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LCA of a Consortium-Based MSW Management System to Quantify the Decrease in Environmental Impacts Achieved for Increasing Separate Collection Rates and Other Modifications

Giulia Costa *, Alessio Lieto and Francesco Lombardi

Laboratory of Environmental Engineering, DICII, University of Rome "Tor Vergata", Via del Politecnico 1, 00133 Rome, Italy; lieto@ing.uniroma2.it (A.L.); lombardi@ing.uniroma2.it (F.L.)

* Correspondence: costa@ing.uniroma2.it; Tel.: +39-06-72597007

Received: 28 March 2019; Accepted: 3 May 2019; Published: 16 May 2019

Abstract: In this study, the collection, transport, and treatment phases (including the management of products and processing residues) of six fractions of municipal solid waste (MSW) generated in the Sinistra Piave Basin (Veneto, Italy), a consortium of 44 municipalities, were analyzed by life-cycle assessment (LCA). Specifically, two different scenarios were assessed for paper and cardboard, glass, multi-material (plastics and metals), food waste, garden waste, and dry residual fraction management, one referring to the year 2015 and the other to 2004. The primary aim was to investigate what consequences the increase in separate collection rates progressively achieved by the consortium (65% in 2004 versus to 80% in 2015) exerted on the management system and its potential environmental impacts. For each scenario, the type of separate collection method employed (door-to-door in 2015, and mixed door-to-door and curbside collection in 2004), the collected amounts, the geographic location of the main sorting/treatment plants, and the type of treatments applied to manage the products and processing residues were considered. The results of the study indicate that, among the variations that occurred in the management system for the two considered years, the increase in separate collection rate achieved was the factor that most affected all of the potential environmental impacts taken into account. In particular, for the 2015 scenario, differently from the 2004 one, all of the categories considered (apart from ecotoxicity) were negative, indicating savings instead of impacts. Treatment was the stage that by far mostly affected potential environmental savings, with regard to paper and cardboard recycling in particular.

Keywords: municipal solid waste management; waste collection; separate collection; recycling; composting; anaerobic digestion; incineration; landfilling; life-cycle assessment

1. Introduction

Italy generated around 29.6 million tons of municipal solid waste (MSW) in 2017 [1]. As for the management systems adopted, very pronounced differences may be noted depending on the geographic area considered, and in general between northern and central or southern regions, both in terms of collection methods and treatment strategies applied. As for collection, increasing targets for the separate collection of specific waste streams to achieve by set timeframes were established [2]. Specifically, by 2012, the separate collection rate was set to reach 65%. The current national waste report states that, in 2017, 16.4 million tons of MSW was intercepted by separate collection, which means that an overall collection rate of just over 55.5% was attained countrywide [1], still below the 2012 target. However, northeast regions such as Veneto, Lombardy, and Trentino Alto Adige reached, and in some areas even greatly exceeded, this target [3,4]. High separate collection rates should lead to higher material recycling and, hence, lower overall environmental impacts of the waste management system (e.g., References [5,6]). Material recycling and recovery in fact divert waste from landfills and thermal treatment, thus leading to a reduction in impacts, including leachate production (in the case of landfilling) and greenhouse gas emissions [7]. Furthermore, recycled materials replace (at least partially) virgin raw materials and, therefore, lead to resource savings (e.g., energy, materials, and water), as well as avoiding emissions to the environment, related to extraction and processing. In a long-term perspective, recycling should lead to a decrease in the requirement of raw materials needed for production and, thus, postpone the exhaustion of scarce resources; however, in addition to the recycling rate, product lifetime and the production growth rate are important factors [8].

The European Union (EU) waste policy, hence, placed increasing emphasis on material recycling and set, within its recent circular economy package legislation, targets on recycling and preparation for reuse of MSW (e.g., 55% by 2025) and for the recycling of packaging materials [9,10]. Specifically, EU member states will be expected to achieve the following recycling rates by 2025: all packaging waste 65%, plastics 50%, wood 25%, ferrous metals 70%, aluminum 50%, glass 70%, and paper and cardboard 75% [10]. In addition, by 2035, MSW landfilling should be reduced to 10% or less of the total amount of generated municipal waste [11]. This means that, in the next few years, EU countries will need to develop and implement integrated waste management strategies (e.g., Reference [12]) that allow to reach high yields not only in terms of separate collection, but also of actual recycling and reuse of selected fractions.

In this study, we chose to focus on one of the most advanced waste management systems applied at a sub-regional level currently in Italy, the Sinistra Piave Basin, made up by 44 small- to medium-size municipalities of the province of Treviso (Veneto region). In 2015, the consortium that runs the waste management service in this area, serving a total population of around 300,000 inhabitants, was awarded first place as a waste recycling consortium in Italy by the environmental association Legambiente, since it achieved an 80% separate collection rate on average for its whole territory. The attainment of this result was made possible by the adoption of multiple measures, both technical and managerial, such as an efficient door-to-door collection service for the main waste fractions and the presence of 38 eco-centers located close to the main towns (average availability of one center per 3500 households) for the collection of the remaining fractions. In addition, the consortium developed communication strategies targeted both at citizens (also in foreign languages) and educational campaigns for children and students at schools, discouraging incorrect practices via regular controls, remote monitoring, and fines.

The main objective of this study was to quantitatively analyze the whole integrated management system applied in the Sinistra Piave Basin—from collection to recycling of specific MSW fractions and treatment of the residual ones—to evaluate the effects of the applied strategies and in particular of the separate collection yields achieved. Specifically, we decided to assess, for two different years for which different separate collection yields were achieved, the potential environmental impacts associated with the consortium-based management system.

The environmental performance of an integrated waste management system is typically evaluated by life-cycle assessment (LCA), a decision support tool that can be used in policy and decision-making to identify the system with the best performance through a comparative analysis of different scenarios that refer to a specific district or region (see Reference [13] and references within). For example, LCA was employed to compare 12 management scenarios, 10 of which differed only in the separate collection yields of specific fractions, in order to identify the system leading to the lowest environmental impacts for Avellino province in southern Italy; the results showed that the best performance could be achieved for the highest separate collection yield, with regard to paper and cardboard recycling in particular [5]. In another study, LCA was employed to analyze the environmental impacts of different integrated waste management strategies for two provinces in northern Italy (Piedmont region); separate collection and downstream recycling resulted in both cases as the most effective tools to improve energy efficiency and to lower environmental impacts [14]. Another LCA carried out for different provinces of Lombardy region in northern Italy analyzed both the environmental impacts related to the actual integrated waste management scenarios at the time of the analysis and those resulting from future modifications; an increase in the separate collection of recyclable fractions (and consequent decrease in residual waste for disposal) again yielded an overall great improvement and benefits at both the environmental and energy level [15].

