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Article

Environmental Impacts of Electricity from Incineration and Gasification: How the LCA Approach Can Affect the Results

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Abstract: Waste-to-energy (WtE) technologies can offer sustainable solutions for waste, which can no more be reused or recycled, such as the part of municipal solid waste (MSW) that is not suitable for recycling processes. This study focused on the environmental consequences of the production of electricity from incineration and gasification of MSW. To this aim, the standardised life cycle assessment (LCA) methodology was used. A life cycle inventory, mainly composed by primary data, is provided. Starting from these data, different highly shared LCA approaches were used to calculate the potential impacts of 1 kWh provided by the two analysed WtE technologies. The different approaches concern the method of accounting for the by-products (through an economic allocation and a system expansion) and the inclusion/exclusion of environmental benefits due to the avoided landfill for the MSW. For each approach, impact-assessment results were calculated with the ReCiPe midpoint (H) method. A comparison was carried out (i) between the results obtained for the same WtE technology but calculated with different approaches and (ii) between the impact results of electricity generated by the two WtE technologies calculated with the same approach. From the study, it emerged that, according to the accounting rules, the impact results can significantly change and, for some impact categories, even lead to opposite conclusions. In the absence of category rules that harmonise the environmental assessments of WtE processes, it is therefore recommended that the development/use/reproduction/comparison of studies focused on the valorisation of waste should be carried out with caution.

Keywords: LCA; incineration; gasification; environmental impacts; municipal solid waste



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1. Introduction

Municipal solid waste (MSW) is a global relevant issue, especially in developing countries [1]. Good management is the key for the transition to environmental, economic, and social sustainability. As part of this transition, the paradigm of waste is changing direction: if in a linear economy it exclusively represents a burden, in a circular economy, it could still represent a resource.

The global generation of MSW amounts to 2.01 billion tonnes per year and is expected to reach 3.40 billion tonnes by 2050 [2]. As a consequence, the production of energy from waste that cannot be reused or recycled can represent a solution in line with the principles of the circular economy and can contribute to the energy diversification [3].

The waste-to-energy (WtE) process is currently provided by different technologies, such as the anaerobic digestion, the production of waste-derived fuels, the (co-)incineration in combustion plants and in cement and lime production or in dedicated facilities, and the indirect incineration following a pyrolysis or gasification step.

Among the WtE technologies, incineration is the most-established process, accounting more than 1400 plants worldwide [4]. Incineration or “direct combustion” is the complete, rapid exothermic oxidation of the waste organic fraction in the presence of an adequate excess of oxygen. An incinerator for MSW is generally composed by a furnace, an after-burning chamber, a heat recovery steam generator, and an emission-control equipment [5]. Incinerators work with many different types of waste, including MSW [6–8], products after

their use phase (such as end-of-life tyres, [9]), solid refuse fuels (SRF) [10,11], industrial waste (IW) [12,13], and industrial hazardous waste (IHW) [14]. Beyond being a solution for the waste management, the incineration provides heat and can generate steam and electricity [15].

Another WtE technology is the gasification, or “indirect combustion”, which refers to a MSW thermochemical decomposition that generates combustible gas (syngas) and a subsequent combustion for energy recovery (two-step oxidation) [16]. As explained in the report of a European project, Germany and Italy have the most-important number of gasification plants, while Scandinavian Countries have the largest plants [17]. Gasification is a thermochemical conversion process of carbon-based feedstocks [18], which can process a wide variety of feedstocks, ranging from biomass to municipal and other solid waste. The organic content of the waste is converted mainly to carbon monoxide, hydrogen, and lower amounts of methane, although the syngas is generally contaminated by undesired products such as particulate, tar, alkali metals, chloride, and sulphide. The obtained syngas can be used to produce chemicals (such as fertilizers [19]) and fuels [20,21] or for power generation [21,22]. As emerges from the study of Cerón et al. [23], the output gas composition and its quality are influenced by the type of biomass and the process parameters. Recent literature analysed the gasification of wheat straw, coconut shells, groundnut shells, and corncobs [19], as well as the sustainability of poultry litter gasification in comparison with Miscanthus and waste wood gasification [24].

