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# Environmental assessment of existing and alternative options for management of municipal solid waste in Brazil

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ABSTRACT

Life cycle assessment (LCA) was used to evaluate and compare three different categories of management systems for municipal solid waste (MSW) in Brazil: (1) mixed waste direct disposal systems, (2) separate collection systems, based on wet-dry streams, and (3) mixed waste mechanical-biological systems, including materials recovery. System scenarios were built around main treatment techniques available and applicable in developing countries, and considered barriers as well as potential synergies between waste management and other industrial production. In the first category systems, we measured the impact magnitude of improper disposal sites (semi-controlled and controlled dumps) still used for approximately 40% of collected MSW, and found that sanitary landfills could decrease it 3–5 fold (e.g. GWP, from 1100–1200 to 250–450 kg CO2 eq. $t^{-1}$ waste). As an alternative, waste incineration did not show significant benefits over sanitary landfilling, due to limitations in energy utilization and the low-carbon background electricity system. Category two of systems, revealed recycling benefits and the necessity as well as potential risks of biological treatment for wet streams. Simple wet-dry collection could result in relatively high levels of contamination in compost outputs, which should be mitigated by intensive pre- and post-treatment. Potential impact of air emissions from biological degradation processes was important even after anaerobic digestion processes. Biogas upgrading and use as vehicle fuel resulted in bigger savings compared to direct electricity production. Lastly, category three, mechanical-biological systems, displayed savings in most environmental impact categories, associated with materials recovery for recycling and refuse-derived fuel (RDF) production and utilization in cement manufacturing.



## 1. Introduction

Historic and current improper waste management in Brazil continues to cause surface and groundwater contamination, contributes to climate change, air quality decay, among other environmental and human health impacts (Rosa et al., 2017; Schalch et al., 2002). Furthermore, according to some projections, generation of municipal solid waste (MSW) in Brazil is likely to increase dramatically in the near-future, in connection with rapid urbanization and economic development (Veloso, 2014).

According to the annual panorama published by the Brazilian Association of Public Cleaning and Special Waste Companies (ABRELPE), the current Brazilian MSW generation is in the order of 78.3 million tons per year (ABRELPE, 2017). Collection coverage reaches approximately 91% of the total waste generated and waste that is not collected is likely either dumped illegally or burned in public open spaces (Alfaia et al., 2017). Brazilian waste management should follow the requirements of the Nacional Policy of Solid Waste (PNRS – Federal Law 12,305/2010): the prohibition of inadequate waste disposal and the proposed hierarchy (avoid generation, reduction, reuse, recycling, treatment and disposal) (Brasil, 2010). Nevertheless, in 2016, 17.4% of the collected mixed MSW was still disposed in semi-controlled dumps (i.e. lixão – in Portuguese) which have no engineering measures (no leachate or gas management), representing only a designated open location for disposal (ABRELPE, 2017). A further 25.2% was placed in controlled dumps (i.e. aterro controlado – in Portuguese), with basic

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engineering measures such as compaction and (daily, intermediate or final) cover. Finally, 58.4% was adequately disposed of in sanitary landfills (i.e. aterro sanitário – in Portuguese) with all proper engineering measures (Hoornweg and Bhada-Tata, 2012).

Only 1.9% of the Brazilian municipalities have composting plants, and as for incineration, so far it has only been used for hazardous waste, such as from health care (ABRELPE, 2017; SNIS, 2017). About 30% of the municipalities have selective collection initiatives, however only 3.6% of the produced waste is actually reported as separately collected. The informal sector, i.e. waste pickers, plays a significant role in separate collection, being responsible for as much as 90% of the recyclables collection in the country (Aquino et al., 2009; MMA, 2012).

### 1.1. Evaluation of MSW management strategies in Brazil

Life Cycle Assessment (LCA) is an internationally standardized method and widely used tool in the support of decision-making (European Commission, 2010). With regard to environmental impact of waste management, from a decision-making perspective, Brazil constitutes a very interesting case study. Unlike many other developing countries, Brazil's electricity production mix is predominantly renewable (dominated by hydropower), which limits possible environmental benefits of energy-from-waste strategies. Moreover, due to a ban instated in the 1970s on diesel passenger cars and commercial vehicles with capacity inferior to 1000 kg, today the Brazilian light vehicle fleet is made up almost entirely by the so-called flexible-fuel vehicles running on a mandatory blend of anhydrous ethanol and gasoline (ethanol share reaching 27%,by volume, in 2015) (Dallmann and Façanha, 2015). This limits to some extent possible utilization of upgraded landfill gas and biogas from anaerobic digestion as vehicle fuel.

Considering the magnitude and complexity of the problem, there are few LCA studies addressing MSW in Brazil. Of the studies available, almost all employ an attributional LCA framework, where allocation is avoided by system expansion in order to credit management systems in the case of energy and materials recovery. Most studies can also be categorized based on the assessment scope, involving: (1) theoretical scenarios for mixed waste treatment, (2) theoretical scenarios including separate collection, and (3) evolution of management in a specific area over time. Studies that assessed theoretical treatment scenarios for mixed waste were mostly concerned with the potential of energy-from-waste. Mendes et al. (2004) and Leme et al. (2012, 2014) compared scenarios based on mixed MSW landfilling (with and without energy recovery) and incineration (Waste-to-Energy - WtE) for the cities of São Paulo and Betim (Belo Horizonte), respectively. They found that in general incineration showed a lower environmental impact than landfilling. Nevertheless, energy recovery did not achieve high savings, considering the low impact of the Brazilian electricity mix. Leme et al. (2014) also determined by a techno-economic analysis that incineration plants face serious economic barriers in Brazil, and it would require that municipal authorities dispose of much higher budgets for waste management.

Among studies addressing theoretical scenarios including separate collection, the work by Reichert and Mendes (2014) stands out. The authors applied LCA methodology as well as economic and social analysis, to compare eight management scenarios (including a reference with approximately 9% recycling) for the city of Porto Alegre. Alternative scenarios included separate collection of dry recyclables and organics in various degrees combined with different approaches to mixed waste treatment, including incineration and mechanical-biological treatment (MBT) based systems (aerobic, anaerobic and with refuse derived fuel (RDF) production). Scenarios with high recycling and full treatment of remaining mixed waste by MBT-based systems performed better in most environmental impact categories, while the scenario based on high recycling was most preferable regarding economic and social effects. Another study by Goulart Coelho and Lange (2016) compared theoretical scenarios that achieved the PNRS targets for the Brazilian southeast (case of Rio de Janeiro), i.e. reduce the recyclables and organic waste sent to landfill to 50% and 55%, respectively. Three scenarios focused on mixed waste treatment, such as incineration and MBT (with ferrous metals recovery and RDF for cement production), while four other scenarios assumed that diversion happened mostly by separate collection. The scenarios based on high separate collection displayed also the highest environmental benefits. Bernstad Saraiva et al., (2017) addressed organic waste in Rio de Janeiro and determined that similar environmental performance could be achieved if biowaste would be separated at the source or by mechanical means in MBT facilities with anaerobic digestion. Most importantly, this work also aimed at showing the influence between choosing an attributional versus a consequential LCA modelling framework. This was demonstrated as very important in a Brazilian decision-making context, due to the specific energy system. Finally, the recent study of Ibáñez-Forés et al. (2017) reports the evolution of MSW management and its related environmental impact between 2005 and 2015, in the city of João Pessoa (North-east Brazil). The city implemented separate collection of recyclables covering approx. 20% of districts. It is possible to determine that in 2015 the covered areas reached a combined recycling rate of 7% (6% from separate collection, 1% by mixed waste materials recovery facility - MRF), while 93% of waste was directed to a sanitary landfill. Despite the low recycling performance, the study showed that environmental impacts decreased over time, recycling contributing savings in several impact categories.

### 1.2. Study objectives

Governments and local authorities in developing countries often aim to emulate successful waste management systems in developed (industrialized, high-income) countries, through initiatives (and legislation) typically focused on technology issues, forgetting socio-economic, cultural and governance aspects, which almost as often results in implementation failures (Campos, 2014; Wilson et al., 2013). Most scientific evaluations of waste management follow the same line shown also for Brazil, with studies targeting treatment or theoretical separate collection scenarios. Successful systems in developed countries incur enormously high costs compared to budgets spent in developing countries (Alfaia et al., 2017; Wilson et al., 2013). However, in the former, these high costs are usually and entirely, covered by household paid waste fees, a situation which is still far from implementation in the latter (at present).

Beyond the urgent enforcement of safe and controlled disposal in Brazil, possible solutions towards wide-spread management of MSW with the aim of resource recovery and recycling have to take offset in local conditions and should apply options that capitalise on possible synergies with other industry sectors. Such solutions could include the implementation of: (1) simple and intuitive source separation, such as into dry and wet streams, where it is feasible, and (2) bypass public participation by wide implementation of MBT or mixed waste MRFs, using concepts that combine dry recyclables recovery, RDF production and the separation and treatment of biodegradable waste. The latter can be realized with technical solutions ranging from very basic to advanced (Cimpan et al., 2015; Münnich et al., 2006). Because no MSW or RDF dedicated WtE facilities exist in Brazil, production of high quality RDF could be prioritized with the objective to substitute fossil fuels in the cement industry. RDF utilization in the cement industry has been shown superior when compared to WtE that produces only power and when background marginal electricity is not carbon intensive (Cimpan and Wenzel, 2013). According to IFC (2017) the alternative fuels co-processing or substitution rate in Brazil was only 8.1% in 2014, while in Europe this was 41%, with high variation between countries (highest 65% in Germany) (de Beer et al., 2017).

The primary objective of the present study was to evaluate and compare from an environmental impact perspective, different system scenarios built around main technological options for the management of MSW in Brazil. System scenarios considered specific conditions, barriers and sector synergies mentioned above, as well as more theoretical situations with implementation of costly and state-of-the-art options (e.g. WtE). The goal of the study is to inform and support decision making towards policy development and strategy planning concerning MSW management in Brazil.

## 2. Materials and methods

### 2.1. LCA methodology

Considering the goal of this study and that MSW management changes can have potentially large effects on other technological and societal systems, the general methodological framework was based on consequential LCA (European Commission, 2011; 2010). This implies system expansion in the case of multi-functionality and when a change in waste management influences background systems (e.g. substitution of energy in the energy system). Interactions with adjoining systems were modelled (where possible) by use the marginal LCI data (as opposed to average data), which denotes processes and technologies most likely to respond due to market mechanisms (i.e. supply-demand changes for goods/services). The functional unit (FU) was the management (i.e. from generation to final disposal/sinks) of 1 t (t = metric tonne) of MSW. The reference flow MSW should be understood as daily-generated household waste, street sweepings and similar waste from small business, service and institutions.

The modelling was performed in EASETECH, a software developed in Denmark specifically for waste management LCA (Clavreul et al., 2014). This software allows detailed mass and substance flow modelling of waste management chains. Life cycle impact assessment (LCIA) was performed with the ILCD recommended method, and included 12 impact categories (listed in Table 1). Normalization factors for emissions and resource extraction, geographically representative as global, were based on DTU (2016) and Sala et al. (2017).