LCA was employed also in this study to analyze the potential environmental impacts associated with an integrated system adopted to manage, in this case, the waste generated in the Sinistra Piave Basin. However, here, the two analyzed scenarios are not different options or future scenarios based on specific assumptions, but two past scenarios referring to two specific years characterized mainly by differences with regard to the type of waste collection system adopted and consequent separate collection yield achieved, as well as other differences concerning the location of treatment plants and the treatments adopted for residual fractions. Specifically, the analysis regarded the evaluation of the impacts related to the management of the following fractions: paper and cardboard, glass, multi-material (plastics and metals), food waste, garden waste, and the dry residual fraction. In the year 2015 (scenario 1), the collection of the abovementioned materials was carried out almost exclusively (98%) by a door-to-door system, whereas, in the year 2004 (scenario 2), a mixed collection method, leading to a lower separate collection yield, was adopted. The aim of this study was to quantify the environmental benefits resulting from the system for which the higher separate collection yields were achieved and to identify the system modifications that lead to the greatest savings in terms of impacts, in order to propose strategies that may yield further potential improvements. The results of the study can be of interest both for MSW management stakeholders and for citizens to motivate their contribution in increasing separate collection yields.

2. Materials and Methods

2.1. Scenarios and Assumptions

This study focused on three main elements of an integrated management system: collection, transport to the main sorting/treatments plants, and recycling/treatment/disposal carried out in both main and secondary plants. In Figure 1, a scheme of the analyzed system is reported. The treatment phase included the transport and sorting or treatment of the residues of the first plant to other specific facilities and the impacts of these second treatment units. Instead, the manufacturing of the containers employed for waste collection and the personnel mobility or construction and maintenance of offices were not included in the analysis.

Data regarding the collection, transport, and treatment of the six types of fractions considered in the study were supplied by the consortium and the local Environmental Protection Agency (ARPAV), as well as by the main plants that received and treated the waste generated in the Sinistra Piave Basin during the two years considered. These data include the following:

- The amounts and main composition of the waste collected in each of the 44 municipalities of the consortium;
- For the collection stage, the amounts collected, the type and frequency of collection, and the distances traveled by the different types of collection vehicles in 28 of the municipalities of the consortium;
- For the transport stage, the geographic location of the main treatment plants and the types of vehicles employed in 28 of the municipalities of the consortium;
- For the treatment phase, the specific characteristics of each type of plant, including energy consumption (or generation), the quantities of recovered materials or generated residues, and their final destination (recycling, treatment, or disposal).

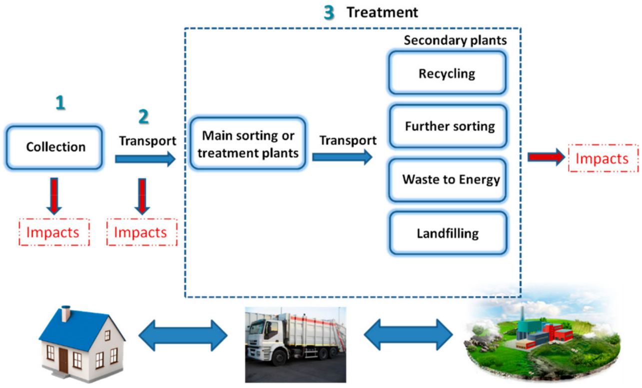


Figure 1. Scheme of the main management phases considered for each of the six waste fractions collected in the Sinistra Piave Basin in the two selected years.

2.1.1. Main Data on MSW Production

In Table 1, total MSW production data for the 44 municipalities of the Sinistra Piave Basin are reported with reference to the two years considered in this study. As can be noted, the population and total amount of waste collected in this area were quite similar in 2004 and 2015, yielding amounts of around 330–340 kg per inhabitant per year, significantly lower than national average values (i.e., 489 kg per inhabitant per year in 2017) [1], indicating the success of waste prevention and reduction activities carried out in the area.

The total amounts of the six fractions collected for the two selected years (see Table 1 and Figure 2) were also comparable, and both corresponded to 84% of the total amount of MSW collected in the consortium, while the other 16% was made up by bulky materials, batteries, and other items not collected at the household level. Except for the multi-material fraction, the amounts of the six main MSW fractions analyzed in this study varied significantly for the two years considered. In particular, glass almost doubled, while a 20–30% increase was found for paper and cardboard, food waste, and garden waste. Correspondingly, the amount of residual dry waste was less than half of that of the amount collected and managed in 2004.

Table 1. Key data on municipal solid waste (MSW) production in the Sinistra Piave Basin for the 2004 and 2015 scenarios, reporting the amounts and % contents of the selected fractions examined in this study [16,17].

Key Data	Scenario 2004		Scenario 2015	
Area extension (km^2)	1107.7		1107.7	
Number of inhabitants	298,304		308,365	
Total amount of MSW collected (t)	98,212		104,832	
	Selected fractions			
	Amounts (t)	**(%)**	**Amounts (t)**	**(%)**
Paper and cardboard	11,722	14.2	16,109	18.3
Food waste	18,914	22.9	23,347	26.5
Garden waste	9346	11.3	14,039	15.9
Multi-material (plastics and metals)	9059	11.0	9242	10.5
Glass	5999	7.3	11,852	13.4
Residual dry waste	27,378	33.2	13,579	15.4
Total amount of selected fractions	**82,419**	100.0	**88,169**	100.0

Since the total amount of selected fractions did not vary substantially in the two scenarios and, therefore, the decrease in residual waste was compensated by the increase of specific fractions such as glass, paper, and food waste, it can be concluded that the observed differences in waste composition can be ascribed to the different collection system adopted, in addition to changes in consumer habits.

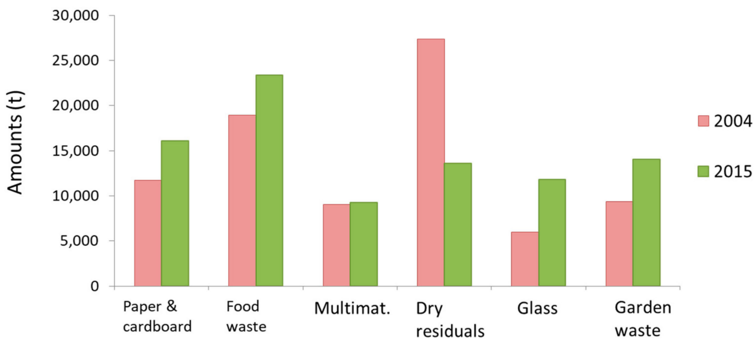


Figure 2. Amounts of the six selected waste fractions collected in the Sinistra Piave Basin in the two selected years.

2.1.2. Collection Stage

In 2015, a door-to-door collection system was applied for all 44 municipalities of the consortium with regard to the six considered fractions, except for garden waste which was predominantly (85%) collected at municipal eco-centers (or collection points). The door-to-door collection frequency applied was twice a month for all fractions, excluding food waste (twice a week) and glass (once a month).

