CrossMark

Greenhouse gas emissions and cost assessments of municipal solid waste treatment and final disposal in Maputo City

Leticia Sarmento dos Muchangos[1] (iD) · **Akihiro Tokai[1]** ·
Atsuko Hanashima[1,2]

Abstract The goal in this study is to assess the greenhouse gas (GHG) emissions and the costs of municipal solid waste treatment and final disposal alternatives, in Maputo City, capital of Mozambique. Three scenarios were considered: first, the current practice of open dumping; second, the sole usage of sanitary landfilling; and third, comprising material recovery through recycling and biological treatment by composting or anaerobic digestion, and the usage of sanitary landfilling. The highest GHG emissions were observed in the second scenario with a released amount of 260,621 t of CO_2-eq/year, followed by the current scenario with 201,112 t of CO_2-eq/year. Both variants of the third scenario showed negative net emissions of −296,008 t of CO_2-eq/year and −223,925 t of CO_2-eq/year, due to the combination of GHG avoiding alternatives such as composting and biogas production as well as recycling. The cost assessment pointed towards the second scenario being the highest cost-saving scenario, requiring less than US\$ 1 million/year and the current scenario as the most cost-intensive scenario, requiring around US\$ 27 million/year—with a contribution from the cost of inaction of US\$ 24 million/year. In addition, sensitivity analysis was performed by changing the waste compositing in relation to different income levels, which have verified the consistency of results. It also showed that with income increase in the future, GHG emissions will increase in the current and for the second scenario, while for the third scenario, the GHG emissions will reduce and the portion of recyclables will significantly increase.

✉ Leticia Sarmento dos Muchangos
leticia@em.see.eng.osaka-u.ac.jp

Akihiro Tokai
tokai@see.eng.osaka-u.ac.jp

Atsuko Hanashima
hanashima@due.osaka-sandai.ac.jp

[1] Laboratory of Environmental Management, Division of Sustainable Energy and Environmental Engineering, Graduate School of Engineering, Osaka University, Room 510, M3 Building, 2-1 Yamada-oka, Suita City, Osaka Prefecture 565-0871, Japan

[2] Department of Human Life and Environment, Osaka Sangyo University, Daito, Osaka 574-8530, Japan

1 Introduction

The complexity and resource requirement for waste management have been increasing, even in cases where the systems are well-established. In urban areas from developing countries, the current economic development, population growth and expansion of urbanisation result in increasing waste generation rates, overburden on the existing infrastructure and environment, and on the limited budget. For instance, solid waste management (SWM) has the highest budged share spending of up to 50% of the available budget (Achillas et al. 2013; Yadav and Samadder 2014; Scarlat et al. 2015). Furthermore, due to current waste treatment and final disposal practices, methane emissions predictions for urban areas from developing countries show an increase by a factor of 3–4, by 2030 (Matthews 2012). Couth and Trois (2011) also reported that according to the United Nations Statistics Division (UNSD) and the Carbon Dioxide Information Analysis Center (CDIAC), sub-Saharan countries presented a substantial increase in carbon dioxide (CO_2) emissions between 1994 and 2004, ranging between 222 and 307%, and that those numbers will continue to grow due to the population increase and the urban development. Even so, Couth and Trois (2010, p. 2336), further stated that, "the scarce data on carbon emissions from waste management in Africa is likely to represent a high percentage of carbon emissions in urban areas."

Mozambique, a sub-Saharan nation presents an analogous situation as the one described previously. For instance, from 2005 to 2008, it exhibited one of the fastest urbanisation growth rates per year, comparing with other nine key Sub-Saharan Africa markets—Angola, Tanzania, Nigeria, Ghana, Kenya, Zimbabwe, South Africa and Zambia. Moreover, over the past decade, economic growth has been over 8%, and the current growth projections remain higher than the 4–5% forecast by the World Bank (Hedrick-Wong and Angelopulo 2011). Despite the noticeable development, sustainability issues and environmental problems are prevalent, with emphasis on municipal solid waste management (MSWM). In the vast majority of the country's urban areas, the cost of waste collection, treatment and final disposal is intensifying the pressure on the municipal budget, and the continued practice of open dumping is causing significant health and environmental threats to city dwellers as well as being a major source of greenhouse gas (GHG) emissions (Stretz 2012; ISWA et al. 2014). Tas and Belon (2014) reported that according to Carbon Africa estimates for Mozambique, in 2014, emissions from uncontrolled dumpsites reached 76,546 t CO_2-eq and, if the current condition remains the same, this value is expected to increase to 1,369,721 t CO_2-eq in 2030. Despite that, due to the lack of country-specific activity data, especially regarding waste management practices, uncertainties still exist regarding the accurate GHG estimates (Tas and Belon 2014).