Biogenic $CO_2$ originating from the waste was considered to be climate neutral, while biogenic carbon that was not emitted after 100 years was considered stored (and accounted as an avoided impact) according to the method in Christensen et al. (2009). Nevertheless, due to mostly warm and wet climate conditions characterizing Brazil, carbon storage was deemed insignificant with the application on soil of compost and digestate, and in the cases of semi-controlled and controlled dumps, in accordance with a previous study by Bernstad Saraiva et al. (2017).

#### 2.1.1. Temporal, geographical and technological scope

The results of this assessment can be considered valid short-to-medium term, i.e. 5 to 10 years. Inventory data for foreground systems refer to current treatment technologies and substantial technological changes are not expected within the period. Technology performance was based on the data from different published research sources and the EU Best Available Techniques for the Waste Treatment Industries (BREF). The geographical scope refers to Brazil, nevertheless, the origin of many foreground processes was Europe, adapted to average Brazilian climate conditions, while the origin of some background processes was European or Global averages (e.g. primary materials and fuels production).

#### 2.1.2. System boundaries

The systems in this evaluation should be understood as the sum of a foreground system and background system, using the definitions of Clift et al. (2000) and European Commission (2010). In the analysis of waste management systems, the foreground system comprises all waste management activities from waste generation, through treatment and recovery of materials and/or energy, to the point where these functional outputs are exchanged with the background systems (the background economy and markets). The background systems represent the economic activities (e.g. energy production, material production) which exchange materials and energy (including the functional outputs from waste management) with the foreground system and thus affect the decisions taken regarding foreground systems.

### 2.2. Description of alternative systems (foreground scenarios)

Table 2 shows the foreground system scenarios and variations evaluated in this work. Category 2 systems are based on a theoretical (but plausible) separate collection efficiency of 20% (for dry streams), whereas the rest is considered as a wet stream. The focus was to highlight the effects of different biological treatment, rather than source separation, which was handled in a generic way.

### 2.3. Life cycle inventory (LCI)

#### 2.3.1. Municipal solid waste (MSW) generation

An average Brazilian waste composition was established after compiling data from a large number of studies representing the different country regions. The composition was first calculated as a weighted average (based on population) of 15 studies (Colvero et al., 2016). The data sources mostly consisted of gravimetric analyses performed on municipal waste sampled at the source of

**Table 1**
Normalization factors ILCD recommended.

| LCD Impact Category | Abbreviation | Unit | Normalization factor |
|---|---|---|---|
| Climate change (GWP) | GWP100 | kg $CO_2$ eq. $PE^{-1}$ $year^{-1}$ | 8400 |
| Ozone depletion | ODP | kg CFC-11 eq. $PE^{-1}$ $year^{-1}$ | 0.0234 |
| Human toxicity, cancer effects | HT, CE | CTUh $PE^{-1}$ $year^{-1}$ | 3.85E−05 |
| Human toxicity, non-cancer effects | HT, non CE | CTUh $PE^{-1}$ $year^{-1}$ | 4.75E−04 |
| Particulate matter | PT | kg PM2.5 eq. $PE^{-1}$ $year^{-1}$ | 5.07 |
| Photochemical ozone formation | POF | kg NMVOC eq. $PE^{-1}$ $year^{-1}$ | 40.6 |
| Terrestrial Acidification | TAD | mol H + eq. $PE^{-1}$ $year^{-1}$ | 55.5 |
| Eutrophication terrestrial | EPT | mol N eq. $PE^{-1}$ $year^{-1}$ | 177 |
| Eutrophication freshwater | EPF | kg P eq. $PE^{-1}$ $year^{-1}$ | 0.734 |
| Eutrophication marine | EPM | kg N eq. | 28.3 |
| Ecotoxicity freshwater | ECF | CTUe | 11,800 |
| Depletion of abiotic resources, mineral, fossils and renewables | DAMR | kg Sb eq. | 0.193 |

**Table 2**
Summary table for the foreground scenarios.

| Main system category | System scenario | System scenario variation |
|---|---|---|
| 1. Mixed waste direct disposal systems | 1.a - Semi-controlled dumps<br>1.b - Controlled dumps<br>1.c - Sanitary or fully controlled landfilling without landfill gas valorisation<br>1.d - Sanitary or fully controlled landfilling with landfill gas valorisation<br>1.e- Incineration WtE by means of moving grate combustion | |
| 2. Separate collection systems – source separation into wet and dry streams (20%:80%) | 2.a - Dry stream sorted in a simple MRF and wet stream sanitary landfilling<br>2.b - Dry stream sorted in an advanced MRF and wet stream sanitary landfilling<br>2.c - Dry stream sorting and wet stream composting<br>2.d - Dry stream sorting and wet stream dry digestion, biogas to electricity production<br>2.e - Dry stream sorting and wet stream pre-treatment and wet digestion, biogas to electricity production | 2.c(w) open air composting<br>2.c(e) enclosed composting<br>2.d(u), 2.e(u) biogas upgraded and used as vehicle fuel |
| 3. Mixed waste mechanical–biological and sorting systems | 3.a - Simple Aerobic MBT<br>3.b - Advanced Anaerobic-aerobic MBT (incl. material recovery)<br>3.c - Simple Biological drying MBT<br>3.d - Advanced Biological drying MBT (incl. material recovery) | 3.b(u) biogas upgraded and used as vehicle fuel |

**Table 3**
Waste composition for Brazil.

| Waste fraction | Generated before informal sector (kg) | Generated before informal sector (%) | FU after informal sector (kg) | FU after informal sector (%) |
|---|---|---|---|---|
| Paper | 75.8 | 7.31 | 60.1 | 6.01 |
| Cardboard | 69.4 | 6.69 | 67.9 | 6.79 |
| Beverage cartons | 3.4 | 0.33 | 2.7 | 0.27 |
| Metals | 18.2 | 1.75 | 11.4 | 1.14 |
| Glass | 25.3 | 2.44 | 22.7 | 2.27 |
| Plastics | 185.4 | 17.87 | 175.3 | 17.53 |
| Organic | 548.5 | 52.88 | 548.5 | 54.85 |
| Other combustibles | 49.9 | 4.81 | 49.9 | 4.99 |
| Othernon-combustibles | 60.0 | 5.78 | 60.0 | 6.00 |
| Hazardous | 1.5 | 0.14 | 1.5 | 0.15 |
| Total | 1037 | 100 | 1000 | 100 |

generation (households) before intervention from the informal sector. According to SNSA (2016) the informal sector is accounted in official sources as capturing 3.6% of generated waste (consisting mostly of dry recyclable materials). In this study, we assumed that in all the systems modelled, the intervention from the informal sector remains constant. Therefore, the initial composition was adjusted to represent the waste after removal of 3.6% materials. The composition before and after (the latter representing the FU of this work) is presented summarized in Table 3. Details can be found in the Supplementary material (SM).

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.wasman.2018.07.007.

*2.3.2. LCI for foreground system processes*

*2.3.2.1. Collection and transport.* Waste collection accounted for route collection and transport to the first handling facility. Collection was modelled considering a regular (rear-loading) truck and different diesel consumption (in litres of diesel per tonne of collected waste ($L\,t^{-1}$). Diesel consumption was set to $3.0\,L\,t^{-1}$ for mixed and wet stream collections, while for dry stream collection it was $6.0\,L\,t^{-1}$. The latter considered the potentially higher dispersion of collection points and lower truck capacity due to low bulk density. Long-distance transportation was largely based on Bassi et al. (2017) and Vergara et al. (2016) and further considering that the MRFs and MBTs would be located near to landfills (see Table 4).

*2.3.2.2. Source separation and material recovery facilities (MRFs).* Source separation programmes are slowly expanding in Brazil. Where implemented, the model is based on separation into dry-wet streams, which should be convenient and easy to follow for citizens. The dry stream is a mixture of different recyclable materials and miss-sorted non-recyclables (contamination). The materials fraction composition was based on the report from Prefeitura Municipal de Campo Grande (2017). The dry stream has to undergo sorting, which can happen in various conditions. We modelled two contrasting cases: (1) a simple MRF, reflecting small scale, low technology plants (mainly manual sorting) which are common in Brazil, and (2) an advanced MRF, reflecting more the state-of-the-art in Europe and the US, characterized by larger scale and mechanical sorting complemented with manual sorting. Consumption of electricity (15 and 20 $kWh\ t^{-1}$, respectively), diesel ($0.7\ L\ t^{-1}$) and steel wire for bales ($0.85\ kg\ t^{-1}$) was estimated considering previous work by Cimpan et al. (2016, 2015). Sorting efficiencies in the two plants are presented in the SM.

*2.3.2.3. Landfilling.* In EASETECH, landfilling is modelled with specialized modules that can be combined and adapted by changing a variety of parameters in order to reflect different types of landfills running in different climatic conditions. Brazil has regional climatic differences, but in this work it was approximated to a tropical humid and wet climate, considering average annual temperatures above 20 °C with average precipitation greater than $1000\ mm\ year^{-1}$ (ABRELPE, 2013). Climate conditions influence

**Table 4**
Collection and transportation vehicles, travelled distances and fuel consumptions.

| Collection and/or waste type | Type of vehicle | Distances (km) | Fuel consumptions ($L\ t^{-1}$) |
|---|---|---|---|
| Mixed waste collection | Collection truck 10 t | – | 3.0 |
| Dry stream collection | Collection truck 10 t | – | 6.0 |
| Wet stream collection | Collection truck 10 t | – | 3.0 |
| Ferrous and non-ferrous metal to recycling | Long haul truck 25 t | 350 | 0.03·distance |
| Glass to recycling | Long haul truck 25 t | 200 | 0.03·distance |
| Paper and cardboard to recycling | Long haul truck 25 t | 400 | 0.03·distance |
| PET, HDPE and LDPE to recycling | Long haul truck 25 t | 350 | 0.03·distance |
| RDF to cement kilns | Long haul truck 25 t | 400 | 0.03·distance |
| Residue streams (sorting, ash) to landfill | Collection truck 10 t | 5 | 0.06·distance |

**Table 5**
Landfill parameters used in EASETECH.

| Technology Description | | Units | Semi-controlled dump | Controlled dump | Sanitary - flare | Sanitary - energy |
|---|---|---|---|---|---|---|
| | | | No top cover, no gas and leachate collection | Top cover, no gas and leachate collection | Top cover, gas and leachate collection | Top cover, gas and leachate collection |
| Construction and Operation | Diesel consumption | $L\ t^{-1}$ waste | 2.02E−04 | 2.02E−04 | 2.02E−04 | 2.02E−04 |
| | Electricity consumption | $kWh\ t^{-1}$ waste | None | None | 8.00E−03 | 8.00E−03 |
| Landfill Gas Generation | Correction factor for decay rate | | 0.4 | 0.8 | 1.0 | 1.0 |
| LFG - Gas Collected | Year 0–5 | % of generated | 0 | 30* | 45 | 45 |
| | Year 5–15 | % of generated | 0 | 45* | 80 | 80 |
| | Year 15–55 | % of generated | 0 | 55* | 95 | 95 |
| | Year 55–100 | % of generated | 0 | 0* | 0 | 0 |
| LFG - Treatment | No treatment | % of collected | 100 | 100 | 0 | 0 |
| | Fugitive emissions | % of collected | 0 | 0 | 2 | 2 |
| | Flare or gas motor | % of collected | 0 | 0 | 98 | 98 |
| LFG – Top cover | Oxidation | % $CH_4$ | 0 | 18 | 36 | 36 |
| Leachate Generation | Net Infiltration | $mm\ yr^{-1}$ | 1000 | 900 | 650 | 650 |
| Leachate Collection | Year 0–80 | % of generated | 0 | 0 | 99.9 | 99.9 |
| | Year 80–100 | % of generated | 0 | 0 | 95 | 95 |
| Leachate Treatment | Type treatment | | None | None | POTW | WWTP |
| Storage of carbon | | % remaining C-biogenic | 0 | 0 | 100 | 100 |

* In the case of controlled dump these percentages denote gas that bypasses the top cover and is released to air unaffected.

the decay rate of (biodegradable) waste materials and thus gas (and methane) generation (Olesen and Damgaard, 2014). 1st order decay rates (k) for methane generation were changed to reflect Brazilian climate conditions. Different types of landfilling practices alter the k values and the generation of leachate. Thus, a methane correction factor (MCF) was used for each of the three landfill types (semi controlled and controlled dump, and sanitary landfill), based on ABRELPE (2013). Regarding the leachate generation, it was considered a 10 m height for the layers for all landfills, a waste density of 1 tonne $m^{-3}$ and 100 years as time horizon (Lagerkvist et al., 2011; Manfredi and Christensen, 2009; Olesen and Damgaard, 2014). The main parameters used are presented in Table 5 and further details are given in the SM.