Direct emissions from the thermo-chemical conversion technologies are air pollutants such as precursors of SO_x, NO_x, HCl, particulate matter, components of hydrocarbon, dioxins, and CO₂ [4,25]. Significant reductions in air pollution are achievable through dedicated flue-gas-treatment technologies [26].

The environmental impacts of WtE have been studied by different research groups through the internationally standardised methodology of life cycle assessment (LCA) [27,28]. However, the results are not always comparable, mainly because of the different approaches that have been used for the assessment, as detailed in Section 2.

This study considered the production of electricity from MSW and compared the environmental performances of 1 kWh of electricity produced from the incineration in a combustion plant and from the gasification with a direct melting system (DMS), respectively. Particular attention was paid to understand how the results can be affected by the approach and the system boundaries of the analysis. This analysis contributes therefore to the debate about WtE treatments, providing detailed assessments mainly based on primary data. The results of this study supplement with new data and perspectives the literature already available on the topic of WtE.

2. Methods: Life Cycle Assessment of WtE Technologies

A quite abundant number of sustainability assessments of WtE are available in the literature. However, different aims, methodological approaches, and focuses of the studies often hinder their comparability.

First of all, the comparability depends on the choice of the functional unit (FU) to which the results are referred. The choice of the FU is directly connected to the goal of the study: if, generally speaking, the FU of LCA studies is referred to the output product(s) of a process/system, in the field of waste management, it can be of interest to assess the impacts with reference to the quantity of waste in input. This is why, in the literature, the results have often been referred to 1 t (or 1 kg, or 1000 t) of MSW [16,29–35]. However, some studies with FU referring to the output(s) of WtE technologies are available as well: in this case, the more often selected functional units are 1 MWh (or 1 kWh) of electricity produced [36,37], 1 MJ of heat production [32], 1 MJ of energy (heat and electricity) produced [38], 1 kWh mixed combustible waste [39], and 1 MW_{HHV} (higher heating value basis) of bio-hydrogen [20].

Another relevant point is the system boundaries of the study, which identify the processes that are included or excluded from the analysis. Yang et al. [38] studied the impacts of biomass gasification, starting the assessment from the agricultural phase. This

is because, in this specific study, the biomass in input (made of a mix of rice husks and straw) was not a waste product but a co-product (having a market value) of rice production. On the contrary, in the high majority of WtE LCA studies, the material in input is a waste product, and, as a consequence, it has no memory of its previous life. It is the “zero burden assumption,” meaning that all activities occurring before the moment of waste generation are excluded from the system boundaries [40].

The method of accounting for biogenic carbon emissions is also a crucial choice: in some studies, biogenic CO₂ is considered as climate neutral (the GWP impact factor is 0) [16,31,32,38,39,41], while, in other cases, it is included in the calculation without any distinction between biogenic and fossil carbon dioxide (the GWP impact factor is 1) [20,34,42]. As suggested by some authors, the exclusion of biogenic carbon can result in an oversimplification when comparing disposal scenarios [42,43].

Finally, another factor influencing the results is the approach to consider the co-products (or by-products) from the process. Therefore, it is possible to (i) use an allocation approach: impacts are allocated to each co-product according to the physical, economic, or other criteria; (ii) use a system expansion approach, where the impacts are related to a main product, while the co-products generate an environmental credit due to the material/energy that they substitute. As discussed in the study of Burnley et al. [33], there is much debate and uncertainly regarding the process that should be considered as avoided.

In this study, a comparative LCA analysis was carried out on an incineration and a gasification plant. The following accounting rules were considered:

The FU is 1 kWh of electricity provided by both the WtE technologies;

The system boundaries follow the “zero burden assumption,” which means that waste enters in the system as burden free;

Biogenic carbon emissions were considered (the GWP impact factor was 1).

By fixing these rules, this study evaluated how different approaches for the accounting of co-products and the avoided landfill for MSW influence the impact assessment of electricity production.

The LCA models were realised with both the allocation and the system expansion approaches. In the first case, by-products were allocated on an economic basis. In line with previous literature [44], the price for electricity was assumed equal to EUR 0.1173 per kWh, and the fly ash and slag prices were set to EUR 0.024 per kg and EUR 0.0025 per kg, respectively. The price of metals was estimated to be EUR 2.903 per kg, an average of the prices of lead and zinc (<https://www.lme.com/> (accessed on 15 November 2021)).