The aim of this study is to assess the emissions and associated costs of the current and alternative solid waste treatment and disposal practices in Maputo City, the capital of Mozambique. The specific objectives are (1) to estimate the overall GHG emissions, (2) to estimate the required capital costs, and the operation and maintenance (O&M) costs for the current and alternative waste treatment and final disposal schemes and, in addition, (3) to analyse the effect that changes in waste composition, caused by an increase in income in the future, might have on the GHG emissions.

Maputo City substantially mirrors the situation occurring in the major urban areas of Mozambique—while on the one hand experiences significant economic growth, on the other hand, waste mismanagement represents a major contributor to the incidence of sanitation and urban environment-related problems (Dávila et al. 2008; Maputo Municipal Council 2008; UN-HABITAT 2010).

2 Municipal solid waste management in Maputo City

Maputo City is located in the southern part of Mozambique, enfolded by Maputo Bay, with a population of around 1.2 million inhabitants. Maputo is the administrative centre of Mozambique, reflected by the dominance of the public sector, services, and commerce (Maputo Municipal Council 2008; Stretz 2012).

The MSW generation per capita is distinct according to specific city areas, which are briefly described in Table 1. In the urban area and the sub-urban area surrounding the urban area, the averages are 1.15 and 0.49 kg day^{-1}, respectively. In the semi-urban area (with urban and rural features), MSW generation is around 0.25 kg day^{-1} and in rural areas is around 0.20 kg day^{-1} (Jenkins 2000; Maputo Municipal Council 2008; Stretz 2012).

The main sources of MSW in Maputo City include: wastes generated within the household (including large items); commercial and public institutions; wet markets and fairs; construction and demolition sites; industries (non-hazardous waste); and waste generated through sweeping activities and park and gardens maintenance (dust, sand and green waste). In addition, the waste composition is summarised in Table 2.

In Maputo City, MSW is collected and final disposed in a mixed state. The waste collection system is employed through several methods, according to the collection area's characteristics. Private contractors are the main service provides, co-sharing the responsibility with the municipality. In the urban area, a private contractor provides services, either by collecting plastic bags in the residential areas or by emptying waste containers located in the areas with both residential and commercial buildings. Waste generated in the sub-urban areas is collected in a two-step collection process: locally based small-scale enterprises collect the waste twice a week from households (door-to-door) and transport and deposit it into large containers. Following, the waste is collected by private contractors

Table 1 Different settlement structures in Maputo City. *Source*: Stretz (2012)

Area	Description	Living standard
Urban area	Commercial areas, high buildings, densely populated	Medium to high
	Less population density, mainly residential	
Sub-urban area	"Older" sub-urban neighbourhoods, dense settlements with limited access	Medium to low
Semi-urban area	Broad roads, more space, single houses	
The Island of Inhaca and the municipal district of KaTembe	Rural characteristics	Low

Table 2 Composition of household waste in Maputo City (wt%). *Source*: Adapted from Stretz (2012)

Waste components	Maputo City average (urban + sub-urban area) (%)
Paper and cardboard	6.30
Organic fraction	50.40
Plastic	7.50
Glass	5.15
Metal	2.00
Rags and rubber	2.85
Personal hygiene items (e.g. nappies)	2.30
Other, inert fraction	23.50

and transported to the municipal final disposal site. Additionally, there are scavengers collecting and selling recyclables as well as consuming food waste, picked up from the waste containers. The semi-urban areas are commonly not served in terms of waste collection. In such cases, homeowners resort to burning and/or burying the waste, or feed the animals with the food waste. Lastly, in the Island of Inhaca and the municipal district of KaTembe, collection services are provided by the municipality; however, those are irregular and unstructured, with the usage of unprepared vacant plots for waste final disposal. As for large non-household waste generators (with a daily generation of more than 25 kg or 50 L), waste collection must be arranged by themselves, either by contracting the services of the municipality or by using private contractors (Ferrão 2006; Maputo Municipal Council 2008; Mertanen et al. 2013; dos Muchangos et al. 2014; Tas and Belon 2014).

Following collection in the urban and sub-urban areas, the MSW is transported and final disposed in the city official dumpsite (*Hulene* dumpsite), reportedly in operation since the late 1970s. Hulene dumpsite is a facility sitting in a swamp, with an area of approximately 17 ha, with heights that reach up to 15 m. It is managed by the municipal authority and the disposal operations occur daily, non-stop, with minimal control and compaction, and the daily cover being depended of soil's availability. Upon arrival, the waste trucks are weighted and a disposal fee is charged to private users. Open fires and auto ignition of the waste are common incidences, mostly incited by the more than 500 scavengers collecting recyclables at the dumpsite (Ferrão 2006; Maputo Municipal Council 2008; Segala et al. 2008; Stretz 2012; Tas and Belon 2014).