For 28 municipalities, specific data regarding the collection of each of the five fractions collected door-to-door, i.e., type of vehicle employed, length of route, presence of a transfer station, etc., were available (see example reported in Table A1 of Appendix A for one municipality). For the other 16 municipalities, the kilometers traveled by vehicles for collecting each type of waste fraction were estimated on the basis of correlations between the length of the collection route and parameters such as the number of inhabitants, extension of the municipality, or amount of waste produced, on the basis of the data available for the other 28 municipalities. In Figure 3, two examples of such interpolations are reported. On the basis of these data, the frequency and capacity of the different types of collection vehicles considered, the amounts of each waste fraction transported, and the collection distance traveled were estimated. For garden waste, it was assumed that citizens went to the eco-center 12 times a year by car (50% gasoline, 35% diesel, and 15% methane), transporting each time around 35–45 kg of garden waste for a 1.5–5.5 km distance (depending on the extension of the municipality).

In 2004, the door-to-door collection system was applied only to collect food waste and the dry residual fraction, and it covered 65% of the population of the consortium, i.e., 29 municipalities out of 44. The remaining fractions (in addition to food waste and the dry residual fraction in 15 municipalities) were collected in curbside bins, except for garden waste, which was mostly collected in eco-centers. In this case, the frequency of collection of the door-to-door system was twice a week for food waste and once a week for the dry residual fraction. The curbside collection frequency was instead three times a week for the abovementioned waste types, while it was weekly for paper and cardboard and twice a week for multi-material and glass. Not having information on the collection routes employed in 2004, the same distances considered in 2015 were assumed for door-to-door collection for corresponding municipalities and waste fractions, while a 30% reduction was hypothesized in the case of curbside collection. This assumption was made on the basis of information provided by service operators and considering that the municipalities of the Bacino di Sinistra Piave are mostly in rural/agricultural areas and, hence, are characterized by a significant fraction of scattered houses.

Also the type of collection vehicles considered in the case of curbside collection differed; in this latter case, it was assumed that only vehicles with a transport capacity above one ton were employed, i.e., it was hypothesized that the small tipper, small dumper, and midi-compactor vehicle types were substituted by dumpers (capacity of 1.9 tons).

In Table A2 (Appendix A), the impacts in terms of ton × km related to the collection of each waste fraction and scenario as a function of the type of vehicle are reported. Figure 4a reports the percentage contribution to these impacts of each vehicle type.

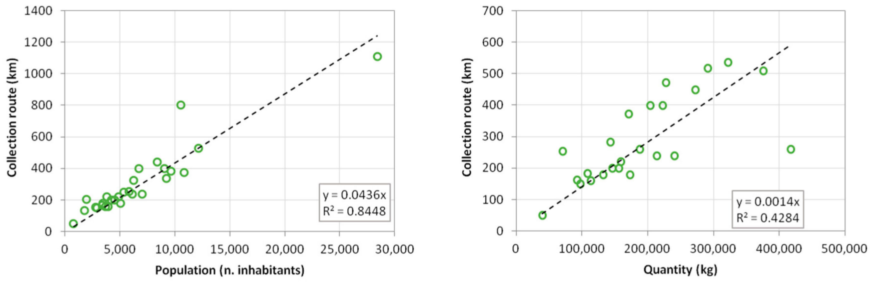


Figure 3. Examples of correlations employed to derive collection route data for (**a**) the dry residual fraction, and (**b**) glass.

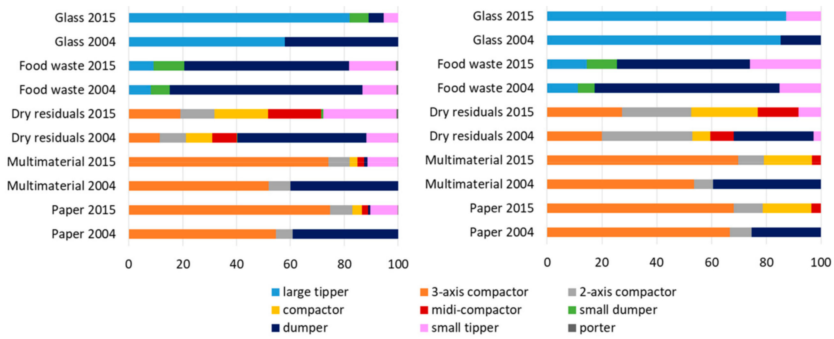


Figure 4. Percentage contribution to the impacts in terms of ton × km related to the (**a**) collection and (**b**) transport of each type of fraction for the two scenarios as a function of the vehicle type; vehicle capacities: large tipper (12 t), three-axis compactor (11.7 t), two-axis compactor (5.65 t), compactor (4.6 t), dumper (1.9 t), midi-compactor (0.7 t), small dumper (0.7 t), small tipper (0.65 t), and porter (0.435 t).

2.1.3. Transport Stage

From the analysis of the destination of each of the waste fractions collected in the Sinistra Piave Basin, it was observed that, in both 2015 and 2004, the six examined fractions were sent to several types of plants, located in or outside the consortium. Regarding the type of plants, paper and cardboard, multi-material, and glass were sent to sorting plants, the residual dry waste was sent to sorting plants (different from the ones to which the other fractions were sent), and food and garden waste was sent to composting plants, eventually equipped with anaerobic digestion. Hence, it was decided to simplify the analysis considering that each type of fraction would be sent only to one plant, selecting the location and characteristics of the plant to which most of each fraction was sent in each considered year. Specifically, it was assumed that paper and cardboard, multi-material, and glass were sent to the same sorting plant, the dry residual fraction to another sorting plant, and the food waste and garden waste fractions to a combined composting and anaerobic digestion plant.

With regard to the transport stage, for each of the waste fractions considered, the amounts of waste collected in each municipality and transported to the sorting or treatment plant, the transport frequency (which was assumed to be the same as the collection one), the type of vehicle employed, and the distance between the municipality and the plant were considered (see example reported in Table A1 of Appendix A).

For the 2015 scenario, the average distance between the plant treating the dry residual fraction and each municipality was 25 km, and that of the sorting facility receiving paper and cardboard, multi-material, and glass was 20 km, whereas the average distance of the composting and anaerobic digestion plant was 63 km. With regard to the 2004 scenario, the plant considered for the treatment of the dry residual fraction was the same as the one of 2015; hence, the same distances were considered.

Also, the selection facility considered for the other materials was close to the current one (average distance of 23 km), while the plant treating food and garden waste presented an average distance from each municipality of 120 km. In the calculation, it was considered that each waste vehicle returned (empty) to the municipality after transporting the waste.

In Table A2 of Appendix A, the impacts in terms of ton × km related to the transport of each waste fraction and scenario as a function of the type of vehicle are reported. Figure 4b reports the percentage contribution to these impacts of each type of vehicle type.

