

Environmental and economic assessment of household food waste source-separation efficiency in a German case study

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ABSTRACT

Biowaste management has a large potential for supporting closed-loop resource and nutrient management. The exploitation of this potential, however, is affected by upstream determinants, such as the consumers' behavior (separation efficiency), which influence the quantity and quality of biowaste intended for valorization. The goal of this study is to determine the effect of these upstream determinants on the environmental and the economic performance of biowaste management, upon implementation of an improved socio-technical collection infrastructure. Interventions were performed in two areas (A and B) in a Northern German city. The methodologies used for the environmental and the economic evaluation are the life cycle assessment and the net present value analysis, respectively. The functional unit of 1 kt of waste (biowaste and residual waste) generated at households was used. Three scenarios are assessed for each area and reflect the before (A_b , B_b), during (A_d , B_d), and after (A_a , B_a) phases of these interventions. The average global warming mitigation potential for A_b and B_b was -8 , for A_d and B_d was -56 , and for A_a and B_a was -22 kg CO₂ eq·kt⁻¹ biowaste. The economic assessment showed that the improved collection system is economically beneficial due to the larger amount of valuable products and lower overall treatment costs. The results suggest that improved sorting can influence the environmental and economic performance of the studied system and both study areas present environmental impact mitigation during the monitoring period. However, our results suggest sustained awareness-raising activities in order to maintain a high efficiency in household waste sorting.

1. Introduction

Household biowaste, especially when collected separately, is holding considerable potential in circular bioeconomy. In the European Union (EU), the largest share of food waste is generated at the consumption stage in households (over 50%) (Caldeira et al., 2017; Schmidt et al., 2019; Stenmarck et al., 2016). The management of household biowaste has a central focus in EU policies, such as in the Bioeconomy Strategy (European Commission, 2018a) and the Waste Framework Directive (European Commission, 2018b, 2008), which aim to promote prevention and optimal valorization of resources.

The performance of biowaste management strategies has been extensively studied highlighting the benefits of anaerobic digestion with energy and nutrient recovery (Tonini et al., 2020) over incineration

(Khoshnevisan et al., 2018; Mayer et al., 2020) and composting (Bernstad and la Cour Jansen, 2011; Tonini et al., 2020). An essential requirement for biowaste valorization in circular bioeconomy is its source-separation. According to the EU Fertilizer Regulation (European Commission, 2019), the organic fraction of mixed municipal waste separated by mechanical or other means is excluded from the material categories that an organic fertilizer can contain. Digestate produced from the anaerobic digestion of source-separated biowaste (SSBW) has indeed better quality than the digestate from mechanically separated biowaste (Angouria-Tsorochidou and Thomsen, 2021). Fertilizing products with low or zero macro-impurity content reduce the risk of pollutants being released to soil (Rodrigues et al., 2020). Nutrient recovery and soil quality improvement, therefore, can be achieved through digestate utilization, when it is produced from SSBW.

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Currently, a large share of food waste is still disposed with the residual waste in many regions in Europe although a separate collection system is already in place, either for food waste or for biowaste including green waste (Dornbusch et al., 2020; Edjoub et al., 2018). The efficiency of source-separation depends on the citizen engagement and the design of the collection system. In Germany, residual waste contains on average 39% biowaste, out of which 88% is food waste (Dornbusch et al., 2020). Bernstad and La Cour Jansen. (2012) argues that the installation of equipment for separate collection of food waste and convenience for sorting in kitchens can increase source-separation. Walk et al. (2019; 2021) argue that when the distribution of information material is supported by changes in the collection infrastructure, significant improvements in the separation efficiency can be achieved. Different actions can increase the circular flows and resource recovery at the end-of-life of biowaste. Environmental studies often lack including citizen's behavioral patterns as an integrated parameter for the efficacy of a waste management system. To the author's knowledge, system value, quantified by improved environmental performance and economic viability, which may be obtained by implementation of an improved socio-technical collection system supporting source-separation at the household level has not been studied yet.

The goal of this study is to evaluate the potential environmental restoration and economic viability of a system designed to include improved biowaste source-separation by motivating and educating citizens through campaigns. The original study was performed in Lübeck, Germany, and its detailed design is described by Walk and Körner (2021). The campaigns involve the introduction of a new biowaste sorting and collection infrastructure and design with a focus on food waste supported by sorting awareness campaigns. They were performed in two areas with different socio-economic background. Therefore, it evaluates the effect of upstream determinants (food waste generators' sorting behavior and collection infrastructure) including socio-economic different backgrounds of the two areas on the environmental and economic performance of biowaste management.

Using the waste characterization data generated during the campaigns and based on the existing waste management infrastructure of Lübeck, a life cycle assessment (LCA) and a net present value (NPV) analysis are conducted.

2. Materials and methods

Two residential areas were selected consisting of multi-family households located in different districts in the city of Lübeck, because issues with SSBW were often identified within multi-family houses (Bauverein, 2018; TRAVE, 2018). The two areas selected, A (in the district Moisling) and B (in the district St. Gertrud), were chosen based on the assessment of the housing associations (Bauverein, 2018; TRAVE, 2018) with the intention to reflect socio-economic differences, in order to account for this variable. In total, 37 households in area A and 46 households in area B were included. The number of inhabitants was estimated at 59 for area A and 78 for area B, based on the statistics of the residential areas (Hansestadt Lübeck, 2019). Socio-economic differences were reflected on the rent, the level of education that was identified through random sample surveys, and the unemployment rate. In area A, the apartments are offered exclusively to individuals under fulfilment of certain criteria, e.g. low income. The rent was publicly subsidized and capped at $5.65\text{€}(\text{m}^2)^{-1}$ in 2018 (TRAVE, 2018), being below the German average. The level of education scored as 3.3 based on the German Qualification Framework (Bundesministerium für Bildung und Forschung, 2013). The unemployment rate of the Moisling district was 11.4% (Hansestadt Lübeck, 2019). In area B, the rent was not subsidized and amounted to $9.00\text{€}(\text{m}^2)^{-1}$ (Bauverein, 2018) being above the German average. The education level scored 4.4. The unemployment rate of St. Gertrud district was 6.8%, same as Lübeck's average (Hansestadt Lübeck, 2019).

