAMPS (v.2019) modeling. Main input data and assumptions. 100% landfilled. Assumed to contain 4% organic material that is deposited in the landfill.

Appendix D

Table A4 builds on real household waste management data in Sweden from 2020. Three scenarios are presented:

- Baseline Scenario 1: Reflects the waste management practices in Sweden as of 2020. "Waste quantities 1" as listed in Table A4 was utilized for this scenario.
- Recycled Waste Scenario 2: In addition to the fractions already recycled in 2020, this scenario assumes that all residual waste and mixed bulky waste are separated and recycled. The compositional analyses of mixed residual household waste and mixed combustible bulky waste (referenced in Appendices A and B, respectively) were used to allocate mixed waste to recyclables. "Waste quantities 2" from Table A4 were applied in this scenario.
- Prevented Recyclables Scenario 3: Assumes that all residual waste and mixed bulky waste, alongside all recyclables already collected separately in 2020, are prevented from entering the waste stream. The same compositional analyses used in the Recycled Waste Scenario are applied here. "Waste quantities 2" from Table A4 were applied in this scenario.

The CO_2-eq tonnage results for each scenario were derived by multiplying the carbon footprints per kg of waste (as presented in Table 1) by the waste quantities (as detailed in Table A4). The "Differences" column indicates the variance between the "Prevented Recyclables" and "Recycled Waste" scenarios, highlighting the CO2_eq ton reduction achieved by prevention compared to recycling. The "Rank" column orders the waste fractions based on the "Differences" column, where a rank of 1 signifies the greatest reduction in CO_2-eq tons and 23 the least, underscoring the relative effectiveness of waste prevention versus waste management strategies.

Table A4. Carbon footprint estimates on household waste at the national level in Sweden.

Waste Fraction	Waste Quantities 1 (Tons) [33]	Waste Quantities 2	Baseline 2020 (Ton-CO_2-eq)	Prevented Recyclables (Ton-CO_2-eq)	Recycled Waste (Ton-CO_2-eq)	Difference (Ton-CO_2-eq)	Rank
WEEE	155,840	161,900	−233,800	−6,152,300	−242,900	−5,909,500	1
Textiles to recycling	3490	65,860	−4200	−1,646,500	−79,000	−1,567,500	3
Tires	12,000	12,000	−1200	−43,200	−1200	−42,000	12
Residual household waste	1,669,090	5010	333,800	−11,500	1000	−12,500	17
Metal packaging (20% Al; 80%Fe)	21,750	49,470	−39,200	−108,800	−89,000	−19,800	15
Food waste to anaerobic digestion	389,572	850,220	−39,000	−1,870,500	−85,000	−1,785,500	2
Food waste to central composting	7658	16,710	200	−36,800	500	−37,300	13
Food waste to home composting	29,140	63,600	2000	−139,900	4500	−144,400	8
Plastic packaging	99,600	333,490	−59,800	−700,300	−200,100	−500,200	5
Plastic (not packaging, bulky)	14,540	137,320	−8700	−288,400	−82,400	−206,000	6
Combustible bulky waste	613,910	494,520	184,200	−989,000	148,400	−1,137,400	4
Metal scrap (bulky) (steel: 50% virgin, 50% recycled)	176,550	176,550	−158,900	−335,400	−158,900	−176,600	7
Flat glass	2900	2900	−600	−3500	−600	−2900	19
Paper waste (non/packaging) (within EPR)	168,400	339,690	−151,600	−373,700	−305,700	−67,900	11
Glass packaging	248,520	291,830	−87,000	−233,500	−102,100	−131,300	9
Corrugated cardboard	56,340	56,340	−16,900	−33,800	−16,900	−16,900	16
Paper packaging	190,860	360,650	−38,200	−180,300	−72,100	−108,200	10
Plasterboard	27,330	27,330	−1400	−8200	−1400	−6800	18
Wood waste, not impregnated wood	538,351	661,130	−215,300	−132,200	−264,500	132,200	22
Construction materials	192,214	192,210	0	−1900	0	−1900	20
Non-combustible/inert waste	94,200	242,220	9400	−2400	24,200	−26,600	14
Garden waste	439,728	611,020	44,000	61,100	61,100	0	21
Total	5,151,980	5,151,980	−481,900	−13,231,200	−1,462,200		

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