2.1.4. Treatment Stage

Based on 2015 data regarding the inputs and outputs of each of the three selection/treatment plants to which the six fractions were sent, the amounts of each type of material prepared for recycling, as well as the amounts of residues sent to treatment/disposal in other types of plants, were estimated, along with transport distances. Specifically, the data considered included the waste codes and amounts of all input and output flows, as well as the destination plants and their geographic location, in the case of outputs.

It should be noted that the treatment plant receiving the food and garden waste fractions in 2004 presented a similar process layout to the one of the 2015 scenario, but a lower fraction of the waste underwent anaerobic digestion (10% instead of 30%). The process consists of a first pressing step involving the input of humid food waste, from which the output solid fraction is mixed with garden waste before undergoing composting in aerated biocells, while the organic-rich liquid is treated by anaerobic digestion, producing heat and electricity. The liquid effluent of this latter treatment is treated biologically and by ultrafiltration and reverse osmosis, and then recirculated in the system. All the produced compost is used in agriculture as soil amendment and landscaping material. As for electricity production, based on data provided from the plants, yearly productions of 701 and 4049 MWh related to the treatment of the food and garden waste of the consortium were estimated for the 2004 and 2015 scenarios, respectively.

For the 2004 scenario, output data were only available for the sorting plant treating the dry residual fraction. Thus, for the other sorting plant, data were estimated based on 2015 data, considering a decrease in selectivity due to the different type of collection method applied. In fact, the results of composition analysis carried out on the collected paper and multi-material streams indicated a significant reduction in the amount of scraps in each of these fractions in 2015 compared to 2004, with scrap contents of 3.7% versus 11% for paper, and 25.6% versus 38% for multi-material, respectively.

For plastics selection, on the basis of data reported for the recycling of dry waste fractions in the Veneto region [18], it was assumed that the recovered plastics were made up by 70% polyethylene terephthalate (PET), 20% high-density polyethylene (HDPE), 7% low-density polyethylene (LDPE), and 3% polypropylene (PP).

To calculate the amounts of each material sent to recycling, the outputs of the main selection plant were multiplied by the reprocessing efficiencies reported by Reference [19] (i.e., 90% for steel, 83.5% for aluminum, 100% for glass, 89% for pulp production from paper, 75.5% for PET, 90% for HDPE, and 60% for LDPE and PP).

As shown in Table 2, regarding the types of treatments applied to the six waste fractions generated by the consortium, in 2004, overall more waste was processed by mechanical sorting, especially by sanitary landfilling. On the other hand, anaerobic digestion was much more applied in 2015, and there was a small increase in composting and incineration. It should be noted that the total amount of treated waste appears to be somewhat higher than the amounts of waste collected for each respective scenario reported in Table 1; this is due to the fact that part of the waste was treated in mechanical sorting plants more than once; for example, the mixed materials from the selection of the packaging waste were treated again for solid recovered fuel production.

As for material recycling, significantly more glass, as well as paper and cardboard, was sent to recycling in 2015. There was only a slight increase in the amounts of plastics and Al sent for recycling, while there was a reduction in the amount of Fe for recycling.

Table 2. Amounts of waste treated, sorted, or disposed of, as well as amounts of recycled fractions, for the two analyzed scenarios. PET—polyethylene terephthalate; HDPE—high-density polyethylene; LDPE—low-density polyethylene; PP—polypropylene.

	Scenario 2004 (t)	Scenario 2015 (t)
	Treated waste	
Sanitary landfilling	24,798.4	6648.7
Incineration	8735.1	9737.1
Mechanical sorting	66,597.3	60,759.1
Composting	25,434.4	26,170.2
Anaerobic digestion	2826.0	11,215.8
	Recovered fractions	
Glass for recycling	5990.1	11,835.1
Paper for recycling	8867.7	14,024.3
Fe for recycling	1075.1	885.7
Al for recycling	87.4	92.5
PET for recycling	2151.6	2832.9
HDPE for recycling	732.8	964.8
LDPE for recycling	171.0	225.1
PP for recycling	73.3	96.5
Compost for recycling	4280.8	5301.8

2.2. LCA Methodology

As a functional unit, the total amounts of the six fractions of MSW collected in the Sinistra Piave Basin in each examined year were considered. This unit was chosen even though the total amounts differed slightly for the two analyzed years (88,000 t in 2015 versus 82,000 t in 2004; see Table 1) due basically to a small increase in population in 2015, since the objective was to evaluate and compare the total impacts related to the integrated management system of the consortium for each year.

The LCA analysis was carried out using the Simapro software version 8 (Pré Consultants) and the Ecoinvent database (versions 3 [20] or 2 [21]) for background data (i.e., Italian energy mix, collection vehicles, and waste treatment facilities such as selection plant, sanitary landfill, and waste incinerator with electricity and heat co-generation; see Table A3, Appendix A). For all collection and transport vehicles, Euro 4 diesel models were assumed. Allocational processes were selected, making reference mostly to global or European scenarios, or Italian ones in the case of the energy mix (see Table A3, Appendix A).

For the food and garden waste fractions, the treatment process was modeled since a representative one was not retrieved in the database (see Table A4, Appendix A).

In this analysis, it was assumed that compost would be used in replacement of peat moss for horticultural applications with a 1:1 substitution rate. For the other recycled materials, the following replacements were assumed [19]: recovered Fe would replace liquid steel (1:1); recovered Al would replace Al ingot (1:1); recovered glass would substitute glass containers (1:1); recovered paper and cardboard would replace pulp by thermomechanical processes (1:0.83); and recovered plastics would substitute the respective granules of PET, HDPE, LDPE, and PP (1:0.81). For each of these materials, the production processes reported in the Ecoinvent database were considered (see Table A3, Appendix A).

For the analysis of potential environmental impacts, the International Reference Life-Cycle Data System (ILCD) methodology for impact characterization and assessment [22], recommended for the European context as reported by References [23,24], was employed.

In this paper, we present the results achieved with regard to the following selected impact categories among the 16 included in the ILCD 2011 midpoint method (V1.03): global warming, acidification, photochemical smog production, eutrophication, human toxicity (carcinogenic effects), ecotoxicity, and consumption of non-renewable resources. These categories were selected as being representative of the main different types of impacts included in this impact assessment methodology.

3. Results and Discussion

3.1. Recycling Yields Achieved

On the basis of the data reported in Tables 1 and 2, the recycling yields resulting for the Sinistra Piave basin, with reference to the six considered waste fractions, were evaluated and compared to the targets established by the recent EU legislation [9–11]. The calculation was performed applying the rules reported in the respective directives. In particular, for MSW recycling, apart from the amounts of paper and cardboard, glass, metals, and plastics for recycling, the amount of biodegradable fractions (food and garden waste) entering aerobic or anaerobic treatment was counted as recycled for both scenarios, since the treatment plants generated compost that was used as a recycled product.