*2.3.2.4. Waste-to-Energy (WtE).* or waste incineration was considered as a landfill alternative in the first category of systems (system 1.e). The process was modelled as state-of-the-art grate incineration, with data from Danish facilities (Møller et al., 2013). The plant efficiency was set to a net of 25% for electricity generation, meaning 25% of the thermal energy contained by the waste input (based on lower heating value) and after self-consumption is accounted. Energy content and GHG emissions consider the chemical characteristics of material fractions, based on the model library (Riber et al., 2009). Considering the lack of infrastructure and need for district heating in Brazil, no heat recovery was assumed. Bottom ash and fly ash were assumed sent to an inert landfill and recovered iron sent to recycling. The model in EASETECH considers wet flue gas cleaning and sorting of bottom ash; SNCR for removal of $NO_x$ and activated carbon to remove dioxins and Hg.

*2.3.2.5. Biological treatment.* The wet stream collected after source separation, in the second category of systems evaluated in this study, is still highly contaminated with other materials (30–40% is not biowaste). Before biological treatment, the stream has to undergo at least a simple pre-treatment to concentrate the biodegradable fractions. This was modelled as basic bag opening (coarse shredding) and screening (trommel screen). The process sequence for biological treatment is described in Table 6, while a summary of the consumption and emissions parameters used for all the biological treatments are shown in Table 7.

*Composting:* Composting processes were based on datasets available in the EASETECH database, which were adjusted to reflect Brazilian conditions. Open windrows composting mass balance, process inputs and emissions were based on Andersen et al. (2010), whereas enclosed/channel composting was based on data from facilities in Italy from EASETECH.

*Dry digestion:* Dry or high-solids digestion is anaerobic digestion performed with waste having total solids (TS) content between 20% and 50%. Existing technologies are well suited for heterogeneous waste streams and do not require intensive pre-treatment. The process modelled in this study uses gas-proof box-shaped reactors, operated in batch mode at mesophilic temperatures (details in the SM).

*Wet digestion:* Wet digestion systems operate with TS content less than 15% and typically utilize continuous stirred tank reactors (CSTR), whereby continuous mixing is ensured by mechanical means and/or biogas injection. The process require a homogenization of the substrate to low particle size, removal of contaminants and addition of moisture to a level that the substrate is pumpable.

**Table 6**
Technology description of biological treatment used in the study.

| Technology descriptions | Composting - Open | Composting - Enclosed | Dry anaerobic digestion | Wet anaerobic digestion | Simple aerobic MBT | Advanced anaerobic–aerobic MBT | Simple biological drying MBT | Advanced biological drying MBT |
|---|---|---|---|---|---|---|---|---|
| System scenario | | 2.c(e) | 2.d | 2.e | 3.a | 3.b | 3.c | 3.d |
| Input waste flow | | Wet stream | Wet stream | Wet stream | Mixed MSW | Mixed MSW | Mixed MSW | Mixed MSW |
| Pre-treatment/Mechanical sorting | Bag opening and screening | Bag opening and screening | Bag opening and screening | Bag opening and screening, pulping | Simple pre-conditioning, Fe metals separation | Complex pre-conditioning, sorting of recyclables | Simple pre-conditioning, Fe metals separation | Complex pre-conditioning, sorting of recyclables |
| Main biological treatment | Open windrows composting | Enclosed windrows/channel composting | Dry AD (batch, mesophilic) | Wet AD (continuous, mesophilic), Digestate liquid-solid separation | Enclosed windrows/channel composting | Dry AD (batch, mesophilic) | Biological drying in liquid-tight box reactors | Biological drying in liquid-tight box reactors, automatic handling |
| Curing/stabilization | Included in main treatment | Included in main treatment | Open windrows | Open windrows (digestate solid fraction) | Included in main treatment | Enclosed windrows/channel composting | No | No |
| Post-treatment | Screening | Screening | Screening | No | Screening | Screening | Densimetric separation of inerts | Densimetric separation of inerts |
| Air treatment | No | Acid scrubber, biofilter | No | No | Dedusting, acid scrubber, biofilter | Dedusting, acid scrubber, biofilter | Dedusting, acid scrubber, biofilter | Dedusting, acid scrubber, RTO |
| Compost (like) application | Agriculture (clay soil) | Agriculture (clay soil) | Agriculture (clay soil) | Agriculture (clay soil) | Land reclamation and landfill cover | Land reclamation and landfill cover | No (inert residue is landfilled) | No (inert residue is landfilled) |

Therefore, an additional pre-treatment by pulping was modelled, which is utilized in many biowaste AD facilities in Europe. The process has, three main steps: shredding, pulping and separation (screening) (Naroznova et al., 2016). Additional grit removal, floating material removal, and dewatering processes can be used to improve the final quality of the biopulp (organic slurry). Details of the pre-treatment are described in the SM.

*Emissions from biological treatment:* Air emissions (especially GHGs) can vary considerably and are dependent on a variety of factors including the matrix of the waste processed, type of technology (open vs. encapsulated) and applied air treatment techniques. A variety of sources were consulted in order to establish a baseline for air emissions in this study, including (among many more) the BREF Waste Treatment Industries (European Commission, 2006); benchmark emissions is UK facilities (DEFRA, 2011); experiments (Germany) and literature (Amlinger et al., 2008); German MBT facilities (Fricke et al., 2005); and Spanish composting and AD facilities (Colón et al., 2015).

*2.3.2.6. Mechanical biological treatment.* MBT facilities for mixed MSW were modelled as a combination of sorting and biological treatment processes. Variations labelled as "advanced" in this work include materials sorting for recycling, where recovery efficiencies were based on Cimpan et al. (2015). Degradation and emissions generation from biological processes were assumed to follow the same patterns as for treatment of the wet fraction, where the same type of process and air treatment was employed. Emissions for the simple biological drying MBT were assumed similar to enclosed composting, with the difference that the high rate of aeration prevents formation of methane. Emissions for the advanced biological drying MBT were based on the LCI data in Rigamonti et al. (2012), for a facility employing regenerative thermal oxidation (RTO).

*Biological drying:* Biological drying or biodrying is a variation of aerobic decomposition (composting) performed in closed reactors, whereby the biological heat produced by microorganisms in the initial stages of decomposition is harnessed and augmented by intense forced aeration which facilitates the fast removal of moisture by convective evaporation (Velis et al., 2009). The process runs between 5 and 15 days (batch-wise), depending on the technology provider. In contrast to classical composting processes, which aim at maximum degradation, the objective in biodrying is the fast removal of moisture, with minimum substrate degradation, until biological activity stops (15–20 °C), rendering the output material storable for short-term. The substrate is biodried within air- and liquid-tight box reactors. Filling/unloading can be done completely automatically by means of cranes or manually by means of wheel loaders. A summary of the consumption and emissions parameters is presented in Table 8.

*2.3.3. Functional outputs and LCI data for background (affected) processes*

The foreground systems modelled in this study result in final recovered material or energy outputs and/or final sinks (i.e. final deposit in ground, emissions to air, water and soil). The former are called functional outputs, because they constitute products that are sold on related markets and can replace alternative supplies of the same function (called avoided or substituted flows). The processes leading to final recovery and the framework used for substitution are presented in the following sections.

*2.3.3.1. Electricity and heat.* Electricity for both process consumption and avoided/substituted production were modelled with LCI data for Brazil imported from the ecoinvent database. A simple technology marginal was chosen to represent the current state and short-term development of electricity production in Brazil in accordance with the analysis carried by Bernstad Saraiva et al.

**Table 7**
Parameters adopted for the biological treatment processes (biogas upgrading and combustion not included here).

| Process consumptions and direct emissions | Unit | Composting - Open | Composting - Enclosed | Wet anaerobic digestion | Dry anaerobic digestion |
|---|---|---|---|---|---|
| *Pre-treatment* | | | | | |
| Electricity (Mechanical) | kWh $t^{-1}$ input | 15 | 15 | 15 | 15 |
| Electricity (Pulping) | kWh $t^{-1}$ input | – | – | 41 | – |
| Water (Pulping) | $m^3$ $t^{-1}$ input | – | – | 1.2 | – |
| *Main biological treatment* | | | | | |
| Electricity | kWh $t^{-1}$ input | 0.2 | 53 | 20 | 30 |
| Diesel | L $t^{-1}$ input | 3 | 1 | 0.5 | 1.5 |
| Heat* | MJ $t^{-1}$ input | – | – | 60.3 | 57.6 |
| *Stabilization and post-treatment* | | | | | |
| Electricity | kWh $t^{-1}$ input | Included in main treatment | Included in main treatment | 50%* (open windrow composting) | 50%* (open windrow composting) |
| Diesel | L $t^{-1}$ input | | | | |
| *Emissions to air* | | | | | |
| $CH_4$ AD (fugitive) | % $CH_4$ biogas | n.a. | n.a. | 2 | 2 |
| $CH_4$ aerobic treatment | % C degraded | 2.24 | 2.24 (0.05) | stabilization based on open windrow composting parameters | stabilization based on open windrow composting parameters |
| $N_2O$ | % N degraded | 15 | 1.4 | | |
| $NH_3$ | % N degraded | 83 | 83 (0.01) | | |
| NMVOCs | kg $t^{-1}$ input | 2 | 2 (0.05) | | |

* Only in scenario systems with biogas upgrading (when biogas is used directly for energy production it is assumed that heat needs are covered on site).