Both areas had implemented source-separation of biowaste, paper, and dry recyclables before the start of the study, as well as collection of

residual waste. The city of Lübeck implemented door-to-door SSBW collection followed by composting in 1996 (Hansestadt Lübeck, 2000) while treatment of SSBW with anaerobic digestion and subsequent composting of the solid digestate together with green and woody waste was implemented in 2005 (EBL, 2017). The multi-family households share the waste bins. The housing association of area A (TRAVE, 2018) reported that there were challenges with correct sorting, high impurity rates, and waste being placed outside of the bins. The housing association of area B (Bauverein, 2018) reported that there were no such challenges, that the inhabitants sorted correctly, and were well organized in communicating and solving issues amongst themselves.

To support the argument on the socio-economic differences, the two areas were statistically tested for their significant differences. The parameters were distinguished between study-phase-dependent and independent parameters. It was assumed that the phase-dependent parameters would change across the different study-phases due to the influence of the interventions, while the study-phase-independent parameters should remain unaffected. The alternative hypothesis for all tests was that area A has a lower performance on reducing waste generation as well as sorting efficiency than area B due to their socio-economic differences. Therefore, t-test was performed (R Core Team, 2021) including the test for normal distribution and difference in variances. The t-test was performed one-tailed, and if no significance was found a two-tailed t-test followed.

The interventions included actions related to the waste collection infrastructure and raising awareness of the residents regarding correct waste sorting. A major change was the collection frequency that intensified from one to three times a week. Furthermore, 5-L collection buckets were given to the residents for free. An intermediate storage facility was also introduced to increase the convenience for biowaste sorting. The storage facility was used to deliver the biowaste buckets by the citizens and as a pick up point for clean/empty buckets. The residents were informed about the collection tests with leaflets, prior to the trial, and an event was organized to introduce the new collection system (Walk et al., 2019; Walk and Körner, 2021). The pilot campaign lasted for five weeks in 2019 and then the original collection system was restored but the residents could keep their buckets. The long-term effects were evaluated after six and 12 months after the five-week pilot.

The study started with the assessment of the initial waste sorting behavior to verify the statements of the housing associations. Characterization of SSBW and residual waste were performed as a mean to determine the share of food waste, green waste, and other waste fractions. A more detailed food waste characterization was performed to determine the avoidable, partly avoidable, and unavoidable, (see Östergren et al., 2014 and Teigiserova et al., 2020 for food waste definitions) as well as the plant- and animal-based fractions of the food waste found in the SSBW and the residual waste (see Appendix A1). Each fraction of food waste was also analyzed for its chemical characteristics (see Appendix A1). The methodology performed for each parameter is described in Walk et al. (2021). According to these chemical characteristics, the quality of the biodegradable fraction in the SSBW and the residual waste was calculated for each scenario (see Appendix A2).

2.1. Life cycle assessment

The environmental performance of the waste management system is evaluated with the LCA methodology (JRC, 2010) performed according to the international standards ISO 14040–44 (International Organization for Standardization, 2006). The modelling follows an attributional approach with system expansion, excluding the footprint associated with waste generation (zero burden) (Björklund et al., 2010). Three scenarios are assessed that reflect the before (A_b , B_b), during (A_d , B_d), and after (A_a , B_a) phases of the interventions. Details for each scenario is provided in Appendix A3 and differences between the modelled scenarios and the real conditions in Lübeck (limitations) are presented in Appendix A4.

2.1.1. Goal and scope

The rationale of the study is to support science-based evidence for the importance of citizen's motivation to improve their environmental performance through better separation of biowaste at the household level and therefore a key element to consider in the design of policy instruments boosting a regenerative circular bioeconomy. The potential environmental and economic benefits can act as a motivating power for public or private investment in citizen education and engagement activities. The study is identified in situation A "micro-level decision support" (European Commission - Joint Research Centre, 2010) because the changes by the interventions are not expected to cause major changes in the background system or economy.

In the studied areas, food waste is disposed in both the SSBW and the residual waste. Both waste streams are treated with similar technologies until the digestate phase-separation (see Fig. 1). The waste from both streams is pre-treated (shredding, sieving, and mixing) and the biodegradable fraction (fine fraction) is anaerobically digested for biogas production. The digestate from the SSBW is phase-separated, with the solid fraction being composted and used as soil improving and fertilizing material and the liquid fraction is being recirculated as process water to the pre-treatment process (substituted water) while a constant stream is sent to the residual waste line before it is sent to the wastewater treatment plant. The digestate from the anaerobic digestion of the fine fraction of the residual waste is phase-separated with the solid fraction being incinerated and the liquid fraction being sent to a wastewater treatment plant (WWTP).

The functional unit (FU) of the system is the management of 1 kt wet weight (ww) waste generated at the household level, excluding source-separated plastic, paper, glass, and metal which are sent for recycling. Area A and B generate different total amounts of waste per capita and, thus, the FU was normalized to 1 kt ww. The different composition of SSBW and residual waste between the two areas is also taken into account and the results are compared on a mass basis according to the area's waste composition.

2.1.2. Life cycle inventory

The foreground system consists of primary data that are produced during the waste sorting experiments, e.g., waste composition, or are provided by the treatment facility, e.g., methane content of biogas, and is supplemented by literature data when primary data were not available, e.g., fugitive emissions from the anaerobic digestion. The background system, such as the energy mix, consists of datasets in the Ecoinvent database (Moreno Ruiz et al., 2020; Wernet et al., 2016).

The SSBW and the residual waste are received at the treatment facility, where they are stored with a mass loss of 0.5% (EBL., 2021a), which is assumed to be water. They are pre-treated for bag and

packaging opening, size reduction and removal of other unwanted materials. The reject fractions of the pre-treatments that contain plastics, paper, glass, and other unidentified materials are incinerated. Materials, such as metals are recovered from the reject fraction. The energy potential and the emission inventories for the waste types incinerated were adopted from the Ecoinvent database (Wernet et al., 2016). The unidentified materials were modeled as textiles because they represented the largest fraction along with the sanitary textiles. Water is added to the fine fraction to achieve a total solid content of 12% (EBL., 2021b) and the mixture is sent for anaerobic digestion (EBL., 2021a). The methane (CH_4) production efficiency is 88% of the biochemical methane potential (BMP) (EBL., 2021a). The latter was calculated from the BMP of the animal ($572 \text{ mLCH}_4\text{g}^{-1}\text{VS}$), vegetable ($425 \text{ mLCH}_4\text{g}^{-1}\text{VS}$), and green ($237 \text{ mLCH}_4\text{g}^{-1}\text{VS}$) waste (Naroznova et al., 2016) found in the fine fractions. The biogas produced has a CH_4 content of 58% (EBL., 2021a). Fugitive emissions from the anaerobic digestion are considered to be 1% of the biogas produced (Møller et al., 2009). The biogas is combusted in a combined heat and power engine and the relevant emissions are presented in Appendix A5.