As shown in Table 3, a substantial increase in recycling rates from the 2004 to the 2015 scenario was achieved in terms of MSW, packaging waste (calculated as the sum of paper and cardboard, glass, and multi-material for recycling divided by the amounts of these fractions collected in the consortium), and paper and cardboard recycling. Also the four-fold decrease in landfilling attained in 2015 compared to 2004 should be highlighted. As for plastics and metal recycling, it was not possible to calculate the separate recycling yields achieved, since in both scenarios these materials were collected together as multi-material; therefore, even if the amounts of each type of material sent to recycling were estimated, their initial content was not known.

With regard to the recently established circular economy targets, for the 2004 scenario, most 2025 targets were already met (in particular for glass and packaging waste recycling). This was even more true for the 2015 scenario which appeared also to comply with the 2035 target on landfilling.

Table 3. Recycling yields for the two analyzed scenarios and European Union (EU) targets for 2025 (recycling) or 2035 (landfilling) [9–11].

	Scenario 2004 (%)	Scenario 2015 (%)	Targets (%)
MSW for recycling	57.5	77.5	≥55
Packaging waste for recycling	71.5	83.2	≥65
Paper for recycling	75.7	87.1	≥75
Glass for recycling	99.9	99.9	≥70
Multi-material (plastics and metals) for recycling	47.4	55.2	≥50 plastics and Al, ≥70 Fe
Landfilling	30.1	7.5	≤10

3.2. LCA Results

3.2.1. Overall Impacts

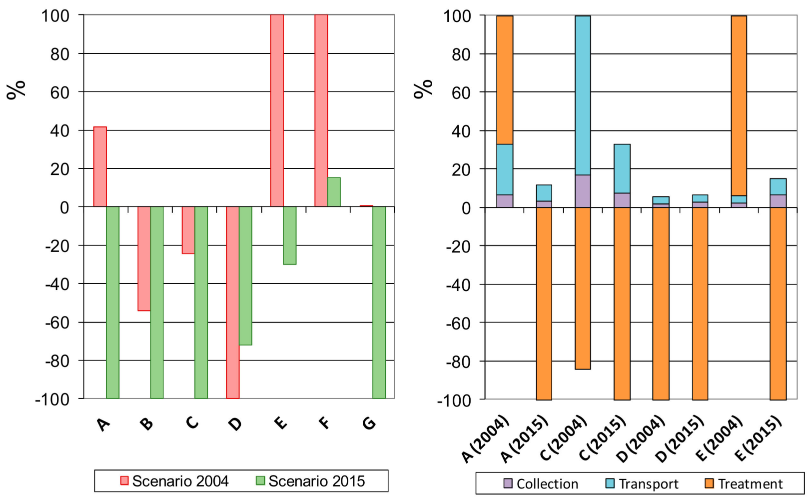
The comparison of the overall impacts due to the management of the six waste fractions collected in the Sinistra Piave Basin in the considered scenarios is shown in Table 4 and Figure 5a for the selected impact categories. The results for all the impact categories considered in the ILCD 2011 midpoint method are reported in Table A5 of Appendix A. It is very interesting to note that, for the 2015 scenario, all considered categories apart from freshwater ecotoxicity were negative (i.e., indicated savings instead of impacts). Instead, for the scenario referring to 2004 data, the climate change, human toxicity, and resource depletion impacts were positive. It is also interesting to note that the negative impacts attained for the 2015 scenario were higher in absolute value than those of the 2004 scenario, except for eutrophication. The fact that high separate collection rates result in negative impacts is in agreement with the findings reported by many previous studies (e.g., References [5,14,15,25,26]).

As shown in Figure 5b, the treatment stage was the management phase that most influenced impacts (both negative and positive ones), while the collection stage led to almost negligible impacts for both scenarios. The transport to the main treatment/sorting plants resulted in significant impacts in the case of photochemical ozone formation and climate change, especially for the 2004 scenario.

Table 4. Overall impacts for selected categories resulting for the two analyzed scenarios, with NMVOC: Non Methane Volatile Organic Compounds; CTUh: Comparative Toxic Unit for humans; CTUe: Comparative Toxic Unit for the ecosystem.

Impact Category	Unit	Scenario 2004	Scenario 2015
Climate change (A)	kg CO_2 eq	14.3×10^6	-34.6×10^6
Acidification (B)	molc H^+ eq	-1.4×10^5	-2.6×10^6
Photochemical ozone formation (C)	kg NMVOC eq	-2.3×10^4	-9.6×10^4
Freshwater eutrophication (D)	kg P eq	−3821.1	−2748.3
Human toxicity, cancer effects (E)	CTUh	1.56	−0.47
Freshwater ecotoxicity (F)	CTUe	8.99×10^9	1.37×10^9
Mineral, fossil, and renewable resource depletion (G)	kg Sb eq	2.3	−329.5

In order to identify the modifications in the waste management system that led to the highest reductions of potential environmental impacts and the strategies that could be applied to further improve the environmental footprint of the consortium's activities, the contributions of the single processes and waste fractions to the impacts for the selected categories were examined. In the next paragraphs, the comparisons of the impacts determined for each scenario with regard to the collection, transport, and treatment stages, as well as the main phases and/or fractions contributing to the specific impacts for each management phase, are reported and discussed.



(a) (b)

Figure 5. Environmental impacts (see Table 4) in terms of characterization for the two scenarios considered: (**a**) overall comparison; (**b**) analysis of the contribution to four types of impacts of the three main management phases.

3.2.2. Collection and Transport Stages

In Figure 6, the potential impacts of the collection and transport stages for the two scenarios are compared in terms of characterization. As can be noted, for each of the analyzed impact categories, the resulting impacts for collection and transport were quite comparable, with transport for the 2004 scenario always leading to the highest impacts.

Regarding the collection phase, it can be seen how, for air pollution-related categories, the two scenarios gave very similar impacts, while, for the others (freshwater ecotoxicity and resource depletion in particular), higher impacts were found for the 2015 scenario. These results may be linked to the fact that overall the impacts in terms of tons of waste transported multiplied by the distance traveled (t × km) were similar in the two scenarios. However, the types of collection vehicles considered in the two scenarios were different especially for some fractions (see Figure 4). The impacts related to one t × km transport of these vehicles reported in the Ecoinvent database compared to the larger ones considered for curbside collection were higher especially for toxicity, ecotoxicity, and resource use.

As for the transport stage, all impacts determined for the 2015 scenario were at least 15% lower compared to those of 2004. This can be related mostly to the fact that the plant treating the food and garden waste in the 2015 scenario was located closer (on average, half of the distance) than the plant that had the same function in the 2004 scenario. Effects of the distance of plant location were also reported in a recent study comparing the environmental performance over the years of an integrated waste management system for a Brazilian case study [27].