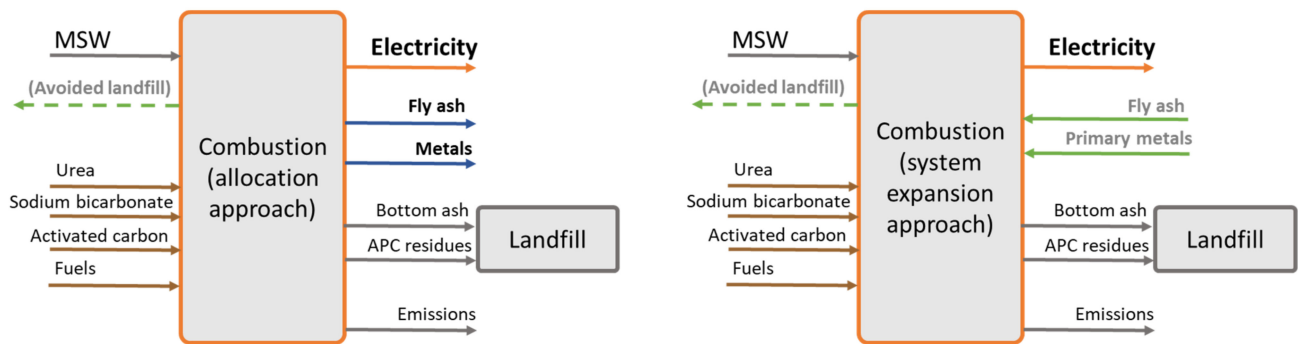
In the LCA model with system expansion approach, it is assumed that fly ash from the analysed WtE technologies substitutes the fly ash from hard coal furnace; the slag substitutes the gravel in construction activities [45]; and secondary lead and zinc substitute the same metals from virgin raw materials.

In addition, for each case, a further analysis was developed by including as well the environmental consequences of the waste management diversion: WtE technologies allow to avoid the impacts related to the landfill (in the next tables, it is indicated with the abbreviation AL, meaning avoided landfill).

Figures 1 and 2 graphically show the processes and flows included in the study.

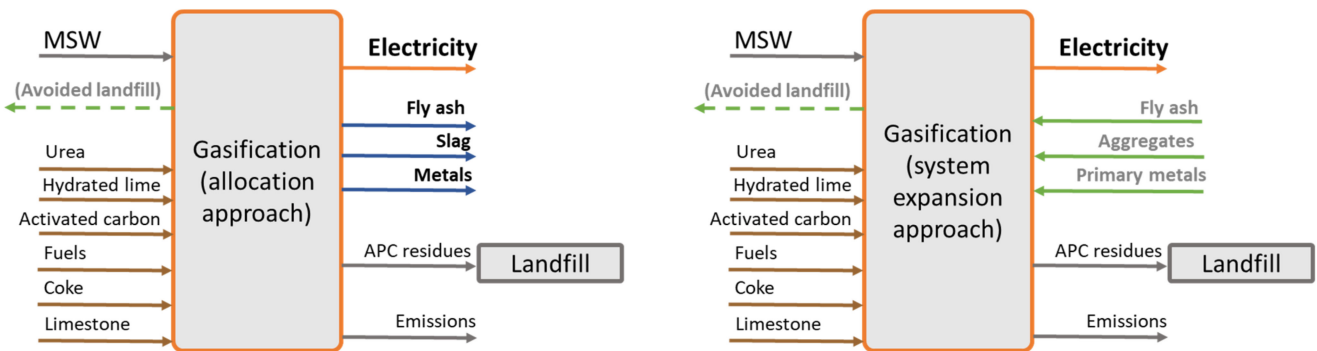
The inventory, detailed in Table 1, is mainly composed by primary data provided by Trattamento Rifiuti Metropolitani (TRM S.p.A.) of Turin (Italy) (for the incineration) and by Nippon Steel and Sumikin Engineering (NSENGI, JFE Engineering corp, Yokohama Kanagawa, Japan), supported, when necessary, by scientific-literature data [46,47].