Digestate from the SSBW line (see Fig. 1) is phase-separated with a centrifuge. The solid digestate is composted and used as fertilizer on agricultural soil offering nutrients and replacing mineral fertilizer in the market, as well as improving the soil quality through carbon sequestration (Gilbert et al., 2020). The liquid digestate is used as process water in the system (in A_b and B_b), substituting tap water. In A_d , B_d , A_a , and B_a , the liquid digestate is used as organic fertilizer, replacing mineral fertilizer. The solid and liquid digestate is assumed to be transported 10 and 25 km, respectively, with a 16–32 kt lorry (Wernet et al., 2016). Nitrogen (N) fertilization is limited by the legislation to $170 \text{ kg N}\cdot\text{ha}^{-1}$ (European Commission, 1991) while some countries also limit the allowed amount of phosphorus (P) applied on soil to $30 \text{ kg P}\cdot\text{ha}^{-1}$ (Danish Environmental Protection Agency, 2017), which result to a N/P ratio of 5.67. To satisfy both regulatory limits, the spreading area was calculated on the base of P when the N/P ratio of the digestate was less and on the base of N when it was more than 5.67, similar to the methodology used in Angouria-Tsorochidou et al. (2022).

The indirect impacts associated with the spreading of digestate include the machinery (Wernet et al., 2016) and fuel consumption. The diesel consumption for spreading of solid and liquid digestate is $11.25 \text{ L}\cdot\text{ha}^{-1}$ and $13 \text{ L}\cdot\text{ha}^{-1}$ (Lacour et al., 2012). The application of composted digestate on soil has negligible environmental impacts in terms of greenhouse gas emissions (Dietrich et al., 2020). Cumulative emissions from composting and composted digestate application are considered comparable to the emissions of not-composted digestate application on agricultural soil (Dietrich et al., 2021). Thus, emission factors for the application of not-composted digestate were adopted

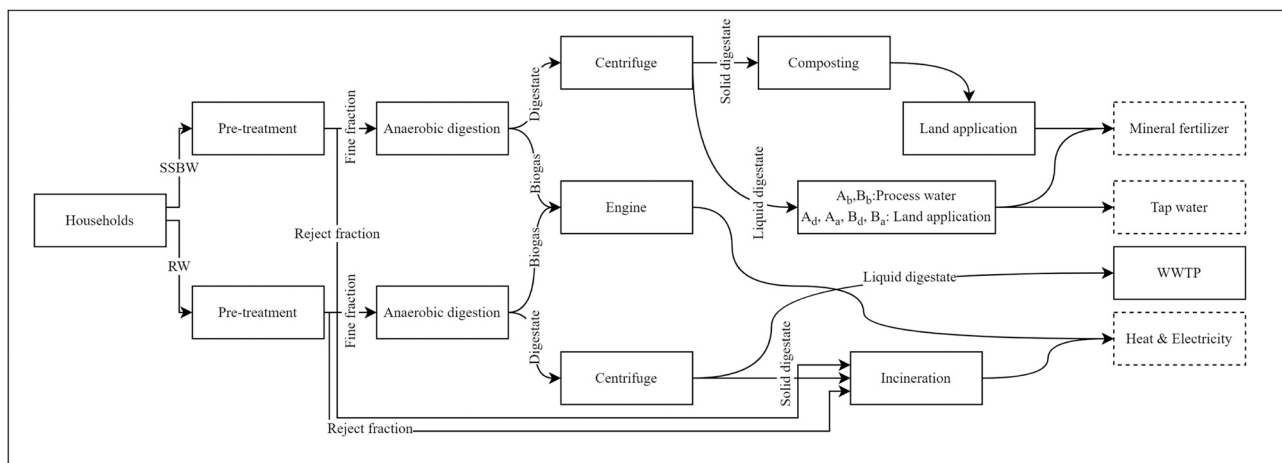


Fig. 1. Flow diagram of the modelled treatment processes (avoided products are indicated with dotted lines).

Table 1
Economic inventory.

Parameter	Unit	Value	Lifetime (years)	Reference
CAPEX				
BW line	€	13,648,781	10	(EBL., 2021b)
BW line	€	9160,498	20	
RW line	€	14,788,425	10	
RW line	€	8414,713	20	
OPEX				
BW line	€kt ⁻¹ total input	1.18	-	
RW line	€kt ⁻¹ total input	1.17	-	
Revenues				
Electricity – BW line	€kWh ⁻¹	0.16	-	
Electricity – RW line	€kWh ⁻¹	0.116	-	
Metals	€kt ⁻¹	50	-	
Phosphate	€kt ⁻¹	77.4	-	(Statista., 2021)
Ammonium nitrate	€kt ⁻¹	1.5	-	(NMSU, 2017)
Water	€(m ³) ⁻¹	2.57	-	(Water news Europe., 2020)

(see Table 1). The solid and liquid digestate substitute phosphate fertilizer (P₂O₅), according to their phosphorus content and ammonium nitrate fertilizer (NH₄NO₃), according to their nitrogen content, with 40% of the nitrogen to be considered for substitution (Møller et al., 2009). Avoided impacts include indirect impacts from the production of mineral fertilizer, that were adopted from the Ecoinvent database (Wernet et al., 2016) and direct impacts from the avoided use of mineral fertilizer that are presented in Appendix 5. Appendix 6 presents the inventory data of the amounts of reject fraction incinerated, heat and electricity produced, as well as information on the fertilizer substitution.

Digestate from the residual waste line (see Fig. 1) is phase-separated with a centrifuge. The solid digestate is incinerated and the liquid digestate is sent to a wastewater treatment plant (WWTP), which are modelled from the Ecoinvent database (Wernet et al., 2016).

2.1.3. Impact assessment

The impact assessment was performed using the life cycle impact assessment methodology ReCiPe 2016 at midpoint level (H) (Huijbregts et al., 2017) and the modelling was performed in the SimaPro software 9.1 (PRé Sustainability, 2021).

2.1.4. Sensitivity analysis

Two sensitivity analyses (S₁, S₂) were performed to evaluate the results of the life cycle impact assessment methodology. S₁ is based on the accuracy of the biogas production values used and S₂ on the composition of the waste type “Others” fraction of the reject.