Regarding the contribution to the collection and transport impacts from the single waste fractions (see Figure 7), it can be noted how, for materials such as glass and paper for which a significantly higher collection rate was attained in 2015, higher impacts were achieved with respect to 2004 for both collection and transport. Instead, a significant reduction in the contribution of the dry residual fraction was found for 2015 collection and transport impacts compared to 2004.

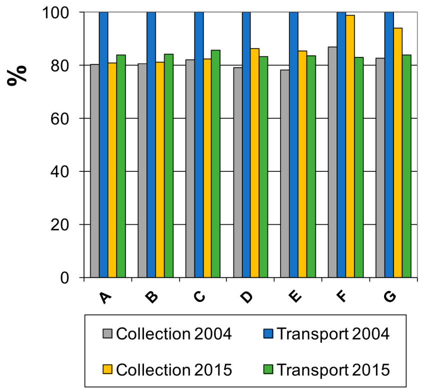


Figure 6. Comparison of impacts (see Table 4) in terms of characterization for the collection and transport stages for the two scenarios considered.

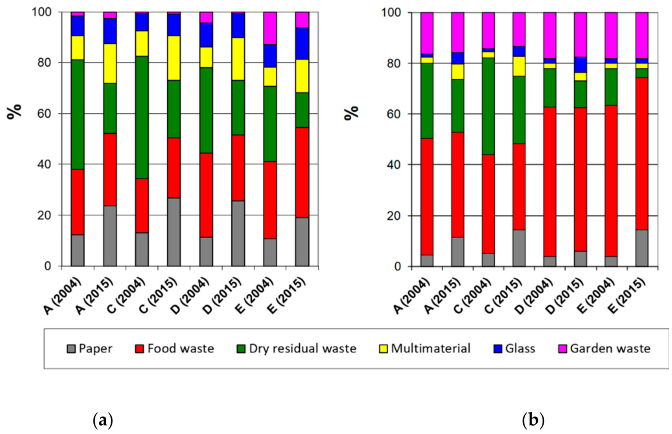
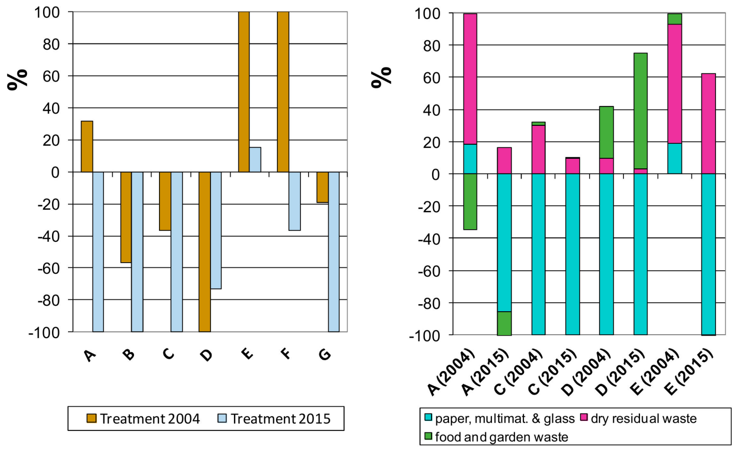


Figure 7. Contributions to the impacts (see Table 4) of (**a**) the collection and (**b**) transport phases in terms of the different fractions for four types of impacts and the two scenarios.

Looking at the contribution to collection impacts, it can be seen (Figure 7a) how, for the 2004 scenario, food and garden waste collection led to over 60% of impacts, while, for the 2015 scenario, there was a more equal distribution of the impacts among the paper, residual, and food waste fractions (in particular for photochemical smog formation and eutrophication). As for the contributions of the different fractions to transport impacts, in Figure 7b, it can be noted that, for all impacts (human toxicity and eutrophication in particular), food waste gave by far the highest impacts for both scenarios. This can be ascribed to the amounts collected, and especially to the location of the treatment plant of these fractions, with an average distance from the consortium more than three or six times higher, for the 2015 and 2004 scenarios, respectively, than that of the sorting plant receiving paper, multi-material, and glass.

3.2.3. Treatment Stage

As shown in Figure 8, differently from the other two analyzed stages, treatment led to savings instead of impacts for most categories, in particular for the 2015 scenario. For the 2004 one, overall positive impacts were found for climate change, ecotoxicity, and human toxicity. This can be related, on the one hand, to the lower material recycling and energy recovery obtained in this case due to the different type of collection method adopted, and, on the other hand, to the higher use of landfilling, in agreement with previous studies [25–28].

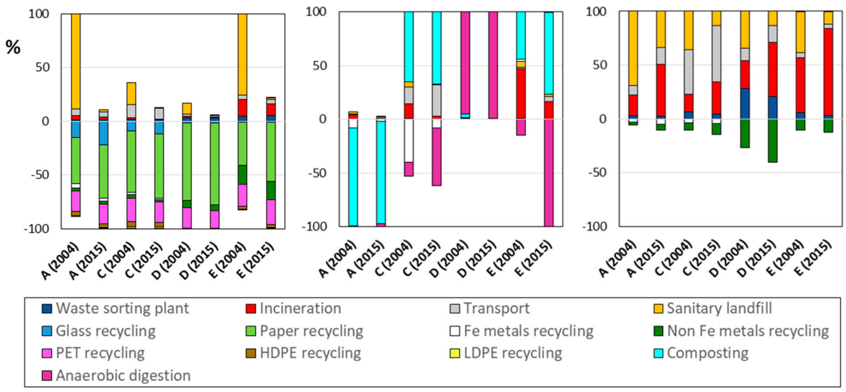


(**a**) (**b**)

Figure 8. (**a**) Comparison of impacts (see Table 4) in terms of characterization for the treatment stage for the two scenarios considered; (**b**) analysis of the contribution to four types of impacts of the treatment of the different types of fractions.

From Figure 8b, it is quite evident that, while the management of the dry residual fraction led to impacts for all categories, paper, multi-material, and glass management yielded savings (apart for global warming and human toxicity for the 2004 scenario), while food and garden waste treatment resulted in negative impacts only for global warming and significant impacts for eutrophication. Hence, it can be noted that almost all savings were achieved for the treatment of the fractions that are recycled in substitution of raw materials (apart from the reduction in climate change attributable to energy recovery from the anaerobic digestion of biowaste). The impacts related to residual waste management were much higher in the 2004 scenario due both to the fact that a higher amount of waste was treated and to the higher use of landfilling as a treatment strategy.