**Table 8**
Parameters adopted for the MBT processes.

| Process consumptions and direct emissions | Unit | Simple aerobic MBT | Advanced anaerobic–aerobic MBT | Simple biological drying MBT | Advanced biological drying MBT |
|---|---|---|---|---|---|
| *Process consumptions* | | | | | |
| Electricity | kWh $t^{-1}$ input | 70 | 80 | 70 | 90 |
| Diesel | L $t^{-1}$ input | 2.5 | 3 | 2.5 | 2 |
| Heat* | MJ $t^{-1}$ input | – | 57.6 | – | – |
| Steel wire | kg $t^{-1}$ input | – | 0.13 | – | 0.13 |
| NG | $m^3$ $t^{-1}$ input | – | – | – | 2 |
| *Emissions to air* | | | | | |
| $CH_4$ AD (fugitive) | % $CH_4$ biogas | n.a. | 2 | n.a. | n.a. |
| $CH_4$ aerobic treatment | % C degraded | 2.24 (0.05) | stabilization based on enclosed windrow composting parameters | 0 | 0 |
| $N_2O$ | % N degraded | 1.4 | | 1.4 | 8.6** |
| $NH_3$ | % N degraded | 83 (0.01) | | 83 (0.01) | 8** |
| NMVOCs | kg $t^{-1}$ input | 2 (0.05) | | 2 (0.05) | 7.7** |
| $NO_x$ | g $t^{-1}$ input | – | | – | 70.00 |
| $SO_x$ | g $t^{-1}$ input | – | | – | 0.15 |
| $CO_2$ fossil (from NG combustion) | kg $t^{-1}$ input | n.a. | n.a. | n.a. | 4.00 |

* Only in scenario systems with biogas upgrading (when biogas is used directly for energy production it is assumed that heat needs are produced on site from natural gas).
** The unit is g $t^{-1}$ input (Rigamonti et al., 2012).

(2017). The authors identified natural gas based electricity production (combined cycle) as the most likely technology to respond in the electricity market. A grid loss factor of 3.9% was applied to differentiate consumption (medium voltage) and substitution (high voltage). Heat consumption in anaerobic digestion was considered to be covered by cogeneration in the case where biogas in used directly in gas motors. Conversely, when biogas was upgraded to biomethane, it was assumed that heat would be provided by a natural gas boiler. More often, some of the biogas would be directly used for heat production and reduce the amount of biomethane production. The total effect would in both cases be relatively close, but here the aim was to test the full potential for diesel substitution.

*2.3.3.2. Reprocessing/recycling and avoided primary production.* Recycling and primary production processes were modelled as generic European and global processes since there is no data available from Brazil. The processes were designed according to EASETECH templates, based on Bassi et al. (2017) and Rigamonti et al. (2012). Recycling was defined by process recovery efficiencies (A) and avoided primary production considered market substitution ratios (B), which are described in the SM.

*2.3.3.3. Upgrading of biogas and use as vehicle fuel.* Biomethane is used widely as vehicle fuel in Europe, replacing compressed natural gas (CNG) or liquefied natural gas (LNG) especially in busses and trucks. The fuel efficiency of biomethane used in internal combustion engines (ICE) is similar to conventional fuels such as gasoline, but is lower than for diesel by 10–15% (Cong et al., 2017; Delgado and Muncrief, 2015). Modelled processes included biogas upgrading by membrane technology (electricity consumption of 0.24 kWh $m^3$(-1)), biomethane compression and distribution (0.065 kWh $m^3$(-1), 2% methane loss). Use of biomethane was considered to substitute production and utilization of diesel in an equivalent application (large commercial vehicle), considering a substitution factor of 1:0.9 (MJ:MJ). Biomethane vehicle emissions were based on emission inventories for regular CNG (with the exception of fossil $CO_2$), an assumption supported by studies such as Hakawati et al. (2017).

*2.3.3.4. RDF to cement kilns.* RDF combustion in a cement kiln was modelled with the EASETECH process template for WtE, by applying the input specific transfer coefficients to air given in Genon and Brizio (2008). The process avoids the thermal energy equivalent of petroleum coke use, including its production and combustion. Coke combustion emissions were calculated based on the same transfer coefficients (used for RDF) applied to the average coke composition in Genon and Brizio (2008). Details on emissions modelling in the SM.

### 2.4. Sensitivity analysis

Uncertainties with regard to overall technology options applied in the system scenarios were tackled to some extent by modelling technologies that could cover a large interval in environmental impacts, hence the large number of system variations (e.g. open and enclosed biological treatment). Nevertheless, many parameters used in this study suffer from large uncertainty and variability, but due to lack of data to many of the processes in a Brazilian context, measuring uncertainty is a near impossible endeavour. In this work, we instead tested the sensitivity of baseline results to the variation of a number of important parameters, namely: carbon storage for landfills, electricity marginal and RDF-coke substitution ratios. Furthermore, scenario variations that contained anaerobic digestion were tested by replacing baseline open post-composting with enclosed post-composting. The summary of the performed sensitivity is shown in Table 9.

## 3. Results

The LCA results are presented in the following sections, as normalized values in mili Person Equivalents (mPE), which allows the comparison between the impact categories. Following an overall comparison of systems, we elaborate by a process contribution analysis and results of the sensitivity analysis. Furthermore, characterization results for all impact categories and scenarios can be found in the SM.

### 3.1. Overall comparison of systems and impact categories

Table 10 shows the normalized net result for each impact category for all system variations, with green and red highlights representing best and worst performing variations. The net represents the sum of environmental burdens and benefits, and thus a positive net denotes an overall impact while a negative one a net saving within an impact category.

At a first glance, it can be observed that the first category, i.e. disposal systems, and in particular 1.a semi-controlled and 1.b controlled dumping, which represent a significant part of current management in Brazil had the highest impact in several categories, including global warming (GWP), ozone depletion (ODP), human toxicity, cancer effects (HT, CE), marine eutrophication (EPM) and freshwater ecotoxicity (ECF). It is important to note that the implementation of some controls, mainly landfill covers, in 1.b can be credited only marginal effects towards mitigating impacts. Concurrently, category three, i.e. mechanical-biological systems showed the highest overall savings in global warming (GWP), ozone depletion (ODP), particulate matter (PT), photochemical ozone formation (POF), terrestrial acidification (TAD), terrestrial eutrophication (EPT) and freshwater ecotoxicity (ECF). Category two, i.e. systems based on wet-dry separate collection, displayed highly mixed results. Systems that included composting or dry/wet

**Table 9**
Parameters and description of the sensitivity analysis performed.

| Sensitivity (parameter/technology) | Variation description | Scenarios where applied |
|---|---|---|
| Carbon storage in sanitary landfills | Remaining C after 100 years was set to 0% | 1.c and 1.d |
| Electricity marginal | Replaced by the Brazilian production mix (MME, 2017) | 1.d, 1.e, 2.d and 2.e |
| RDF-coke substitution ratios | Changed from 1:1 to 1:0.9 (energy content based) | 3.a, 3.b, 3.c and 3.d |
| Post-composting of digestate after anaerobic digestion | Changed from open to enclosed processes | 2.d, 2.d(u), 2.e and 2.e(u) |

**Table 10**
Normalized net results in mili Person Equivalents (mPE) for Climate Change (GWP), Ozone Depletion (ODP), Human Toxicity, Cancer Effects (HT, CE), Human Toxicity, non-Cancer Effects (HT, non CE), Particulate Matter (PT), Photochemical Ozone Formation (POF), Terrestrial Acidification (TAD), Eutrophication Terrestrial (EPT), Eutrophication Freshwater (EPF), Eutrophication Marine (EPM), Ecotoxicity Freshwater (ECF) and Depletion of Abiotic resources, Mineral fossil and Renewable (DAMR).

| | Cat. 1 – Mixed waste disposal systems | | | | | Cat. 2 – Wet-dry separate collection systems | | | | | | | | Cat. 3 – Mechanical –biological systems | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1.a | 1.b | 1.c | 1.d | 1.e | 2.a | 2.b | 2.c(w) | 2.c(e) | 2.d | 2.d(u) | 2.e | 2.e(u) | 3.a | 3.b | 3.b(u) | 3.c | 3.d |
| GWP | 146.7 | 132.4 | 30.5 | 24.7 | 25.5 | 16.2 | 15.0 | 18.3 | -4.9 | 6.3 | 0.7 | 4.8 | 0.0 | -37.8 | -42.6 | -48.5 | -51.7 | -48.5 |
| ODP | 38.6 | 37.2 | 15.5 | 16.6 | -0.8 | 13.1 | 13.0 | 1.7 | 1.7 | 1.0 | 0.4 | -1.4 | -2.0 | -3.7 | -4.1 | -4.8 | -5.5 | -5.2 |
| HT, CE | 38.0 | 37.7 | 8.9 | 6.6 | -15.4 | -14.2 | -18.9 | -131.4 | -131.6 | -134.4 | -137.5 | -159.2 | -162.5 | 36.0 | 24.2 | 19.3 | -38.2 | -49.3 |
| HT, non CE | 27.8 | 27.6 | 18.2 | 16.4 | 16.2 | 12.9 | 10.8 | 1025.9 | 1024.8 | 1023.9 | 1017.8 | 457.6 | 451.1 | 138.1 | 141.2 | 134.2 | 126.7 | 129.5 |
| PT | 1.5 | 1.5 | 2.0 | 2.3 | -1.7 | -8.3 | -11.0 | 22.0 | -11.3 | 22.3 | 12.9 | 20.3 | 10.3 | -30.1 | -29.0 | -27.2 | -35.9 | -34.4 |
| POF | 16.4 | 15.0 | 8.8 | 16.7 | 16.6 | 8.2 | 7.0 | 22.4 | -3.1 | 17.7 | -0.2 | 5.6 | -13.3 | -15.2 | 6.1 | -12.3 | -15.9 | -21.2 |
| TAD | 1.3 | 1.3 | 2.5 | 6.4 | 9.7 | -1.2 | -2.7 | 132.1 | -5.2 | 134.9 | 127.8 | 125.9 | 118.4 | -37.6 | -25.0 | -30.0 | -47.3 | -35.7 |
| EPT | 2.0 | 2.0 | 3.2 | 11.0 | 19.4 | 5.9 | 5.3 | 192.6 | 0.3 | 196.9 | 189.3 | 183.3 | 175.3 | -1.4 | 2.6 | -5.3 | -5.5 | -5.8 |
| EPF | 2.5 | 2.5 | 2.0 | 1.8 | -0.9 | 2.0 | 1.2 | -25.9 | -25.9 | -26.0 | -29.1 | -25.5 | -28.7 | 0.5 | 4.9 | 1.6 | -4.9 | -0.6 |
| EPM | 48.0 | 47.7 | 3.0 | 7.5 | 11.2 | 4.6 | 4.3 | 11.9 | 3.4 | 17.5 | 13.0 | 16.0 | 11.3 | 7.2 | 12.7 | 8.1 | -3.7 | -3.5 |
| ECF | 21.1 | 20.9 | 7.4 | 5.9 | -7.5 | 3.2 | 2.3 | 9.2 | 9.4 | 7.7 | 8.2 | 0.0 | 0.5 | 19.4 | 18.1 | 17.6 | -6.6 | -7.2 |
| DAMR | 0.0 | 0.0 | 0.1 | 0.0 | -1.9 | -7.6 | -10.0 | -9.8 | -10.0 | -10.1 | -11.3 | -10.0 | -11.3 | -4.4 | -9.9 | -11.2 | -4.8 | -10.3 |

digestion of wet waste had high savings in HT, CE and EPF. The same systems showed high impacts in non-cancer effects (HT, non CE). The system variations which included open composting technologies, including after prior digestion of the wet stream (i.e. 2.c(w), 2.d, 2.d(u), 2.e and 2.e(u)), displayed particularly high impacts in PT, POF, TAD and EPT. These impacts seemed mitigated with enclosed composting, i.e. in 2.c(e). All category two systems contributed savings in depletion of abiotic resources, mineral fossil and renewable (DAMR), although category three systems based on advanced (recovery) models showed similar results. Surprisingly, category two systems based on digestion of the wet stream did not have GWP savings and performed similar to variants with composting or sanitary landfilling of the wet stream.