Fly ash, slag, and metals (lead and zinc) are considered as by-products. Therefore, fly ash and slag are generally used for the production of concrete [48–54], while metals can be recycled. Outputs of bottom ash and APC residues are considered waste to be disposed in landfills.



Left: model with allocation approach. Right: model with system expansion approach. Legend of arrow's colours/type: grey = waste/emission; brown = input materials/energy; orange = main output product; blue = by-products; green = avoided products; green dashed = avoided waste treatment.

Figure 1. Processes and flows included in the LCA study for the impact assessment of electricity produced by a combustor.



Left: model with allocation approach. Right: model with system expansion approach. Legend of arrow's colours/type: grey = waste/emission; brown = input materials/energy; orange = main output product; blue = by-products; green = avoided products; green dashed = avoided waste treatment.

Figure 2. Processes and flows included in the LCA study for the impact assessment of electricity produced by a gasifier.

Table 1. Life cycle inventory of the production of electricity through the incineration and the gasification WtE technologies.

Flow (Unit of Measure)	Quantity	
	Incineration	Gasification
Outputs		
Electricity (kWh)	711.94	643.61
Bottom ash (kg)	219	0
Slag (kg)	0	145
Metals (kg)	17.8	16
Fly ash (kg)	17	28.6
APC residues (kg)	14.7	8.4
Emission into air: carbon dioxide, biogenic (kg)	577.5	577.5
Emission into air: carbon dioxide, fossil (kg)	350.7	496.5
Emission into air: carbon monoxide (g)	311.1	41.8
Emission into air: oxygen (kg)	638	311
Emission into air: nitrogen (kg)	5100	4080

Table 1. Cont.

Flow (Unit of Measure)	Quantity	
	Incineration	Gasification
Emission into air: nitrogen oxides, as NO ₂ (g)	559	73.6
Emission into air: ammonia (g)	30.7	0
Emission into air: sulphur dioxide (g)	62.2	11.6
Emission into air: VOC (g)	5.59	0
Emission into air: mercury (g)	0.311	0.07
Emission into air: cadmium (g)	0	0.006
Emission into air: lead (g)	0	0.24
Emission into air: zinc (g)	0	0.33
Emission into air: aluminium (g)	0	0.4
Emission into air: copper (g)	0	0.011
Emission into air: heavy metals, unspecified (g)	3.4221	0
Emission into air: hydrofluoric acid (g)	3.11	0.84
Emission into air: hydrochloric acid (g)	31.11	13
Emission into air: dust (g)	31.11	3.52
Emission into air: dioxins and furans (ng)	322	3.06
Inputs		
Municipal solid waste (t)	1	1
Urea (kg)	3	4.46
Hydrated lime (kg)	0	8.22
Sodium bicarbonate(kg)	17.8	0
Activated carbon (kg)	2.1	0.48
Auxiliary fuels (kg)	1.89	0.36
Coke (kg)	0	40
Limestone (kg)	0	50

3. Results

The impact assessment was calculated with the ReCiPe Midpoint (H) V1.13 method, using the SimaPro 9 software. In line with previous literature [9,30,36], the impact categories of the global-warming potential (GWP), the acidification potential (AP), the eutrophication potential (EP), the human-toxicity potential (HTP), the marine-ecotoxicity potential (MEP), the ozone-depletion potential (ODP), and the fossil-depletion potential (FDP) were analysed. Table 2 resumes the impact results for the production of 1 kWh of electricity through the combustion and gasification technologies, for the approaches previously detailed: (i) the allocation approach and (ii) the system expansion approach, both of them calculated including/excluding the avoided landfill (the abbreviation *AL* means that the benefits of the avoided landfill were included).

From these calculations, it clearly emerges that the chosen LCA approach had a strong influence on the impact results. For the sake of clarity, the numbers of Table 2 were employed to realise two graphs with relative results: Figures 3 and 4 show the relative impacts for the production of electricity through the combustor and the gasificator, respectively. As it can be noticed, except for the impact category of HTP, the results significantly changed according to the followed approach. As expected, when the avoidance of landfill for the MSW was included in the system boundaries, the final impact results were lower (in some cases, the result was negative, meaning that the production of electricity

from WtE technologies had higher benefits than impacts). The difference between allocation and system-expansion approaches was significant as well: for example, for the GWP impact category, the impact of electricity from both the WtE technologies had results that were 35% less if calculated with the allocation approach in comparison with the result obtained with the system-expansion approach. This trend cannot be extended to the other impact categories: for example, for most of the analysed impact categories, the impact of electricity from combustor results were lower if calculated with a system-expansion approach.