Examining more in detail the processes leading to the most significant impacts or savings for the management of packaging materials (see Figure 9a), it can be noted that the recycling of paper in particular, in addition to PET and glass (for climate change), or non-ferrous metals (for human toxicity), gave the highest negative impacts, while landfilling was the process that yielded the most relevant impacts (for the 2004 scenario in particular). For food and garden waste treatment (see Figure 9b), composting yielded negative impacts with regard to climate change, owing to the substitution of peat moss, but yielded positive impacts for photochemical smog and human toxicity, due to air emissions of the composting process. Anaerobic digestion resulted instead in savings for photochemical smog and human toxicity, due to electricity production substituting national grid one; however, it was the main contributor of eutrophication impacts, in relation to the water treatment process. In addition, the recovery of ferrous metals contributed to negative emissions, in particular for photochemical smog in 2004. As for the management of the dry residual fraction, as can be noted in Figure 9c, incineration yielded the most significant contribution to impacts for human toxicity in both scenarios, and eutrophication and climate change for the 2015 scenario. Landfilling was shown to contribute especially to the climate change impacts of the 2004 scenario, whereas the transport of the material to the treatment plants had a major role in photochemical smog impacts for both scenarios. Finally, as for negative impacts, the most relevant, in particular for eutrophication, was non-ferrous metal recycling.



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

Figure 9. Analysis of the contribution to four types of impacts (see Table 4) for (**a**) paper, multi-material, and glass treatment; (**b**) food and garden waste treatment; and (**c**) dry residual waste management.

4. Conclusions

This LCA study was aimed at evaluating the environmental impacts of a consortium-based waste management system with regard to six MSW fractions and two different years. The aim was to investigate the effect of the increase in separate collection rates achieved in over 10 years on potential environmental impacts, and to identify the sub-processes that are more critical in terms of impacts for one or more categories, thereby identifying potential further improvements that could be implemented. Overall, the results of the study indicated that an efficient separate collection system, coupled with plants employing technologies that allow obtaining material and energy recovery, may lead to great benefits in terms of the reduction of environmental impacts.

The overall recycling efficiencies achieved appear to already be in line with the EU targets for 2025. An exception may be plastics and metals that were collected together as a multi-material fraction and for which it was not possible to separately evaluate the recycling yields achieved. In order to be able to better monitor the amounts collected, treated, and prepared for recycling for each of these types of materials, dedicated separate collection strategies could be implemented also for specific types of plastics presenting the highest recycling potential (e.g. PET). In this study, a rough estimate was made with regard to the composition of the recovered plastics; therefore, it would be important in the future to analyze the average composition of the collected plastics and the actual recycling rates achieved for each type.

The collection and transport stages were shown to exert a lower contribution to impacts (the former in particular) compared to treatment. However, in addition to the optimization of the geographic location of the treatment plants, another improvement that may reduce the impacts of these phases is the change in type of collection/transport vehicles employed. To this regard, the consortium started substituting its vehicles with biomethane-fueled ones, since the anaerobic digestion plant treating the organic waste fraction was recently equipped with a biogas upgrading unit.

As for the treatment stage, material recycling, which was significantly promoted by the change in type of collection method, was shown to be a key contributor for reducing impacts. As for the treatment/disposal of residues, although there was an almost four-fold decrease in landfilling, ecotoxicity impacts were still positive. Therefore, in order to allow reducing impacts even more, further waste diversion from landfills, if feasible, should be pursued. With regard to the better performance found for eutrophication for the 2004 scenario compared to the 2015 one, it should be considered that the wastewater treatment model process that was used to estimate the emissions is less advanced with respect to the one implemented in the plant receiving the consortium's organic waste; therefore, the impacts may have been overestimated. Also, in this case, a more accurate modeling of the process based on plant data could be performed.

Author Contributions: G.C. and F.L. conceived and designed the study; G.C., A.L., and F.L. analyzed the data and performed the calculations; G.C., A.L., and F.L. wrote the paper. G.C. revised the paper.

Funding: This research was partly funded by Consorzio per i Servizi di Igiene del Territorio—BacinoTV1.

Acknowledgments: The authors would like to acknowledge the important contributions received by Lorena Franz and Lucio Bergamin (ARPA Veneto), Gianpaolo Vallardi, Piergiorgio Tonon, and Roberta Nardi (Consorzio per i Servizi di Igiene del Territorio and Consiglio di Bacino "Sinistra Piave"), and Paola Muraro (ATIA ISWA Italy) with regard to the data and assumptions of the study. The authors would also like to acknowledge the contribution of the student Marco Popolla for the elaboration of the data.

Conflicts of Interest: The authors declare no conflicts of interest. The funders provided only the data on waste collection and transport and had no role in the design of the study, the analyses, or interpretation of the data, in the writing of the manuscript, or in the decision to publish the results.

Appendix A

Table A1. Example of data on collection and transport of the five considered waste fractions for one municipality referring to the year 2015.

Fraction	Yearly Collected Amount [kg]	Type of Collection	Annual Collection Frequency	Type of Vehicle	Vehicle Capacity [kg]	Length of Collection Round [km]	Waste Transfer to Other Vehicle	Transfer Station	Distance to Treatment Plant [km]	Transport Distance [km]
Paper & cardb.	140,050	door to door	24	3-axis compactor	11700	80	NO	NO	36	36
Paper & cardb.	140,050	door to door	24	Small tipper	650	80	YES	NO	36	-
Multimaterial	113,290	door to door	24	3-axis compactor	11700	80	NO	NO	36	36
Multimaterial	113,290	door to door	24	Small tipper	650	80	YES	NO	36	-
Dry residuals	145,700	door to door	24	3-axis compactor	11700	-	NO	YES	37	50
Dry residuals	145,700	door to door	24	Midi-compactor	700	90	NO	NO	37	37
Dry residuals	145,700	door to door	24	Small tipper	650	50	YES	NO	37	-
Food waste	237,040	door to door	104	Dumper	1900	100	NO	YES	78	91
Food waste	237,040	door to door	104	Dumper	1900	101	NO	YES	78	91
Glass	112,700	door to door	12	Large tipper	12000	-	NO	NO	36	36
Glass	112,700	door to door	12	Small tipper	650	80	YES	NO	36	-
Glass	112,700	door to door	12	Small tipper	650	80	YES	NO	36	-

Table A2. Impacts in terms of ton × km related to the yearly (a) collection and (b) transport of each type of vehicle, waste fraction, and scenario.