S₁ assessed the biogas production reported from the waste management company of Lübeck for the fine fraction of the SSBW and the residual waste line, compared to the biogas production calculated in the study based on the theoretical BMP (Naroznova et al., 2016) including the plant’s efficiency of 88% (EBL., 2021a). The waste management company and the calculated amounts present differences. The reason for not using the reported values for life cycle impact assessment methodology was that they represent average plant values and do not account for the different biowaste composition in the scenarios.

S₂ assessed the fraction of waste type “Others”, which is modelled as textiles, and contains a share of non-textile materials that was not possible to further determine their origin. It included e.g. distinct amounts of stones and soil. A sensitivity analysis (S₂) was performed by halving the amount of the combustible fraction, i.e., textiles in waste type “Others” (see Appendix A6).

2.2. Net present value

The economic feasibility of the system was evaluated with the NPV analysis, for which the present value of future economic flows is balanced with the capital investment that occurs at the start of the project.

Costs are calculated as negative and revenues as positive values. The NPV is calculated with Eq. 1 and an economically feasible investment is indicated with a NPV > 0. The lifetime of the project is set at 20 years with a discount rate of 6% (Ascher et al., 2020; Chang and Pires, 2015). The future yearly costs and revenues were discounted to their current value and their sum was calculated using Eq. 1.

$$NPV = \sum_{t=1}^n \frac{R_t}{(1+i)^t} \quad (1)$$

where, R_t is the net cash inflow-outflow during year t, i is the discount rate, and t the number of project years.

The capital (CAPEX) and operational (OPEX) expenditures of the SSBW and residual waste lines, that were acquired from the waste management company of Lübeck (EBL., 2021b) were considered as costs. The CAPEX includes all the equipment costs for the residual and the biowaste lines, including groundwork of the facility and the units of pre-treatment, treatment, anaerobic digester, exhaust gas, water cycling, and composting. The OPEX includes, but is not limited to, all the operational costs including salaries and wages, utility expenses, and supplies. The sales of metals sorted during the pre-treatment (EBL., 2021b), the organic fertilizer, the substituted water, and electricity were considered as revenues. The heat produced is used internally in the plant. A detailed inventory is presented in Table 1.

3. Results and discussion

3.1. Selected areas

The amount of waste types in the residual waste and SSBW are presented in Table 2 (see Appendix A7 for the distribution of waste types per week). In A_b and B_b the participants disposed more food waste in the residual waste (A_b = 448 kg food waste-FU⁻¹, B_b = 328 kg food waste-FU⁻¹) than in the SSBW (A_b = 30 kg food waste-FU⁻¹, B_b = 65 kg food waste-FU⁻¹). During the interventions they improved their sorting behavior, with A_d = 226 kg food waste-FU⁻¹ and B_d = 160 kg food waste-FU⁻¹ in the residual waste and A_d = 363 kg food waste-FU⁻¹ and B_d = 312 kg food waste-FU⁻¹ in the SSBW. In the following period, both areas decreased their sorting efficiency (A_a = 269 kg food waste-FU⁻¹, B_a = 219 kg food waste-FU⁻¹ in the residual waste and A_a = 139 kg food waste-FU⁻¹, B_a = 194 kg food waste-FU⁻¹ in the SSBW) but still had an improved performance compared to A_b and B_b, which calls for a repetition of the interventions at least once per year.

The amount of miss-sorted waste (impurities) in the SSBW bin was higher before the interventions (A_b = 6% and B_b = 14%). During the interventions, the share of impurities decreased to 0.5% for A_b and to 1.2% for B_b and after the interventions, they were measured to 3% for A_b and 5% for B_b. The largest share of impurities was mostly waste

Table 2

Results from the characterization campaigns in the two areas [$\text{kg}\cdot\text{inhabitant}^{-1}\cdot\text{week}^{-1}$]. The values are means with standard deviation. The value of 0 indicates that the waste type was not present (thus, not measured) while the 0.00 ± 0.00 indicates that the waste type was measured but was in very small amounts.

	Waste type	Before (b)		During (d)		After (a)	
		Biowaste	Residual waste	Biowaste	Residual waste	Biowaste	Residual waste
Area A	$n^a_{\text{measurements}}$	2	2	12 ^b	12 ^b	4	4
	$n_{\text{characterizations}}$	1	1	12 ^b	1	4	4
	Scenario	A _b		A _d		A _a	
	Food waste	0.11 ± 0.03	1.59 ± 0.28	1.13 ± 0.14	0.70 ± 0.06	0.59 ± 0.14	1.14 ± 0.25
	Green waste	0.07 ± 0.02	0.14 ± 0.03	0.04 ± 0.02	0.20 ± 0.02	0.19 ± 0.10	0.27 ± 0.29
	Paper	0.00 ± 0.00	0.44 ± 0.08	0.00 ± 0.00	0.19 ± 0.02	0.02 ± 0.01	0.42 ± 0.09
	Plastic	0.00 ± 0.00	0.25 ± 0.04	0.00 ± 0.00	0.19 ± 0.02	0.00 ± 0.00	0.30 ± 0.10
	Glass	0.00 ± 0.00	0.13 ± 0.02	0	0.08 ± 0.01	0.00 ± 0.00	0.21 ± 0.07
	Metal	0.00 ± 0.00	0.06 ± 0.01	0.00 ± 0.00	0.04 ± 0.00	0.00 ± 0.00	0.06 ± 0.03
	Others	0.01 ± 0.00	0.74 ± 0.13	0.00 ± 0.00	0.53 ± 0.05	0.00 ± 0.00	1.02 ± 0.11
Total	0.19 ± 0.05	3.35 ± 0.60	1.18 ± 0.16	1.93 ± 0.18	0.81 ± 0.16	3.42 ± 0.81	
Area B	Scenario	B _b		B _d		B _a	
	Food waste	0.19 ± 0.00	0.94 ± 0.01	0.92 ± 0.05	0.47 ± 0.02	0.59 ± 0.06	0.66 ± 0.09
	Green waste	0.28 ± 0.00	0.24 ± 0.00	0.04 ± 0.02	0.37 ± 0.02	0.40 ± 0.14	0.21 ± 0.12
	Paper	0.07 ± 0.00	0.20 ± 0.00	0.01 ± 0.00	0.29 ± 0.01	0.05 ± 0.01	0.34 ± 0.04
	Plastic	0.00 ± 0.00	0.20 ± 0.00	0.00 ± 0.00	0.16 ± 0.01	0.00 ± 0.00	0.17 ± 0.00
	Glass	0	0.18 ± 0.00	0	0.15 ± 0.01	0	0.15 ± 0.02
	Metal	0	0.05 ± 0.00	0	0.03 ± 0.00	0.00 ± 0.00	0.02 ± 0.01
	Others	0.00 ± 0.00	0.51 ± 0.01	0.00 ± 0.00	0.51 ± 0.01	0.00 ± 0.00	0.44 ± 0.06
	Total	0.54 ± 0.01	2.31 ± 0.03	0.97 ± 0.06	1.98 ± 0.09	1.04 ± 0.19	1.99 ± 0.15

^a Number

^b Measurements were done three times per week but summed up to weekly values.

paper and paper bags used for wrapping, especially in area B. The non-biogenic material in the residual waste ranged from 48% to 59% in all scenarios, implying a large share of biowaste discarded in the residual waste.