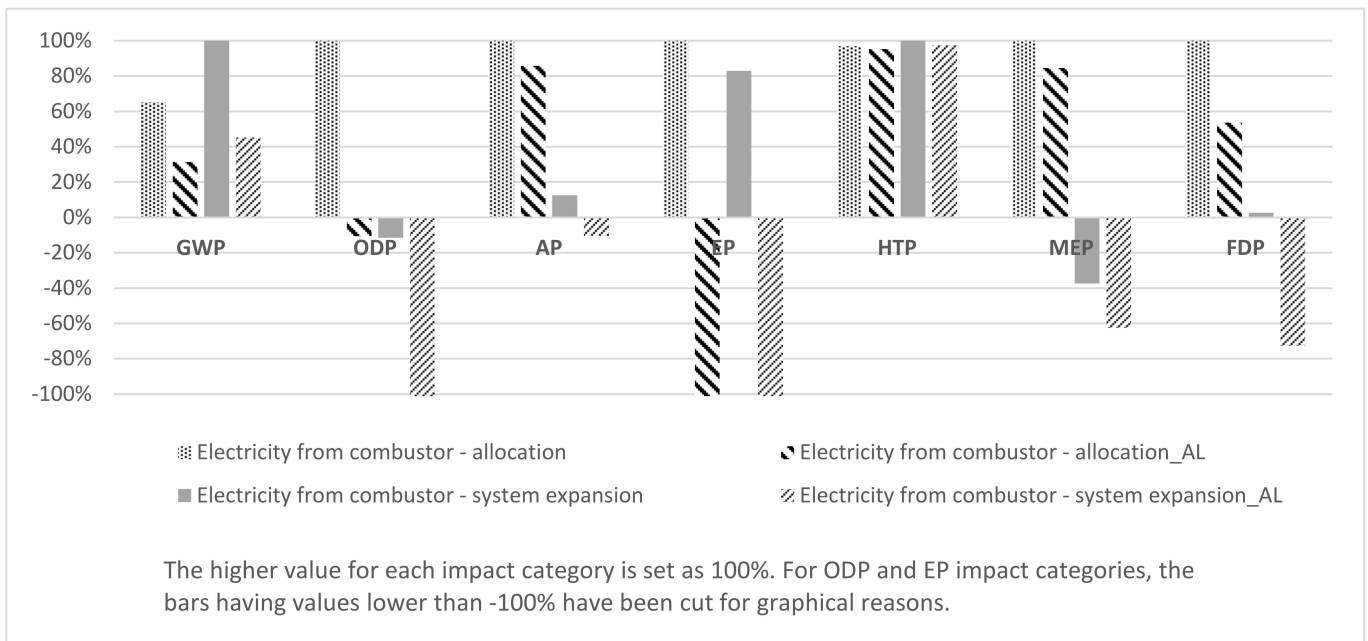


Figure 3. Relative impact results of the production of electricity from the combustor.