### 3.2. Process contribution analysis

Process contributions are illustrated with Figs. 1–3. Bars above and below the X axis denote burdens and savings, respectively.

#### 3.2.1. Systems based on direct disposal of mixed waste (category 1)

Fig. 1 illustrates process contributions to category 1 systems, in which it is clear that improper landfilling (scenarios 1.a and 1.b, semi-controlled and controlled dumps respectively) has a high burden in many impact categories. The biggest contributors for these high impacts are landfill gases (in GWP and ODP) and untreated leachate (in HT, CE, HT, non CE, EPM and ECF). Fully controlled sanitary landfilling reduced the GWP in scenario 1.c by roughly 5 times compared to 1.a, from 1232 to 256 kg $CO_2$ eq. per ton of waste, as well as the high impact of untreated leachate in several categories. Landfill gas utilization for the production of electricity in 1.d had a beneficial effect on GWP, HT (both cancer and non-cancer) and ECF, but it contributed burdens in ODP, PT, POF, TAD, EPT and EPM. This was connected mainly to emissions of nitrogen oxides ($NO_x$) and CFCs in the combustion process (gas motors). In our model, the process for flaring had a higher efficiency in destruction of CFCs and generated lower NOx compared to the process for gas motor. This difference in process emissions does not necessarily discredit gas utilization, but signals the importance of choosing the right technology, which would ensure emission reduction across the board.

Scenario 1.e, with the combustion WtE plant, performed best in several categories. However, for categories POF, TAD, and EPT it also presented the highest impacts. These high impacts once more came mainly from $NO_x$ emissions. Nevertheless, energy recovery and substitution of marginal electricity scenario 1.e contributed significant savings in GWP, ODP, HT (CE), PT, EPF, ECF and DAMR (due to steel recycling). The results of this scenario are similar to those of Reichert and Mendes (2014) and confirm that strategies based on energy recovery are not significantly better than sanitary landfilling, even if they displace natural gas based electricity. The results for WtE are highly dependent on the type of electricity assumed displaced in Brazil.

#### 3.2.2. Systems based on source separation into wet and dry streams (category 2)

Fig. 2 captures the breakdown of normalized results for the second category systems, which are based on source separation of dry-wet streams in a ratio of 20:80. The first two systems variants, 2.a and 2.b, combine dry stream sorting and subsequent materials recycling with simple disposal of the wet stream by sanitary landfilling. Variants from 2.c to 2.e include wet stream pre-treatment and composting (2.c) or anaerobic digestion (2.d and 2.e).

In systems 2.a and 2.b, around 11% and 12% per FU, of waste materials are directed to recycling after sorting of the dry stream.

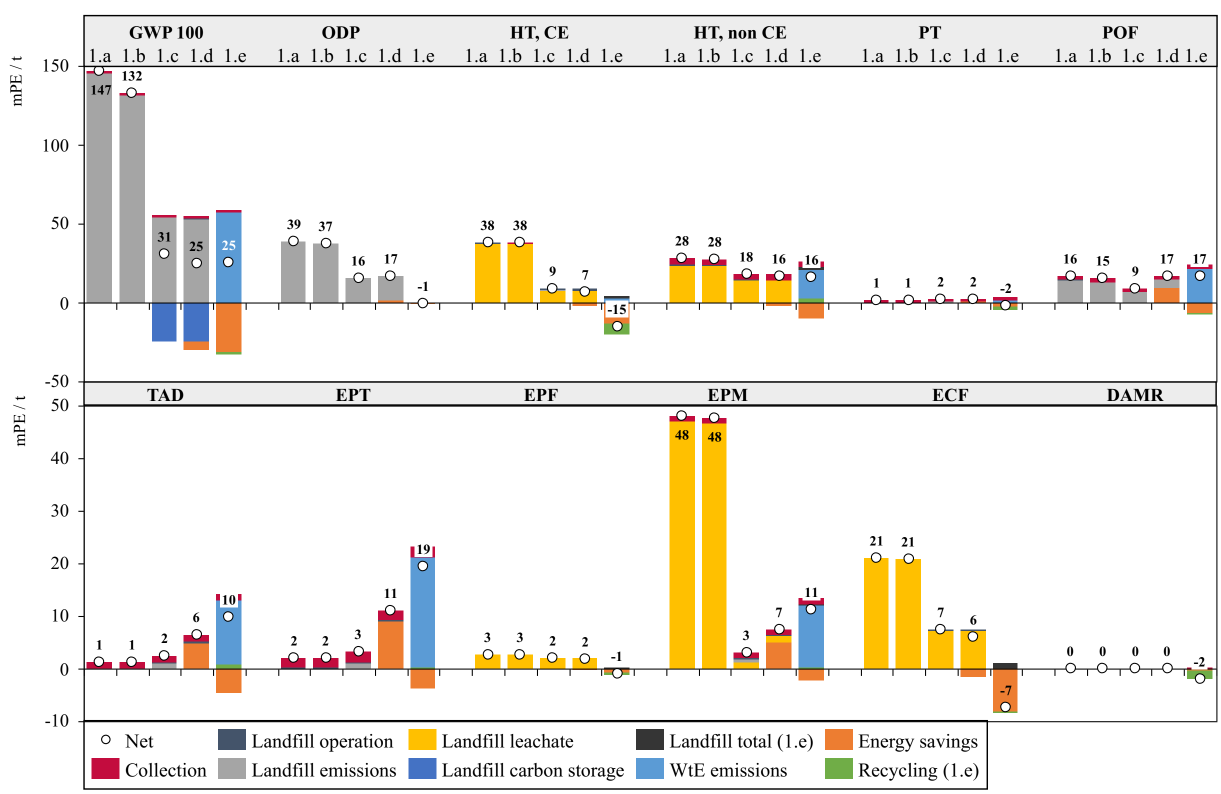


**Fig. 1.** Normalized results in mili Person Equivalents (mPE) for Category 1 systems for: Climate Change (GWP), Ozone Depletion (ODP), Human Toxicity, Cancer Effects (HT, CE), Human Toxicity, non Cancer Effects (HT, non CE), Particulate Matter (PT), Photochemical Ozone Formation (POF), Terrestrial Acidification (TAD), Eutrophication Terrestrial (EPT), Eutrophication Freshwater (EPF), Eutrophication Marine (EPM), Ecotoxicity Freshwater (ECF) and Depletion of Abiotic resources, Mineral fossil and Renewable (DAMR).

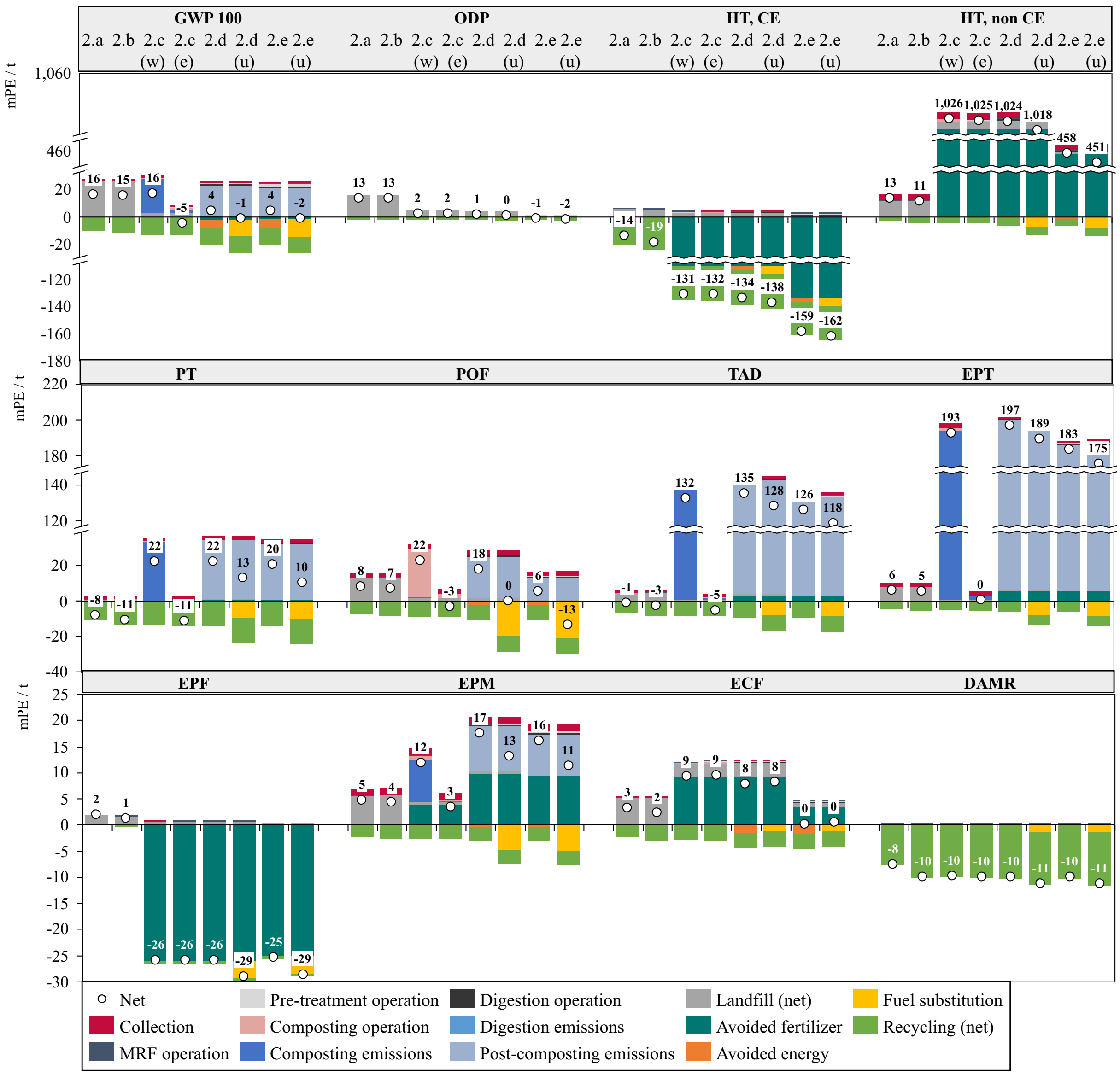


**Fig. 2.** Normalized results in mili Person Equivalents (mPE) for Category 2 systems for Climate Change (GWP), Ozone Depletion (ODP), Human Toxicity, Cancer Effects (HT, CE), Human Toxicity, non Cancer Effects (HT, non CE), Particulate Matter (PT), Photochemical Ozone Formation (POF), Terrestrial Acidification (TAD), Eutrophication Terrestrial (EPT), Eutrophication Freshwater (EPF), Eutrophication Marine (EPM), Ecotoxicity Freshwater (ECF) and Depletion of Abiotic resources, Mineral fossil and Renewable (DAMR).