The areas showed different waste generation and waste sorting behavior in all scenarios. Area A had significantly higher total waste, biowaste, and food waste generation compared to area B. The significant difference in the generated food waste is due to the higher amount of avoidable food waste. Unavoidable and partly avoidable food waste were not significantly different. The share of food waste in total waste was significantly higher in area A but no significant difference was found in the share of biowaste in the total waste. The background reasons for the higher waste generation in area A was not further investigated and strong correlations between the sorting efficiency and the socio-economic status should be handled with caution.

The amount of waste found in the SSBW and residual waste depends on the study phase. More food waste was sorted correctly in A_d and B_d (62%, 66%) and A_a and B_a (34%, 47%) compared to A_b and B_b (6%, 17%). The change in the total amounts of waste is not only influenced by the interventions but also by external factors, such as seasonal differences, which is also supported by Edjabou et al. (2018). The total amount of residual waste and biowaste was significantly higher in area A. No significant difference between the areas was found in the total amounts found in the biowaste bins. Table 3 presents the parameters that showed significant differences between the two areas while the statistical analysis of all parameters is presented in Appendix A8.

3.2. Life cycle assessment

The aggregated results of the LCA are presented in Fig. 2 and Table 4 and the detailed results in Appendix A9. The impacts presented derive from several processes summarized into eight groups. The *pre-treatment* includes the infrastructure and water consumption of the process. The *anaerobic digestion* takes into account the infrastructure and emissions associated with the process and the *biogas combustion* the infrastructure, emissions, and heat and electricity production as avoided products. The *solid digestate* from the SSBW includes emissions from storage and application on land and avoided mineral fertilizer production and use. The *solid digestate* from the residual waste includes the *reject incineration*

with energy generation as avoided impact. The *liquid digestate* from the biowaste in A_b and B_b includes the avoided impacts from tap water substitution and in A_d, B_d, A_a, and B_a it includes emissions from the digestate application activities. For the residual waste includes its treatment in a WWTP. The *digestate transport and spreading* is only calculated for the biowaste line. The *electricity consumption* includes the total electricity demand of the plant.

The two areas are modelled individually with normalized waste composition to 1 kt ww generated. The black dot indicates the *net* performance per FU (normalized results). The red dot indicates the net environmental impact if the difference in the amount of biowaste and residual waste generated per inhabitant per week between the two areas was taken into account (*increased waste generation*).

The lowest environmental impact was observed during the intervention (scenarios A_d, B_d), except for the categories associated with the emissions from the digestate utilization, such as terrestrial ecotoxicity and marine eutrophication. The environmental performance before the interventions (scenarios A_b, B_b) presents the highest environmental impacts. The only impact categories that consistently present net negative environmental impact in all scenarios are the fossil resource depletion and freshwater eutrophication.

3.2.1. Global warming

The global warming potential for A_b, A_d, and A_a is 78, 56, and 135 kg CO₂ eq·FU⁻¹ and for B_b, B_d, and B_a it is 108, 55, and 56 kg CO₂ eq·FU⁻¹, respectively. The sorting performance is better in A_d and B_d in terms of source-separation and reduced impurities and the fact that the trend in the global warming potential does not reflect this improvement in area A is attributed to the residual line. This is due to the fact that the residual waste in A_a includes more non-biogenic materials compared to A_b and, thus, the impact from the incineration of these materials increases the environmental burden. The SSBW performs better in all scenarios compared to the residual waste. In fact, the SSBW presents consistently global warming mitigation potential while the residual waste line presents environmental degradation.

Area A performs better than area B due to the higher amount of food waste disposed in the SSBW. The improved environmental performance is determined from the SSBW (A_d = -63 kg CO₂ eq·FU⁻¹, B_d = -50 kg CO₂ eq·FU⁻¹) rather than the residual waste (A_d = 118 kg CO₂

Table 3
Parameters showing significant differences, with the Welch Two Sample t-test.

$\alpha = 0.05$				
Unit	Parameter	Mean area A	Mean area B	p-value
kg-inhabitant ⁻¹ .week ⁻¹	Biowaste + Residual waste	3.64 ± 0.78	2.97 ± 0.09	1.45·10 ⁻⁰²
	Total biowaste ^a	2.09 ± 0.39	1.79 ± 0.10	2.79·10 ⁻⁰²
	Total food waste ^a	1.76 ± 0.26	1.28 ± 0.12	1.26·10 ⁻⁰⁴
	Total avoidable food waste ^a	0.96 ± 0.14	0.50 ± 0.06	6.24·10 ⁻⁰⁷
	Total residual waste ^b	2.81 ± 0.93	2.05 ± 0.17	1.87·10 ⁻⁰²
Total food waste ^a (% of total waste)		49.49 ± 8.34	42.70 ± 4.05	2.35·10 ⁻⁰²

^a Study-phase-independent, ^bStudy-phase-dependent

eq·FU⁻¹, B_d = 105 kg CO₂ eq·FU⁻¹). On average, treating 1 kg of SSBW has a global warming potential of -0.2 kg CO₂ eq·FU⁻¹. In area A, the increased biowaste generation results in improved environmental performance in A_d (-76 kg CO₂ eq) compared to the results obtained per FU (-63 kg CO₂ eq), implying that waste generation offers environmental benefits, which requires attention when interpreting LCA results. Higher waste generation means higher biogas production and avoided electricity. However, this is a result of the waste being considered as *zero burden* when entering the waste management system.