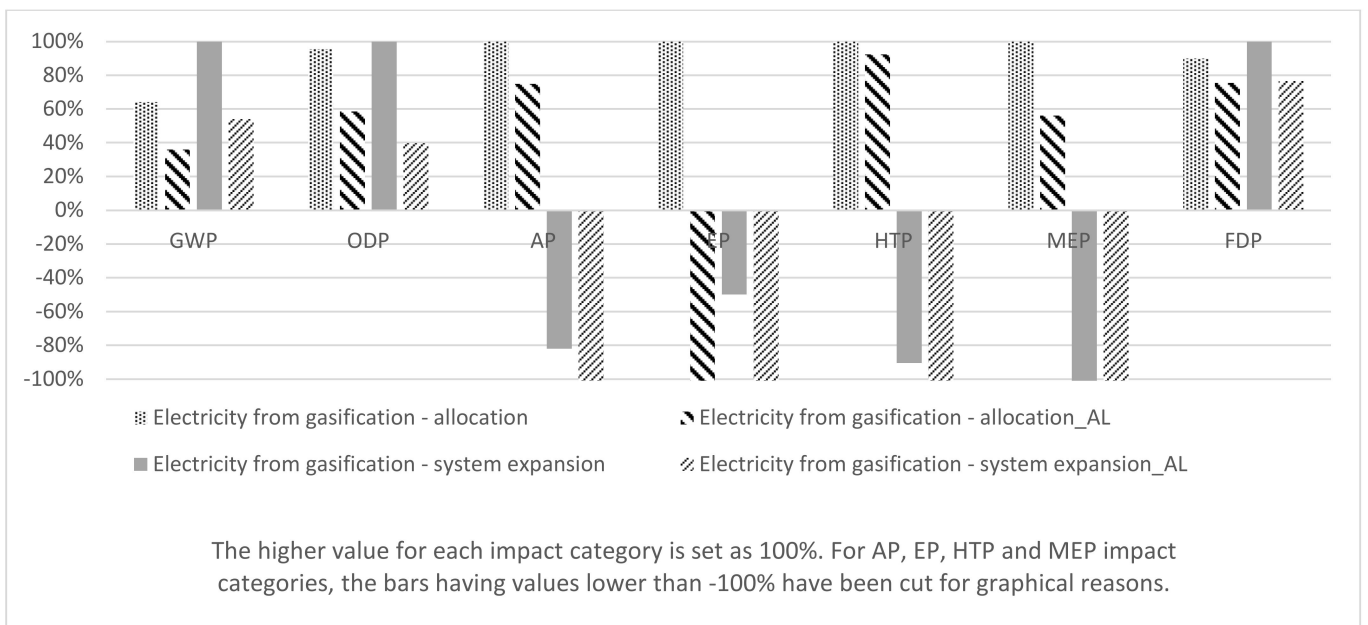


Figure 4. Relative impact results of the production of electricity from the gasifier.

Table 2. Impact assessment results for the production of 1 kWh of electricity with the incineration and gasification WtE technologies.

Impact Category	Unit	Electricity from Combustor—Allocation	Electricity from Combustor—Allocation_AL	Electricity from Combustor—System Expansion	Electricity from Combustor—System Expansion_AL	Electricity from Gasification—Allocation	Electricity from Gasification—Allocation_AL	Electricity from Gasification—System Expansion	Electricity from Gasification—System Expansion_AL
GWP	kg CO ₂ eq	8.37×10^{-1}	4.04×10^{-1}	1.28	5.81×10^{-1}	1.08	6.07×10^{-1}	1.69	9.12×10^{-1}
ODP	kg CFC-11 eq	2.58×10^{-9}	-2.73×10^{-10}	-2.97×10^{-10}	-4.92×10^{-9}	8.11×10^{-9}	4.97×10^{-9}	8.49×10^{-9}	3.38×10^{-9}
AP	kg SO ₂ eq	6.07×10^{-4}	5.21×10^{-4}	7.57×10^{-5}	-6.33×10^{-5}	3.74×10^{-4}	2.79×10^{-4}	-3.07×10^{-4}	-4.60×10^{-4}
EP	kg N eq	2.92×10^{-5}	-1.06×10^{-3}	2.42×10^{-5}	-1.75×10^{-3}	1.11×10^{-5}	-1.19×10^{-3}	-5.52×10^{-6}	-1.97×10^{-3}
HTP	kg 1,4-DB eq	2.02×10^{-1}	1.98×10^{-1}	2.08×10^{-1}	2.02×10^{-1}	4.69×10^{-2}	4.33×10^{-2}	-4.25×10^{-2}	-4.84×10^{-2}
MEP	kg 1,4-DB eq	4.50×10^{-4}	3.80×10^{-4}	-1.68×10^{-4}	-2.82×10^{-4}	1.75×10^{-4}	9.79×10^{-5}	-6.12×10^{-4}	-7.37×10^{-4}
FDP	kg oil eq	1.06×10^{-2}	5.66×10^{-3}	2.84×10^{-4}	-7.67×10^{-3}	3.36×10^{-2}	2.82×10^{-2}	3.74×10^{-2}	2.86×10^{-2}

Results are provided for the different approaches and system boundaries analysed (al-location/system expansion, inclusion/exclusion of the benefits due to the avoidance of landfill for MSW). Abbreviation AL = avoided landfill.