It can be noticed that emissions from landfilling in scenarios 2.a and 2.b in general overcame potential savings from materials recycling, for GWP, ODP, HT (non CE), EPM and ECF. Nevertheless, dry stream recycling contributed savings in almost all impact categories. For example, GWP was halved compared to complete mixed waste sanitary landfilling in 1.c. Operation of the MRFs had an insignificant impact.

Further on, the addition of wet stream treatment resulted in interesting observations. At first glance results suggested system 2.c(e), which is based on wet stream enclosed composting, as having the least impact in most categories. Open air windrow composting (2.c(w)) was affected by larger emissions to air than enclosed composting and further, as the process was also modelled for digestate stabilization, it negatively affected all scenario systems based on wet stream digestion. In scenarios with digestion, although methane is largely removed (aside from fugitive emissions), N-based emissions remain largely unchanged. The potential impacts are connected to input and process specific emissions of methane, $N_2O$, $NH_3$ and NMVOCs. The system choice for digestate stabilization was thus indicated as a hot spot and tested in a specific sensitivity analysis, which changed substantially the initial picture.

Compost and stabilized digestate application on (agricultural) soil also displayed relatively extreme results, either savings in HT, CE and EPF or high burdens in HT, non CE, EPM and ECF. The process displayed in Fig. 2, named "avoided fertilizer", accounts the net effect of application to soil and avoided mineral fertilizers. The savings were tracked to heavy metals, specifically chromium that is avoided from the use of the mineral fertilizers. The burdens were similarly traced to heavy metals (chromium, nickel, lead and mercury) present in the compost after the treatment of the wet fraction. More precisely, the heavy metals came from the fraction "other non-combustibles" part of waste matrix. The systems with wet digestion, which included a secondary pre-treatment, namely pulping, had a smaller impact in HT, non CE and ECF, due to the better overall removal of this fraction from the input to the digestion process.

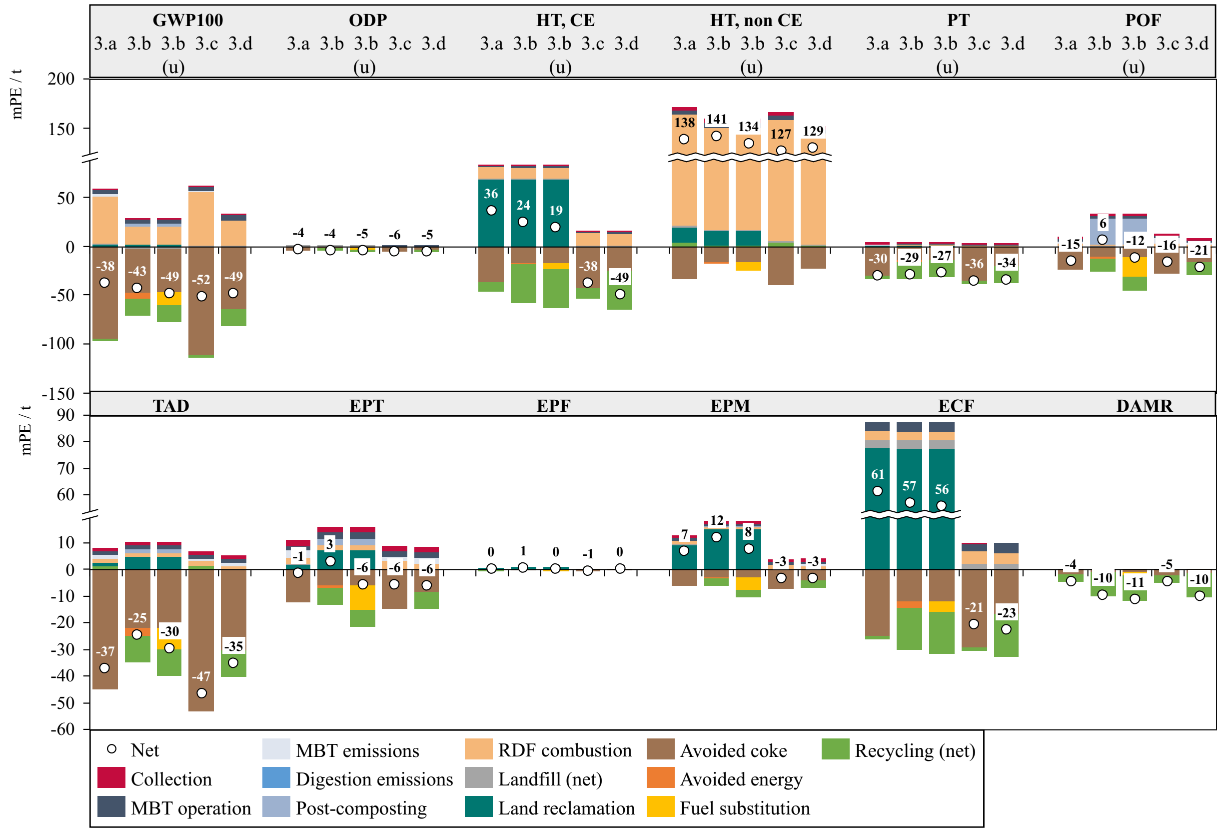


**Fig. 3.** Normalized results in mili Person Equivalents (mPE) for Category 3 systems for Climate Change (GWP), Ozone Depletion (ODP), Human Toxicity, Cancer Effects (HT, CE), Human Toxicity, non Cancer Effects (HT, non CE), Particulate Matter (PT), Photochemical Ozone Formation (POF), Terrestrial Acidification (TAD), Eutrophication Terrestrial (EPT), Eutrophication Freshwater (EPF), Eutrophication Marine (EPM), Ecotoxicity Freshwater (ECF) and Depletion of Abiotic resources, Mineral fossil and Renewable (DAMR).

Lastly, both systems with dry and wet digestions produced similar amounts of biogas, with the slightly higher wet digestion efficiency being compensated by additional loss of organics in the pulping process. Utilization of the biogas directly for electricity production resulted in small savings in several categories, while biogas upgrading and utilization as vehicle fuel showed significantly higher benefits (scenarios 2.d(u) and 2.e(u)).

### 3.2.3. Systems based on mechanical–biological treatment (category 3)

The results for category three systems are illustrated in Fig. 3. System variants here achieved net savings in all but a few impact categories, the results beings relatively similar, but favouring to some extent the two variants based on mixed waste treatment in biological drying MBTs. The operation of the MBTs, just like MRFs in category two systems, did not incur any significant impacts. RDF production and utilization in cement manufacturing contributed large savings connected to avoided petroleum coke production and combustion. Direct emissions from RDF combustion resulted in a bigger impact, compared to savings by avoided coke, in only one impact category, namely HT, non CE. The contribution to this impact was due to release to air of volatile heavy metals (specifically Hg and Pb). Considering that no specific and intensive mechanical treatment of the RDF was included to upgrade this treatment output, the results are positive towards demonstrating the big potential for the application of RDF in the cement industry in Brazil.

Around 14% of the input waste was further recovered in outputs destined for recycling (i.e. metals, plastics, paper and cardboard) in system variants 3.b and 3.d, which intended to represent versions of facilities where material recovery would take place besides treatment of the organics and RDF production. In these variants recycling contributed significant savings to different impact categories.

"Land reclamation", which is a low-grade utilization of the compost-like output or stabilized digestate from aerobic or aerobic-anaerobic MBTs respectively, resulted in impacts for HT, CE and ECF due to heavy metals (zinc, copper and chromium mainly). This was somewhat expected, as these systems have input mixed waste and the stabilized outputs would typically not achieve the requirements to be used as fertilizer, without substantial pre- or post-processing.

## 3.3. Sensitivity results

The sensitivity analysis results for climate change can be observed in Fig. 4 where they are compared with the baseline net results.

Setting carbon storage in sanitary landfills to 0% after 100 years, resulted in almost doubling the GWP impact (81% increase for 1.c and 100% for 1.d). This change would favour combustion WtE as the better alternative to direct disposal of mixed waste. The change in this parameter does not affect other impact categories. The change of digestate post-treatment technology from open windrows composting to enclosed composting for system scenarios 2.d, 2.d(u), 2.e and 2.e(u), resulted, as expected, in a substantial performance improvement in all the categories previously dominated by air emissions from open composting (i.e. GWP, PT, POF, TAD, EPT and EPM).

Replacing marginal electricity (i.e. based on combined cycle natural gas) with the Brazilian average production mix in the

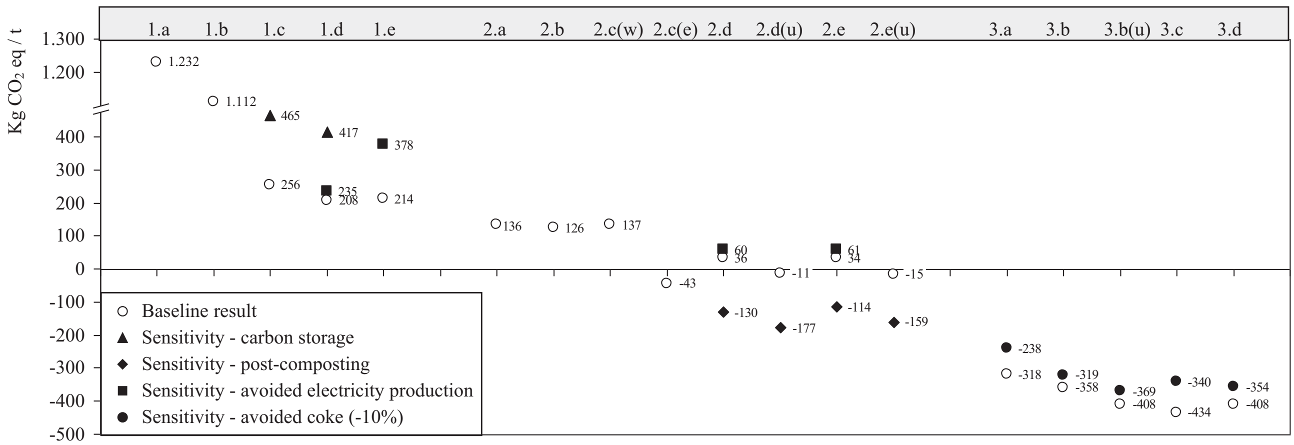


**Fig. 4.** Sensitivity results in kg $CO_2$ eq. for Climate Change (GWP).

scenarios with avoided electricity, i.e. 1.d, 1.e, 2.d and 2.e., resulted in an increase of GWP. However, only 1.e (combustion WtE) was severely affected, by almost doubling the GWP impact. Impact in other categories did not increase, but on the contrary, HT, non CE and TAD decreased for all the scenarios. This was traced to avoided emissions of zinc and arsenic from ethanol production, which is part of the Brazilian electricity mix.