Ascher et al. (2020) found a global warming mitigation potential of -92 kg CO₂ eq·kt⁻¹ food waste treated and Evangelisti et al. (2014) calculated it to -65 kg CO₂ eq·kt⁻¹ SSBW. In this study, the average global warming mitigation potential for A_b and B_b was -70, for A_d and B_d was -160, and for A_a and B_a was -91 kg CO₂ eq·kt⁻¹ SSBW treated. The studies have different system boundaries, such as inclusion of biowaste collection and transport contributing to global warming potential (Ascher et al., 2020) and inclusion of the reject incineration only in the current study. Khoshnevisan et al. (2018), who modelled a system with similar boundaries including reject incineration and nutrient recovery, found a global warming mitigation potential of -758 to -941 kg CO₂ eq·kt⁻¹ SSBW, depending on the pre-treatment method. The differences in the net global warming potential can be attributed to the inclusion of infrastructural components in this study, the use of marginal electricity and heat data in the study performed by Khoshnevisan et al. (2018), but also the assumptions and modelling approach used.

3.2.2. Marine Eutrophication

For scenarios A_b, A_d, and A_a the total marine eutrophication potential is 0.09, 0.4, and 0.21 kg N eq·FU⁻¹ and for B_b, B_d, and B_a it is 0.11, 0.31, and 0.28 kg N eq·FU⁻¹, respectively (Fig. 2). The impacts are mostly influenced by the SSBW, reflecting the trends of the sorting behavior and specifically the higher amounts of SSBW collected. For the SSBW, the most contributing processes are the *solid* and *liquid digestate* utilization while insignificant mitigation is achieved from the biogas combustion.

The digestate evaluation is based on the potential impacts of N and P contained in the digestate. The production and use of 1 kg N as NH₄NO₃ has a marine eutrophication potential of 0.16 kg N eq (Wernet et al., 2016). Exemplified by A_d, the application of the solid digestate on agricultural land has a marine eutrophication potential of 0.3 kg N eq while the avoided impacts from the P₂O₅ and NH₄NO₃ production and use is -0.002 and -0.1 kg N eq, respectively, resulting in a marine eutrophication potential of 0.2 kg N eq. The application of NH₄NO₃ has lower marine eutrophication related emissions compared to digestate application (Yoshida et al., 2016), which are not balanced with the avoided impacts from NH₄NO₃ production.

3.2.3. Freshwater eutrophication

For scenario A_b, the freshwater eutrophication mitigation potential is -0.08, for A_d, and A_a it is -0.07, for B_b and B_a it is -0.06, and for B_d it is -0.07 kg P eq·FU⁻¹ (Fig. 2). In both, the SSBW and the residual

waste the mitigation potential derives from the biogas utilization, and for the RW additional mitigation comes from the reject incineration. The highest share of avoided emissions is from the avoided spoils of phosphate from lignite and hard coal mining. However, this is mostly a modelling approach rather than the real conditions. The actual highest contribution to freshwater eutrophication is from the solid digestate application. *Solid digestate* models the impacts from land application (A_d = 0.05 kg P eq) and the avoided impacts from the production and use of P₂O₅ (-0.0519 kg P eq) resulting in a net zero environmental impact. Thus, the process highlights impacts of secondary importance.

The partitioning of P during the raw digestate phase-separation is 96% to the solid and 4% (Angouria-Tsorochidou et al., 2022) to the liquid fraction, explaining the minor eutrophication potential from the *liquid digestate*. (-0.002 kg P eq). The residual waste follows a similar trend in this impact category, with the highest share of impacts being associated to the treatment of the liquid at the WWTP (0.002 kg P eq).

3.2.4. Mineral resource scarcity

For scenarios A_b, A_d, and A_a the mineral resource scarcity is 0.02, -0.1, and 0.05 kg Cu eq and for B_b, B_d, and B_a it is 0.05, -0.03 and 0.002 kg Cu eq (Fig. 2). The environmental burdens in this impact category are due to the iron ore and other metals used for the infrastructure, with the highest share being from the infrastructure of the anaerobic digestion plant. Mitigation derives from the avoided zinc concentrate (-0.06 kg Cu eq) used for P₂O₅ production and the avoided P₂O₅ rock extraction (-0.051 kg Cu eq). The avoided impacts from the biogas valorization are also associated to the iron ore and the avoided infrastructure necessary for the energy mix used.

The increased avoided impacts in scenarios A_d, and B_d after the interventions are due to the higher amount of SSBW and, thus, fertilizer, produced. In this category the SSBW presents consistently net negative impacts while the residual waste net positive impacts.

3.2.5. Other impact categories

Some other LCA impact categories are presented in Table 4. In some of these impact categories area A performs better than area B, such as in terrestrial acidification (TA) and ecotoxicity (TE). The incineration of the reject fraction from the residual waste line creates the largest environmental burden while the biogas utilization offers the largest impact mitigation (see Appendix A9) in all impact categories. An exception is presented in the fossil resource scarcity that has a net negative environmental impact in all scenarios. It is also worth noting that for the fossil resource scarcity the residual waste line performs better than the biowaste line (exemplified by A_d, biowaste line: -33 kg oil eq, residual waste line: -52 kg oil eq) due to the energy generated from the incineration of the reject fraction.

The results suggest that improved sorting can influence the environmental performance of the system and both areas present impact mitigation during and after the interventions. While the biogas potential is completely utilized, solid and liquid digestate can only be marketed as organic fertilizer if the BW is collected as a source-separated fraction. The digestate utilization has an environmental impact of