Collection (t*km)												
	Paper 2004	**Paper 2015**	**Multimat 2004**	**Multimat 2015**	**Dry res 2004**	**Dry res 2015**	**Food w 2004**	**Food w 2015**	**Glass 2004**	**Glass 2015**	**Garden w 2004**	**Garden w 2015**
large tipper	0	0	0	0	0	0	88,489	101,700	268,551	448,223	0	0
3-axis compactor	339,810	694,720	225,984	463,762	120,384	65,134	0	0	0	0	0	0
2-axis compactor	39,272	77,208	34,648	50,033	102,134	42,248	0	0	0	0	0	0
compactor	0	31,960	0	18,175	102,822	67,838	0	0	0	0	0	0
midi-compactor	0	21,449	0	15,747	93,609	67,079	0	0	0	0	0	0
small dumper	0	0	0	0	3042	2,674	74,778	128,387	0	38,505	0	0
dumper	243,961	8581	174,239	6950	501,062	0	772,267	685,074	195,502	30,045	0	0
small tipper	0	93,714	0	70,175	121,758	91,917	136,634	193,734	0	29,506	0	0
porter	0	1,880	0	1440	2680	1909	6633	9517	0	0	0	0
private cars	0	0	0	0	0	0	0	0	0	0	28,820	44,800
tot	*623,043*	*929,511*	*434,871*	*626,282*	*1,047,491*	*338,800*	*1,078,801*	*1,118,411*	*464,053*	*546,279*	*28,820*	*44,800*
Transport (t*km)												

Table A2. *Cont.*

	Paper 2004	Paper 2015	Multimat 2004	Multimat 2015	Dry res 2004	Dry res 2015	Food w 2004	Food w 2015	Glass 2004	Glass 2015	Garden w 2004	Garden w 2015
large tipper	0	0	0	0	0	0	271,408	229,173	134,402	227,908	0	0
3-axis compactor	178,833	201,196	73,692	117,766	131,187	88,160	0	0	0	0	0	0
2-axis compactor	21,038	31,221	9432	15,773	215,764	81,356	0	0	0	0	1,037,617	837,896
compactor	0	52,498	0	29,824	42,286	78,651	0	0	0	0	0	0
midi-compactor	0	10,346	0	5515	56,760	48,051	0	0	0	0	0	0
small dumper	0	0	0	0	0	0	146,263	175,380	0	0	0	0
dumper	68,192	0	54,092	0	190,253	0	1,632,360	773,544	23,159	0	0	0
small tipper	0	0	0	0	17,642	26,201	367,829	414,172	0	33,089	0	0
porter	0	0	0	0	0	0	0	0	0	0	0	0
private cars	0	0	0	0	0	0	0	0	0	0	0	0
tot	***268,063***	***295,261***	***137,216***	***168,878***	***653,892***	***322,418***	***2,417,859***	***1,592,268***	***157,560***	***260,997***	***1,037,617***	***837,896***

Table A3. Ecoinvent processes selected for inventory modeling.

Modelled Process	Ecoinvent Process Selected
Collection and transport vehicles	
Large tipper	Freight lorry, 16–32 mt, Euro 4, Glo, alloc def. U
3-axis compactor	Freight lorry, >32 mt; Euro 4, Glo, alloc def. U
2-axis compactor	21 mt lorry for waste collection service Glo, alloc def. U
Compactor	21 mt lorry for waste collection service Glo, alloc def. U
Dumper	Freight lorry, 7.5–16 mt, Euro 4, Glo, alloc def. U
Midi-compactor	Freight lorry, 3.5–7.5 mt, Euro 4, Glo, alloc def. U
Small dumper	Freight lorry, 3.5–7.5 mt, Euro 4, Glo, alloc def. U
Small tipper	Freight lorry, 3.5–7.5 mt, Euro 4, Glo, alloc def. U
Porter	Freight lorry, 3.5–7.5 mt, Euro 4, Glo, alloc def. U
Treatment and recovered materials	
Sorting	Waste sorting plant, RER, alloc def. U
Incineration	MSW incineration treatment, Italy, alloc def. U
Landfilling	MSW sanitary landfill, alloc def. U
Transport vehicle to treatment plant	Freight lorry, 16–32 mt, Euro 4, Glo, alloc def. U
Recovered glass	Container glass, production mix at plant RER S
Recovered paper and cardboard	Thermo-mechanical pulp {GLO}Alloc Def, U
Recovered Iron	Steel, liquid, at plant/RNA

Table A3. *Cont.*

Modelled Process	Ecoinvent Process Selected
Recovered Aluminum	Aluminum, primary, ingot {GLO}| production alloc def. U
Recovered PET	Polyethylene terephthalate, granulate, bottle grade {GLO}| alloc def. U
Recovered HDPE	Polyethylene, high density, granulate {RER}| production | alloc def. U
Recovered LDPE	Polyethylene, low density, granulate {RER}| production | alloc def. U
Recovered PP	Polypropylene, granulate {RER}| production | alloc def. U

Table A4. Modeling of the food and garden waste processing plant treating one ton of waste.

Inputs/Processes/Emissions	Amounts
Biowaste composting	
Inputs/processes	
Peat moss for horticultural use	−2 m^3
Diesel	1.4 kg
Composting facility	4×10^{-6} p
Electricity low voltage, Italy	0.024 MWh
Emissions to air	
CO_2 fossil	4.55 kg
CO_2 biogenic	280 kg
Ammonia	0.011 kg
Fine particle matter (PM_{10})	0.021 kg
H_2S	0.00116 kg
Volatile Organic Compounds (VOC)	0.0174 kg
Biowaste anaerobic digestion	
Inputs/processes	
Electricity low voltage, Italy	−0.36 MWh
Treatment of wastewater from anaerobic digestion of whey	8.45 m^3
Anaerobic digestion plant with methane recovery	6×10^{-5} p
Electricity low voltage, Italy	0.0102 MWh
Emissions to air	
CH_4	1 kg

Table A5. Overall environmental impacts of the two analyzed scenarios for all categories included in the International Reference Life-Cycle Data System (ILCD) 2011 midpoint method.

Impact Category	Unit	Scenario 2004	Scenario 2015
Climate change	kg CO_2 eq	14.3×10^6	-34.6×10^6
Stratospheric ozone depletion	kg CFC-11 eq	−0.5	−1.5
Human toxicity, cancer effects	CTUh	1.6	−0.5
Human toxicity, non cancer effects	CTUh	113.1	8.0
Particle matter	kg $PM_{2.5}$ eq	-1.4×10^4	-2.5×10^4
Ionizing radiation	kBq U235 eq	-3.7×10^6	-6.7×10^6
Ionizing radiation E (interim)	CTUe	−9.0	−18.3
Photochemical ozone formation	kg NMVOC eq	-2.3×10^4	-9.6×10^4
Acidification	molc H^+ eq	-1.4×10^5	-2.6×10^6
Terrestrial eutrophication	molc N eq	-1.2×10^5	-3.1×10^5
Freshwater eutrophication	kg P eq	−3821.1	−2748.3
Marine eutrophication	kg N eq	1.4×10^5	4.8×10^5
Freshwater ecotoxicity	CTUe	9.0×10^9	1.4×10^9
Land use	kg C deficit	-5.0×10^6	-3.0×10^7
Water use	m^3 water eq	-3.3×10^7	-6.0×10^7
Mineral, fossil, and renewable resource depletion	kg Sb eq	2.3	−329.5

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