Lastly, a decrease in coke substitution ratio in category three systems, from 1:1 to 1:0.9, resulted in proportional effects in relevant savings. The 10% change in the substitution ratio, affected especially systems in 3.a and 3.c, determining a decrease in GWP savings by 22–25% (from −318 to −238), while systems 3.b and 3.d displayed a decrease of only 11–13%.

## 4. Discussion

The system scenarios evaluated in this work were intended to be technology-centric, and thus as potential management scenarios for Brazil they are not at all exhaustive, especially if one is to consider the variety of technology combinations possible. This work thus mainly clarifies the potential environmental burdens and benefits of the techniques compared, which should then be used in the planning of integrated management systems that consider particularities of specific catchment areas (e.g. population density, socio-economic conditions). The impacts of current management of collected MSW in Brazil can be very roughly estimated by aggregating systems 1.a, 1.b and 1.c analysed here. The normalized results for this exercise are illustrated in Fig. 5. For GWP, as example, they

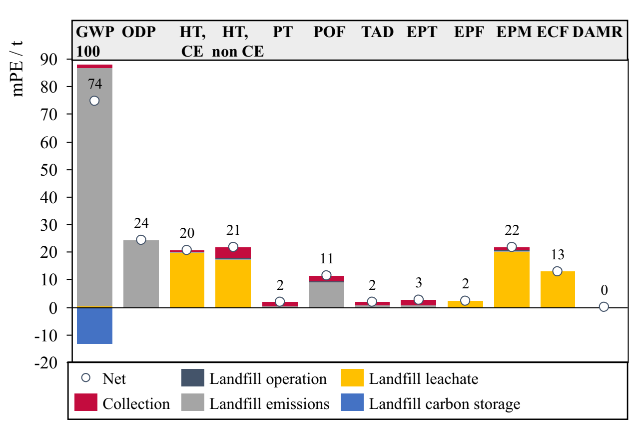


**Fig. 5.** Impact for the average management of MSW collected in Brazil in 2016, considering the ratios given in the introduction (17% semi-controlled dumps, 25% controlled dumps and respectively 54% sanitary landfills with gas flaring).

suggest a potential impact of 626 kg $CO_2$ eq. $t^{-1}$, which extrapolated to national level would account for around 48 million t of $CO_2$ eq. related to the disposal of MSW collected in one year.

Separate collection programmes are slowly expanding in Brazil, but not uniformly, as they are typically implemented in limited (typically affluent) areas. Where dry recyclables collection has been implemented, even after many years, diversion rates only reach around 10% (Ibáñez-Forés et al., 2017). Separate collection based on a three-stream system (dry recycling, biowaste and residual waste) is theoretically possible, but realistically unlikely to have significant coverage in short-to-medium term. Nevertheless, in this work, we observed that simple dry-wet collection could pose problems with regard to the possible quality of compost outputs, as the wet stream is still contaminated even with comprehensive pre- and post-treatment. Separate collection of only biowaste is of course not a guarantee that the stream will be substantially cleaner, but it should be especially prioritized in cases where large homogeneous quantities are generated, such as retail, service industry and food production.

From scenarios 2.a, 2.b, 3.a and 3.b it was possible to calculate theoretical recycling rates for the scenarios (on dry recyclables). The highest recycling rate was achieved in scenario 3.b and 3.d, with a 14.5% recovery rate. The presence of MRFs and MBTs with expanded sorting to recover various materials for recycling is fully established in places such as North America and Europe. In Europe, residual waste processing is increasingly seen as a solution to areas with inherently low citizen participation in separate collection, such as urban areas with high population densities and regions where cultural and socio-economic barriers persist (Trulli et al., 2018). The efficiency of such recovery systems has been confirmed, even when compared to or supplementing well running separate collection systems (Brouwer et al., 2018; Dahlbo et al., 2018; Feil et al., 2017). The present work also confirms their environmental feasibility in a Brazilian context. Further, MBT systems can be modular, with various degrees of automation and corresponding manual labor requirements, and connected infrastructure costs, fitting various local situations. Nevertheless, it should be considered that in the more advanced concepts, these plants require significant investment and trained personnel. Moreover, operation is highly dependent on diligent maintenance and plant efficiency is influenced more than for other technologies, such as WtE, by operational practice.

In Brazil, there is considerable urgency for both comprehensive, science informed, long-term strategy planning and immediate action to mitigate the impact of current improper practices. Progress on the ground is slowed by considerable economic, social and local political challenges (Campos, 2014). As put by Rodić and

Wilson (2017), "no technology could on its own solve the problems related to economic and social sustainability of waste management activities", pointing further out that necessary action in developing countries has to be focused on governance issues. Comprehensive analyses of MSW management in Brazil are further hindered by several aspects. Brazil does not have standards for waste gravimetric analysis, such as for which fractions to consider and how to deliver the results. Therefore, the data found regarding waste compositions has uncertainties that are difficult to estimate. Furthermore, physico-chemical properties used in most studies to date, including the present work, are not based on analyses of Brazilian waste. Variations in composition and physico-chemical properties can alter, sometimes significantly, LCA results (Bisinella et al., 2017). Another aspect that limits precision, is the always present intervention by the informal sector, which plays an important role in the waste management system in Brazil. The efficiency and scale of their interception is difficult to measure and thus typically ignored. Most municipal analyses do not even mention these workers, which limits the possibility to include environmental, economic and social contributions of the informal sector to the whole system.

## 5. Conclusions

This comparison between three different sets (categories) of systems provides an overview of the current and alternative technology-centric waste management alternatives for Brazil. The first category of systems assessed options for direct disposal of MSW, including still prominent improper waste disposal systems in Brazil, namely dumps, alongside sanitary landfills and combustion WtE. The results confirmed the high environmental cost of improper disposal (still 41.6% of the current disposal in Brazil) and provided evidence that combustion WtE does not offer significant benefits over sanitary landfilling, due to limitations in energy utilization and the low-carbon background electricity system. Category two of systems, based on source separation into wet and dry streams, showed a better environmental performance. Recycling contributed significant savings, however particular attention needs to be focused on treatment of biodegradable waste. The use of technologies including treatment of air emissions from degradation processes were shown essential, even after prior anaerobic digestion processes. Biogas upgrading and use as vehicle fuel resulted in bigger savings compared to electricity production. The use of compost outputs was indicated as potentially detrimental due to contamination levels (heavy metals) in the wet stream. As for category three systems, mechanical-biological systems had environmental benefits in most impact categories. The major contributor was RDF production and utilization in cement production, substituting petroleum coke. MSW-derived RDF utilization needs further investigation in a Brazilian context, to test technical and economic feasibility and validate environmental feasibility. Lastly, MBT systems that include extended capabilities to recover recyclable materials, could also make significant contributions to recycling in Brazil.

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## References

ABRELPE, 2017. Panorama dos resíduos sólidos no Brasil 2016. https://doi.org/ISSN 2179-8303.

ABRELPE - Associação Brasileira de Empresas de Limpeza Pública e Resíduos Especiais, 2013. Atlas Brasileiro de Emissões de GEE e Potencial Energético na Destinação de Resíduos Sólidos.

de S.M. Alfaia, R.G., Costa, A.M., Campos, J.C., 2017. Municipal solid waste in Brazil: A review. Waste Manage. Res. https://doi.org/10.1177/0734242X17735375. 0734242X1773537.

Amlinger, F., Peyr, S., Cuhls, C., 2008. Green house gas emissions from composting and mechanical biological treatment. Waste Manage. Res. 26, 47–60. https://doi.org/10.1177/0734242X07088432.

Andersen, J.K., Boldrin, A., Christensen, T.H., Scheutz, C., 2010. Mass balances and life-cycle inventory for a garden waste windrow composting plant (Aarhus, Denmark). Waste Manage. Res. 28, 1010–1020. https://doi.org/10.1177/0734242X10360216.

De Aquino, I.F., De Castilho Jr., A.B., Pires, T.S.D.L., 2009. A organização em rede dos catadores de materiais recicláveis na cadeia produtiva reversa de pós-consumo da região da grande Florianópolis: uma alternativa de agregação de valor. Gestão & Produção 16, 15–24. https://doi.org/10.1590/S0104-530X2009000100003.

Bassi, S.A., Christensen, T.H., Damgaard, A., 2017. Environmental performance of household waste management in Europe - an example of 7 countries, Technical University of Denmark. Lyngby, Denmark. https://doi.org/10.1016/j.wasman.2017.07.042.

Bernstad Saraiva, A., Souza, R.G., Valle, R.A.B., 2017. Comparative lifecycle assessment of alternatives for waste management in Rio de Janeiro – Investigating the influence of an attributional or consequential approach. Waste Manage. 68, 701–710. https://doi.org/10.1016/j.wasman.2017.07.002.

Bisinella, V., Götze, R., Conradsen, K., Damgaard, A., Christensen, T.H., Astrup, T.F., 2017. Importance of waste composition for Life Cycle Assessment of waste management solutions. J. Clean. Prod. 164, 1180–1191. https://doi.org/10.1016/j.jclepro.2017.07.013.

Brasil, 2010. Lei 12.305/2010. Institui a Política Nac. Resíduos Sólidos 21.

Brouwer, M.T., Thoden van Velzen, E.U., Augustinus, A., Soethoudt, H., De Meester, S., Ragaert, K., 2018. Predictive model for the Dutch post-consumer plastic packaging recycling system and implications for the circular economy. Waste Manage. 71, 62–85. https://doi.org/10.1016/j.wasman.2017.10.034.

Campos, H.K.T., 2014. Recycling in Brazil: Challenges and prospects. Resour. Conserv. Recycl. 85, 130–138. https://doi.org/10.1016/j.resconrec.2013.10.017.

Christensen, T.H., Gentil, E., Boldrin, A., Larsen, A.W., Weidema, B.P., Hauschild, M., 2009. C balance, carbon dioxide emissions and global warming potentials in LCA-modelling of waste management systems. Waste Manage. Res. 27, 707–715. https://doi.org/10.1177/0734242x08096304.

Cimpan, C., Maul, A., Jansen, M., Pretz, T., Wenzel, H., 2015. Central sorting and recovery of MSW recyclable materials: A review of technological state-of-the-art, cases, practice and implications for materials recycling. J. Environ. Manage. 156, 181–199. https://doi.org/10.1016/j.jenvman.2015.03.025.

Cimpan, C., Maul, A., Wenzel, H., Pretz, T., 2016. Techno-economic assessment of central sorting at material recovery facilities - The case of lightweight packaging waste. J. Clean. Prod. 112, 4387–4397. https://doi.org/10.1016/j.jclepro.2015.09.011.

Cimpan, C., Wenzel, H., 2013. Energy implications of mechanical and mechanical-biological treatment compared to direct waste-to-energy. Waste Manage. 33, 1648–1658. https://doi.org/10.1016/j.wasman.2013.03.026.

Clavreul, J., Baumeister, H., Christensen, T.H., Damgaard, A., 2014. An environmental assessment system for environmental technologies. Environ. Model. Softw. 60, 18–30. https://doi.org/10.1016/j.envsoft.2014.06.007.