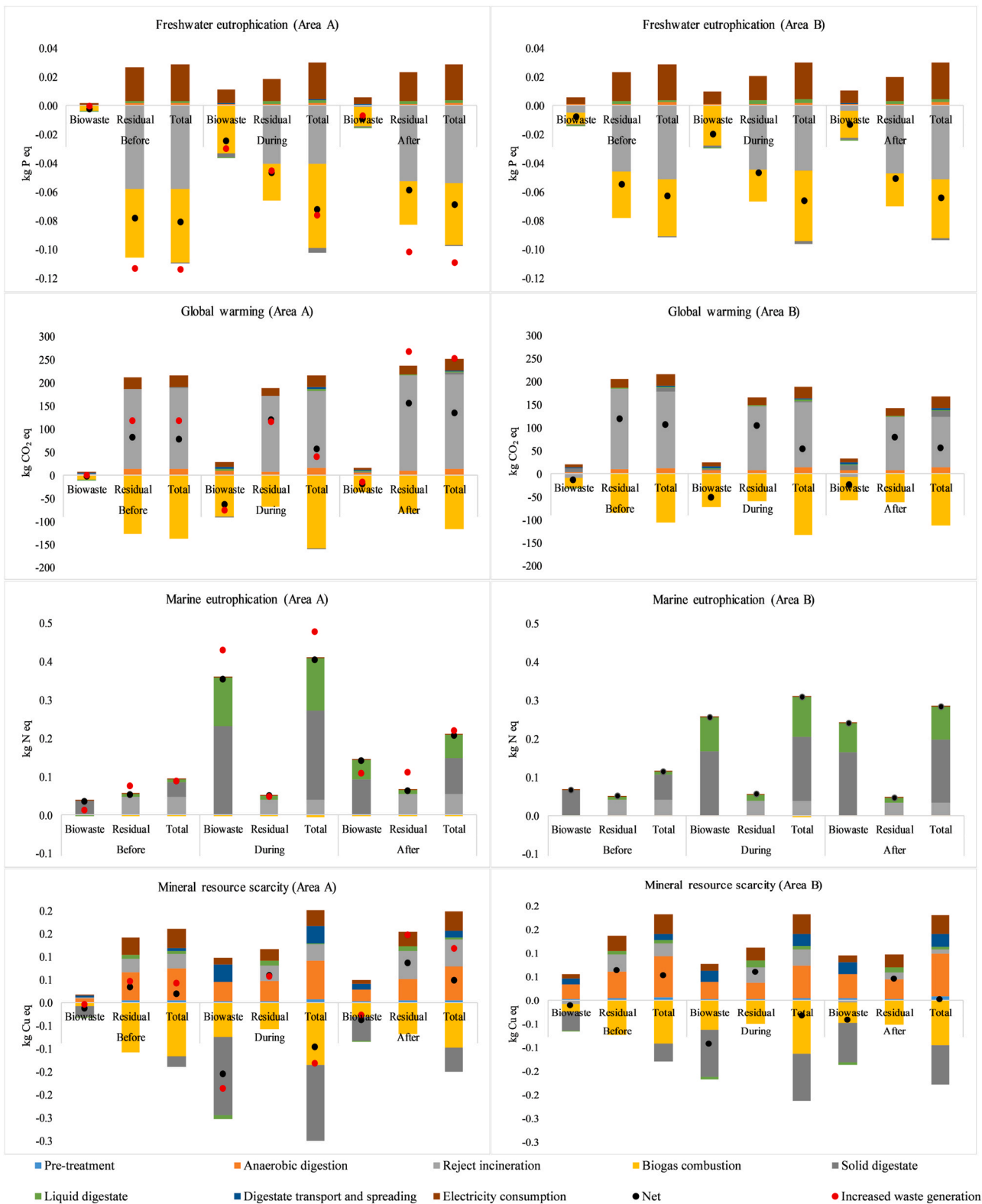


Fig. 2. LCA results.

0.01 kg CO₂ eq, 0.001 kg N eq, $-6 \cdot 10^{-6}$ kg P eq, and $-0.3 \cdot 10^{-4}$ kg Cu eq per kg, exemplified by B_d and the mitigation potential is lost when biowaste is disposed in the residual waste.

Another interesting aspect is that the solid digestate incineration does not have a contribution to global warming, since only biogenic CO₂ emissions are generated during the process. The solid digestate utilization has the potential, through the storage (CH₄) and application

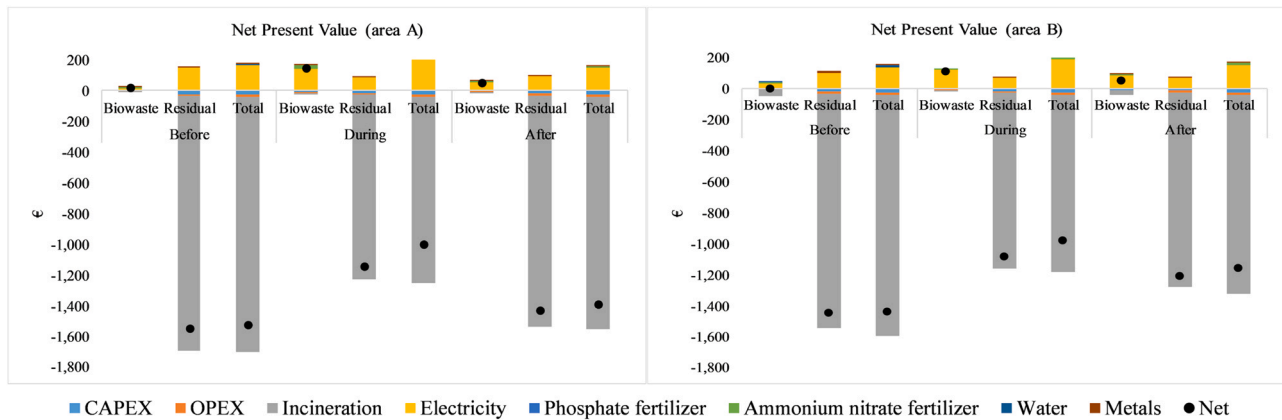
activities (N₂O) to contribute to global warming. However, this shift is reflected in mineral resource scarcity, where the nutrients are lost when the digestate is incinerated. The solid digestate utilization presents a net negative environmental impact only in scenario A_d where the largest amount of food waste is noticed in the biowaste bin allowing for higher carbon sequestration and fertilizer substitution.

LCA studies on biowaste, residual waste, and municipal waste

Table 4

Net results of the life cycle assessment per FU in other impact categories (IC: impact category, TA: terrestrial acidification, TE: terrestrial ecotoxicity, FE: freshwater ecotoxicity, ME: marine ecotoxicity, HCT: human carcinogenic toxicity, HnCT: human non-carcinogenic toxicity, FRS: fossil resource scarcity).

IC	TA	TE	FE	ME	HCT	HnCT	FRS
Unit	kg SO ₂ eq	kg 1,4-DCB	kg 1,4-DCB	kg 1,4-DCB	kg 1,4-DCB	kg 1,4-DCB	kg oil eq
A _b	0.01	33.51	10.60	14.08	-0.26	274.32	-89.30
B _b	0.04	90.43	10.44	13.87	1.30	233.77	-71.50
A _d	0.07	33.54	7.48	9.97	-0.50	230.92	-85.30
B _d	0.07	40.74	8.05	10.72	0.22	239.64	-78.42
A _a	0.12	71.75	11.41	15.17	0.80	354.44	-78.22
B _a	0.03	77.89	8.61	11.44	1.07	460.97	-72.76

**Fig. 3.** Results of the net present value analysis.

management systems do not account for the burdens associated with the stages before the waste disposal (Bernstad and la Cour Jansen, 2011; Evangelisti et al., 2014; Khoshnevisan et al., 2018; Thomsen et al., 2017), such as in food production. They are instead considered as *zero burden* (Björklund et al., 2010). The household biowaste contains a large fraction of food waste that is avoidable (20–60% in this study) and partly avoidable (11–22% in this study) (see Appendix A1). When these types of waste are considered as *no burden*, there is a risk that food waste treatment has better environmental performance than food production and consumption. As a simplified example, the production of 1 kg of lettuce has a global warming potential of 0.15 kg CO₂ eq (Wernet et al., 2016) while this study suggests that the treatment of 1 kg SSBW has a global warming mitigation potential of -0.1 kg CO₂ eq, allowing space for interpreting that it is as environmentally sustainable to dispose the lettuce as consuming it. Such communication pitfalls should be avoided.