The graph in Figure 5 shows the relative impact comparison between the electricity generated through the combustor and through the gasifier, calculated with the allocation approach, excluding the benefits of the avoided landfill. As it can be noticed, for most of the analysed impact categories, the electricity from the gasification results were more sustainable than the electricity from the combustor, except for the GWP, ODP, and FDP impact categories (for example, for the GWP, the electricity from the combustor results had an impact of 23% less than the electricity from the gasifier).

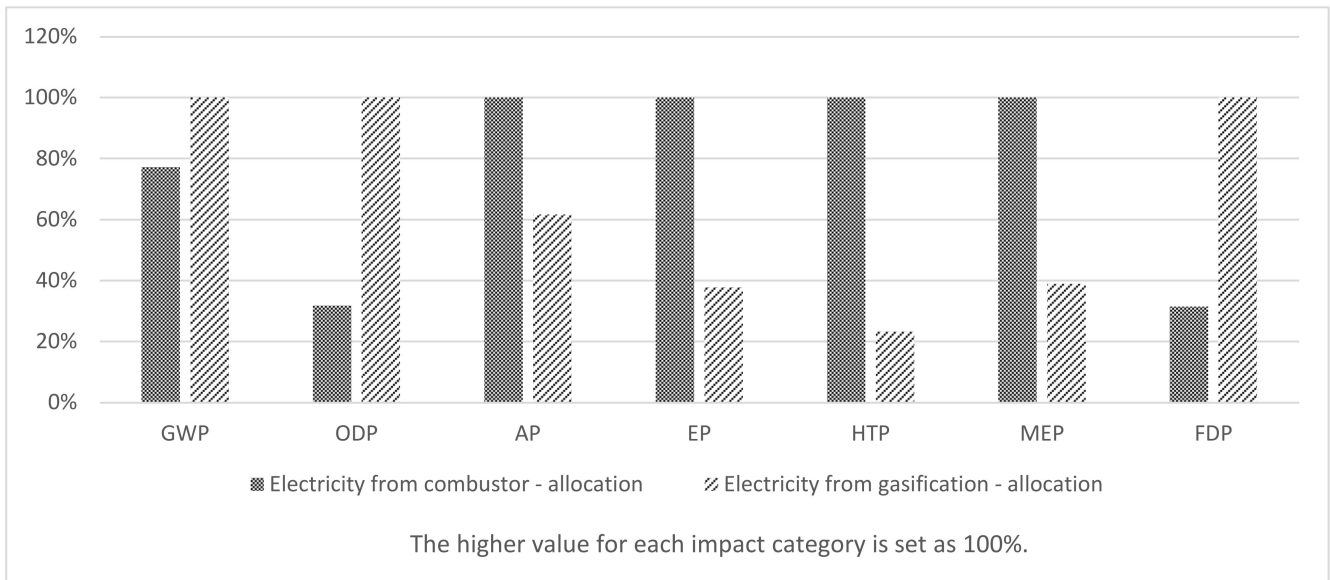


Figure 5. Relative comparison of impacts for the production of electricity from combustor and gasifier (calculated with allocation approach, without considering credits due to the avoided landfill for MSW).

From a contribution analysis, it can be deduced that for both the analysed WtE technologies, the direct emissions at the plant are generally responsible for a significant share of the total impact (except for the categories of ODP and FDP, which are not influenced by the direct emissions at the combustor/gasificator). Figures 6 and 7 show, for each analysed impact category, the contribution of direct emissions over the total impact for the production of electricity.