Clift, R., Doig, A., Finnveden, G., 2000. The application of life cycle assessment to integrated solid waste management. Trans IChemE 78, 279–287. https://doi.org/10.1205/095758200530790.

Colón, J., Cadena, E., Pognani, M., Maulini, C., Barrena, R., Sánchez, A., Font, X., Artola, A., 2015. Environmental burdens of source-selected biowaste treatments: comparing scenarios to fulfil the European Union landfill directive. The case of Catalonia. J. Integr. Environ. Sci. 12, 165–187. https://doi.org/10.1080/1943815X.2015.1062030.

Colvero, D.A., Pfeiffer, S.C., Carvalho, E.H. de, 2016. Materiais recicláveis provindos dos resíduos urbanos: caso de estudo para o estado de Goiás, Brasil. In: Ramísio, P.J., Lopes, G.A., Pinto, L.M.C., Leite, F., M.J.R. (Ed.), A Engenharia Sanitária Nas Cidades Do Futuro: Livro de Comunicações Do 17o Encontro de Engenharia Sanitária e Ambiental/ENASB. ISBN: 978-989-20-6908-8. Lisboa, Portugal, p. 888. https://doi.org/10.22181/17ENASB.2016.

Cong, R.G., Caro, D., Thomsen, M., 2017. Is it beneficial to use biogas in the Danish transport sector? – An environmental-economic analysis. J. Clean. Prod. 165, 1025–1035. https://doi.org/10.1016/j.jclepro.2017.07.183.

Dahlbo, H., Poliakova, V., Mylläri, V., Sahimaa, O., Anderson, R., 2018. Recycling potential of post-consumer plastic packaging waste in Finland. Waste Manage. 71, 52–61. https://doi.org/10.1016/j.wasman.2017.10.033.

Dallmann, T., Façanha, C., 2015. Brazil is not ready for diesel cars.

de Beer, J., Cihlar, J., Hensing, I., Zabeti, M., 2017. Recent Development on the Uses of Alternative Fuels in Cement Manufacturing Process.

DEFRA, 2011. Emissions from Waste Management Facilities, WR 0608. Department for Environment, Food and Rural Affairs (Defra), London, UK.

Delgado, O., Muncrief, R., 2015. Assessment of Heavy-Duty Natural Gas Vehicle Emissions: Implications and Policy Recommendations. Washington DC.

DTU, 2016. EASETECH Impact categories and impact methods Kgs. Lyngby, Denmark.

European Commission, 2011. Supporting Environmentally Sound Decisions for Waste Management. A Technical Guide to Life Cycle Thinking (LCT) and Life

Cycle Assessment (LCA) for Waste Experts and LCA Practitioners. Publications Office of the European Union, Luxembourg.
European Commission, 2010. General guide for Life Cycle Assessment - Detailed guidance, First. ed, International Reference Life Cycle Data System (ILCD) Handbook. Publications Office of the European Union, Luxembourg. https://doi.org/10.2788/38479.
European Commission, 2006. Best Available Techniques for the Waste Treatment Industries – reference document. Integrated Pollution Prevention and Control Bureau, Joint Research Centre, Seville, Spain.
Feil, A., Pretz, T., Jansen, M., Thoden Van Velzen, E.U., 2017. Separate collection of plastic waste, better than technical sorting from municipal solid waste? Waste Manage, Res. 35, 172–180. https://doi.org/10.1177/0734242X16654978.
Fricke, K., Santen, H., Wallmann, R., 2005. Comparison of selected aerobic and anaerobic procedures for MSW treatment. Waste Manage. 25, 799–810. https://doi.org/10.1016/j.wasman.2004.12.018.
Genon, G., Brizio, E., 2008. Perspectives and limits for cement kilns as a destination for RDF. Waste Manage. 28, 2375–2385. https://doi.org/10.1016/j.wasman.2007.10.022.
Goulart Coelho, L.M., Lange, L.C., 2016. Applying life cycle assessment to support environmentally sustainable waste management strategies in Brazil. Resour. Conserv. Recycl. https://doi.org/10.1016/j.resconrec.2016.09.026.
Hakawati, R., Smyth, B.M., McCullough, G., De Rosa, F., Rooney, D., 2017. What is the most energy efficient route for biogas utilization: Heat, electricity or transport? Appl. Energy 206, 1076–1087. https://doi.org/10.1016/j.apenergy.2017.08.068.
Hoornweg, D., Bhada-Tata, P., 2012. A Global Review of Solid Waste Management. World Bank Urban Dev. Ser. Knowl. Pap. 1–116. https://doi.org/10.1111/febs.13058.
Ibáñez-Forés, V., Bovea, M.D., Coutinho-Nóbrega, C., de Medeiros-García, H.R., Barreto-Lins, R., 2017. Temporal evolution of the environmental performance of implementing selective collection in municipal waste management systems in developing countries: A Brazilian case study. Waste Manage. 72, 65–77. https://doi.org/10.1016/j.wasman.2017.10.027.
IFC Increasing the Use of Alternative Fuels at Cement Plants : International Best Practice 2017 Washington DC.
Lagerkvist, A., Ecke, H., Christensen, T.H., 2011. Waste characterization: Approaches and methods. Solid Waste Technol. Manag.
Leme, M.M.V., Rocha, M.H., Silva, E.E.L., Lopes, B.M., Ferreira, C.H., 2012. Environmental assessment of energy recovery technologies for the treatment and disposal of municipal solid waste using Life Cycle Assessment (LCA): A case study of Brazil. In: Proceedings of ECOS 2012 – the 25th International Conference on Efficiency, Cost, Optimization, Simulation and Environmental Impact of Energy Systems. Perugia, Italy, pp. 1–9.
Leme, M.M.V., Rocha, M.H., Lora, E.E.S., Venturini, O.J., Lopes, B.M., Ferreira, C.H., 2014. Techno-economic analysis and environmental impact assessment of energy recovery from Municipal Solid Waste (MSW) in Brazil. Resour. Conserv. Recycl. 87, 8–20. https://doi.org/10.1016/j.resconrec.2014.03.003.
Manfredi, S., Christensen, T.H., 2009. Environmental assessment of solid waste landfilling technologies by means of LCA-modeling. Waste Manag. 29, 32–43. https://doi.org/10.1016/j.wasman.2008.02.021.
Mendes, M.R., Aramaki, T., Hanaki, K., 2004. Comparison of the environmental impact of incineration and landfilling in São Paulo City as determined by LCA. Resour. Conserv. Recycl. 41, 47–63. https://doi.org/10.1016/j.resconrec.2003.08.003.
MMA, M. do M.A., 2012. Plano Nacional de Resíduos Sólidos - PLANARES. Brasília, DF.
MME, M. de M. e E., 2017. Resenha Energética Brasileira - Exercício de 2016. Brasília - DF.
Møller, J., Jensen, M.B., Kromann, M., Neidel, T.L., Jakobsen, B., 2013. Miljø- og samfundsøkonomisk vurdering af muligheder for øget genanvendelse af papir, pap, plast, metal og organisk affald fra dagrenovation, Miljøprojekt nr. 1458. https://doi.org/978-87-92903-80-8.
Münnich, K., Mahler, C.F., Fricke, K., 2006. Pilot project of mechanical-biological treatment of waste in Brazil. Waste Manage. 26, 150–157. https://doi.org/10.1016/j.wasman.2005.07.022.
Naroznova, I., Møller, J., Larsen, B., Scheutz, C., 2016. Evaluation of a new pulping technology for pre-treating source-separated organic household waste prior to anaerobic digestion. Waste Manage. 50, 65–74. https://doi.org/10.1016/j.wasman.2016.01.042.
Olesen, O.U., Damgaard, A., 2014. Landfilling in EASETECH - Data collection and modelling of the landfill modules in EASETECH.
Prefeitura Municipal de Campo Grande, 2017. Plano de Coleta Seletiva de Campo Grande/MS. Estudo de caracterização física dos resíduos sólidos - versão 01. Campo Grande, MS, Brasil.
Reichert, G.A., Mendes, C.A.B., 2014. Avaliação do ciclo de vida e apoio à decisão em gerenciamento integrado e sustentável de resíduos sólidos urbanos. Eng. Sanit. e Ambient. 19, 301–313. https://doi.org/10.1590/S1413-41522014019000001145.
Riber, C., Petersen, C., Christensen, T.H., 2009. Chemical composition of material fractions in Danish household waste. Waste Manage. 29, 1251–1257. https://doi.org/10.1016/j.wasman.2008.09.013.
Rigamonti, L., Grosso, M., Biganzoli, L., 2012. Environmental assessment of refuse-derived fuel co-combustion in a coal-fired power plant. J. Ind. Ecol. 16, 748–760. https://doi.org/10.1111/j.1530-9290.2011.00428.x.
Rodić, L., Wilson, D.C., 2017. Resolving governance issues to achieve priority sustainable development goals related to solid waste management in developing countries. Sustain 9, 1–18. https://doi.org/10.3390/su9030404.
Rosa, B.P., Paula, B.C.D.L., Soares, E., Coleone, A., Campos, F., Cep, P., 2017. Impactos causados em cursos d' água por aterros controlados desativados no Município de São Paulo, Sudeste do Brasil. Rev. Bras. Gestão Ambient. e Sustentabilidade 4, 63–76.
Sala, S., Crenna, E., Secchi, M., Pant, R., 2017. Global normalisation factors for the Environmental Footprint and Life Cycle Assessment. https://doi.org/10.2760/88930.
Schalch, V., Leite, W.C. de A., Fernandes Junior, J.L., De Castro, M.C.A.A., 2002. Gestão e Gerenciamento De Resíduos.
SNIS, S.N. de I. sobre S., 2017. Diagnóstico do Manejo de Resíduos Sólidos Urbanos - 2015 173.
SNSA - Secretaria Nacional de Saneamento Ambiental, 2016. Diagnóstico do manejo de resíduos sólidos urbanos - 2014. Brasília, DF, Brasil. https://doi.org/10.1017/CBO9781107415324.
Trulli, E., Ferronato, N., Torretta, V., Piscitelli, M., Masi, S., Mancini, I., 2018. Sustainable mechanical biological treatment of solid waste in urbanized areas with low recycling rates. Waste Manage. 71, 556–564. https://doi.org/10.1016/j.wasman.2017.10.018.
Velis, C.A., Longhurst, P.J., Drew, G.H., Smith, R., Pollard, S.J.T., 2009. Biodrying for mechanical-biological treatment of wastes: A review of process science and engineering. Bioresour. Technol. 100, 2747–2761. https://doi.org/10.1016/j.biortech.2008.12.026.
Veloso, S., 2014. BRICS and the challenges of fighting inequality. Rio de Janeiro.
Vergara, S.E., Damgaard, A., Gomez, D., 2016. The efficiency of informality: quantifying greenhouse gas reductions from informal recycling in Bogotá. Colombia. J. Ind. Ecol. 20, 107–119. https://doi.org/10.1111/jiec.12257.
Wilson, D.C., Velis, C.A., Rodic, L., 2013. Integrated sustainable waste management in developing countries. Proc. ICE - Waste Resour. Manage.