3.3. Sensitivity analysis

The detailed results of the sensitivity analysis are presented in Appendix A10. The sensitivity analysis (S₁) is performed as a comparison to B_b. The waste management company handling the waste from the city of Lübeck reports a biogas production of 54 m³ biogas·kt⁻¹ for the SSBW, 144 m³ biogas·kt⁻¹ fine fraction for the residual waste, and a CH₄ potential of 0.35 Nm³·kg⁻¹VS (for the biowaste line). The CH₄ potential calculated in this study was on average for A_b and B_b 0.3 Nm³·kg⁻¹VS (for the biowaste line), implying correct utilization of literature. Improved sorting in A_d and B_d increased the calculated CH₄ potential to 0.38 Nm³·kg⁻¹VS in this study.

The results show that S₁ present a lower global warming mitigation potential (-3.4 kg CO₂ eq) compared to B_b (-11.6 kg CO₂ eq) for the SSBW and similar global warming potential for the RW (S₁ = 124 kg CO₂ eq, B_b = 120 kg CO₂ eq). Marine eutrophication (0.16 kg N eq), freshwater eutrophication (-0.06 kg P eq), and mineral resource scarcity (0.06 kg Cu eq) are also increased in S₁. This is attributed to the fact

that emission factors (Table 2) and consumption of energy and materials remain the same but the substitution of electricity and heat decreases in S₁. The difference between S₁ and B_b is more than threefold in all aforementioned impact categories.

Reducing the fraction of *others* by 50% (S₂), exemplified with the A_d, radically reduces the environmental impact of the residual waste while the impacts from the SSBW remain the same. The global warming potential of S₂ is calculated to 1 kg CO₂ eq (residual waste line = 64 kg CO₂ eq), reduced by 46% compared to A_d. The marine eutrophication caused by the residual waste was reduced by 40% (residual waste line: 0.03 kg N eq), the mineral resource scarcity by 83% (residual waste line = 64 kg Cu eq) while the freshwater eutrophication remains the same. The results imply that the waste types present, especially in the residual waste can influence the performance of the whole system.

3.4. Economic assessment

The results of the economic assessment are presented in Fig. 3 and the detailed results in Appendix A11. The results suggest that better sorting improves the economic performance of the system. For the SSBW, the NPV of A_b, A_d, and A_a is 12€, 141€, and 45€ while for B_b, B_d, and B_a it is -1 €, 110€, and 48€, per FU, respectively. For the residual waste, the NPV of A_b, A_d, and A_a is -1551 €, -1142 €, and -1434 € while for B_b, B_d, and B_a it is -1446 €, -1085 €, and -1205 € per FU, respectively.

Similar to the LCA results, the SSBW presents economic benefits while the residual waste economic costs. Major costs come from the incineration of the reject fraction while benefits come from the electricity generated.

In the current waste management practices in many German cities, which is mostly reflected in scenarios A_b and B_b, the costs are usually covered by waste fees. The improved economic performance in A_d, A_a, B_d and B_a implies that there could be reduction of the required fees or, alternatively, investments in more eco-friendly technologies and, thus, a more self-sustainable system.

4. Conclusions

The current study aimed at assessing the effect of the upstream, to biowaste management, determinants on the environmental and economic performance of biowaste management. Interventions in the waste collection infrastructure and waste sorting campaigns were conducted in two areas in the city of Lübeck to evaluate the potential for improved sorting and the results acquired were assessed through the life cycle assessment and the net present value methodology. The results of the study indicate that improved sorting has a positive effect on the environmental performance. For every kg of SSBW treated with anaerobic digestion, there is a global warming mitigation potential of $-0.2 \text{ kg CO}_2 \text{ eq}$.

Circular bioeconomy suggests that with cleaner loops there is the potential to improve the environmental performance of waste management, which is supported by this study. However, attention should be given to the fact that the first priority in the waste hierarchy is waste prevention and, thus, avoidable food waste should not be present in the waste or not be accounted as *no burden* materials. There is also the need for more case studies and further investigation on the performance of the system when only unavoidable food waste is present in the biowaste bin.

The economic assessment showed that biowaste treatment with electricity and organic fertilizer production can be economically viable (110–141€ per FU), if the impurity share is kept low (A_d , B_d). On the other hand, residual waste requiring incineration can create an economic burden.

There is great potential for improved environmental efficiency sourced from the consumer separation efficiency. The system changes evaluated and the sorting campaigns helped improved the sorting efficiency and reduce the impurities to less than 1%, excluding paper. However, these shifts in the sorting behavior should not be considered permanent. Instead, results indicate a need for sustained awareness-raising activities in order to maintain a high efficiency in household waste sorting.

Credit author statement

Steffen Walk and Ina Körner focused on the planning and implementation of the interventions and the waste sorting campaigns, as well as the acquisition of data from the Lübeck waste management company (EBL). Elisavet Angouria-Tsorochidou and Marianne Thomsen focused on the life cycle assessment, material flow analysis, and net present value modelling, as well as the data acquisition from literature.

CRedit authorship contribution statement

Elisavet Angouria-Tsorochidou: Conceptualization, Methodology, Investigation, Data acquisition, Validation, Writing – original draft, Writing – review & editing. **Steffen Walk:** Conceptualization, Methodology, Investigation, Writing – review & editing. **Ina Körner:** Conceptualization, Methodology, Writing – review & editing, Funding acquisition. **Marianne Thomsen:** Conceptualization, Methodology, Supervision, Writing – review & editing, Funding acquisition.

Data Availability

Data are included in Supplementary material.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at [doi:10.1016/j.clwas.2023.100092](https://doi.org/10.1016/j.clwas.2023.100092).

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