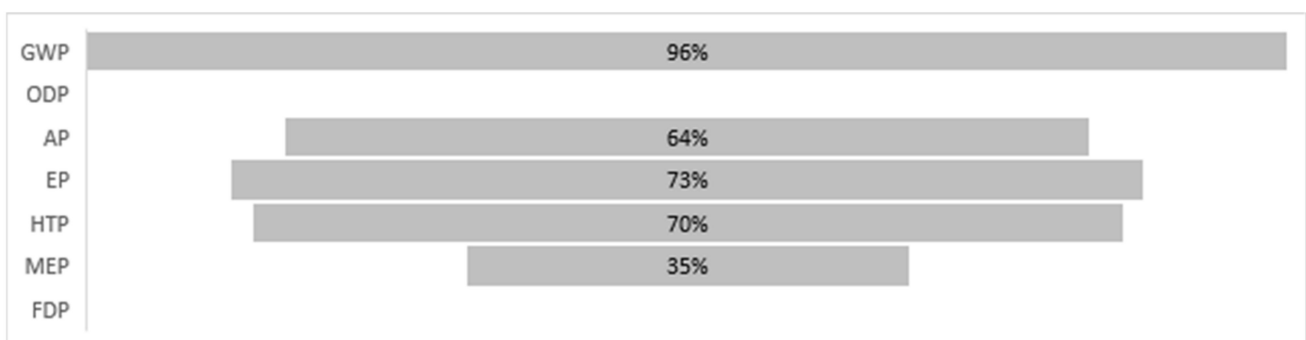


Figure 6. Contribution of direct emissions from the combustor over the total impact for the production of electricity.

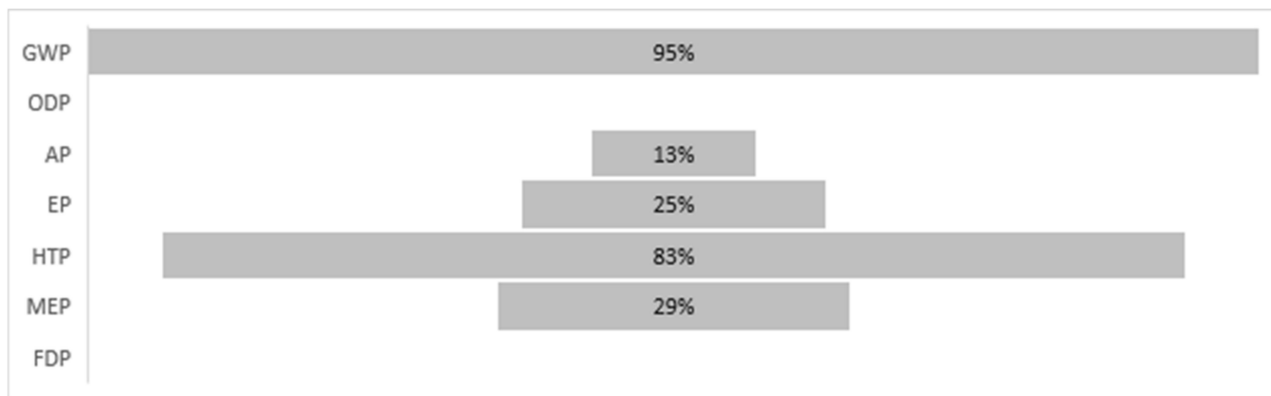


Figure 7. Contribution of direct emissions from the gasification over the total impact for the production of electricity.

4. Discussion and Conclusions

This study provided inventory data for the development of LCA analyses on the electricity production from two WtE technologies: incineration and gasification. Starting from this inventory, LCA models were developed with different system boundaries (including/excluding the benefits due to the avoided landfill for MSW) and different approaches for the accounting of by-products (allocation/system expansion). It resulted that, according to the chosen method of modelling, the impact values significantly change, and even the comparison between the two technologies can lead, for some impact categories, to opposite conclusions. Moreover, as detailed in Section 2, other accounting rules chosen by the LCA practitioner (the functional unit, the method of accounting for biogenic carbon emissions, and the eventual burden given to the waste in input) add variability to the final impact results. As a consequence, especially with processes connected to the valorisation of waste, the LCA analysis has to be employed with particular attention to the consistency between the goal of the study and the fixed accounting rules. Additionally, the use or integration of results from LCA literature studies have to be used with caution, checking beyond the quality of the inventory and the settings of the study.

These limitations of LCA studies have been raised also in other application fields [55–60]. To harmonise and allow a better comparability of similar product/services, product category rules have been developed, for example, in the context of product environmental footprint (PEF) [61] and on the environmental product declarations (EPDs) [62]. However, to the authors' knowledge, no category rules have been developed until now for WtE technologies. As a consequence, the development/use/reproduction/comparison of studies focused on the valorisation of waste should be carried out with caution.

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