3 Materials and methods

The estimations for GHG emissions and for cost requirements were largely based in the Life Cycle Assessment or Analysis (LCA). LCA is a decision-support tool/a set of tools, used to quantify the impacts (social, environmental, economic) of product or process life cycle. LCA's studies range from comparative assessments of substitutable products delivering similar functions, to comparative assessment of alternative production processes, including comparing waste management strategies, fact that has been demonstrated to offer valuable inputs to identify appropriate solutions for better management of solid waste (Morrissey and Browne 2004; Laurent et al. 2014; Wilson et al. 2015). LCA is

normalised by ISO 14040 and 14044 documents and can be described by four phases: goal and scope definition, inventory analysis, impact assessment and interpretation (Bjarnadóttir et al. 2002; Yay 2015). LCA for waste management systems in high-income countries usually includes a wide range of impact categories: require detailed knowledge of resource inputs, waste flows and compositions, operational characteristics of facilities and the final destination of recovered materials, energy and residues and the evaluated waste scenarios are likely to be complex and cover all flows starting from the household (Barton et al. 2008). However, those aspects limit its application in developing countries' contexts, where lack of baseline data and site-specific coefficients are usually unavailable (Zurbrügg et al. 2014). Thus, as suggested by Barton et al. (2008), in cases that it is not appropriate to go to a high level of detail or sophistication and for a non-specific overview of options and if the goal is to make an initial assessment of ranking options in terms of GHG emissions, a detailed approach is not warranted or necessary. A few examples of LCA tool application in developing countries context are the following: an evaluation of the environmental performance of liquid household waste is Kumasi, Ghana (Dahlman 2009); an assessment of the current and possible patterns of GHG emissions, based on the case Tianjin, China (Zhao et al. 2011); a comparative analysis of several solid waste management scenarios in Basrah, Iraq (Elagroudy et al. 2011); an identification of the environmental aspects of a less impactful MSWM system in Sakarya, Turkey (Yay 2015); and an analysis of Jordan's MSWM, to identify the most environmentally-friendly and economically-viable alternative to the current situation, among 10 different scenarios (Ikhlayel et al. 2016).

In this study, a simplified LCA approach was applied to provide an initial overview and compare MSW treatment and final disposal options for Maputo City. First, the goal and scope definition clarified the boundaries and basic data sources used in the calculations. For instance, the MSW amounts were collected from a material flow analysis study completed by dos Muchangos et al. (2017), corresponding to the year of 2014; the waste compositions corresponded to the ones presented in Table 2; and the study's results were presented in CO_2-equivalent per year for the GHG emissions, and American dollars ($US) per year for the costs. Next, three scenarios for analysis where selected coupled with the inventory analysis for each process within the scenarios. The first scenario reflected the current MSW treatment and final disposal situation in Maputo City, and the other two scenarios were based on the work of Barton et al. (2008) that proposed a number of options for MSWM focusing on the case in developing countries. Lastly, the GHG emissions results were computed using the "Estimation Tool for Greenhouse Gas from Municipal Solid Waste Management in a Life Cycle Perspective", developed by the Institute for Global Environmental Strategies (IGES),[1] whereas the costs were derived from equations to calculate the total present worth of capital and O&M costs.

3.1 Description of scenarios

The initial scenario (Scenario 1) reflects the business-as-usual in Maputo City. Small MSW portions are either reused or recycled at source (4.3%), recovered through recycling related activities (1.4%) or composted (0.2%). The bulk portion is transported and then disposed of in the official dumpsite (58%), and the remaining portion is either disposed of in smaller dumpsites known or managed by the authorities (1.8%), left uncollected with an unknown

---

[1] GHG calculator for solid waste ver. II-2013. Available in http://enviroscope.iges.or.jp/modules/envirolib/view.php?docid=4273.

final destination (10%) or illegally dumped (25%) (dos Muchangos et al. 2017). Accordingly, the key assumptions for Scenario 1 are the following:

- The official dumpsite is assessed as an unmanaged deep landfill (open dump), receiving waste generated in the urban, sub-urban and semi-urban areas.
- Material recovery and waste reused and recycled at source are considered negligible; thus, the quantities are included in the total amount of MSW final disposed of in Hulene dumpsite.
- The remaining MSW generated in the rural areas is assessed collectively and as being finally disposed of in unmanaged shallow landfills.

The alternative scenarios are the Scenarios 2, 3 (A and B).

In Scenario 2, the consideration is that all MSW is directed to and finally disposed of in a large-scale mixed MSW sanitary landfill.

Scenario 3 contains two variants, with material recovery through recycling and final disposal through landfilling, being the common processes. The main assumptions in this scenario are:

- After generation the waste is sorted at the source, with differentiation between organic, recyclable and non-recyclable materials.
- MSW is collected through a selective collection process: a collection system for recyclable materials, a separate system to collect organic matter and a system to collect the non-recyclables.
- The MSW is directed to an integrated facility with an additional sorting operation, where the recyclables go through a material recovering process, to be prepared for a subsequent recycling process, while the organic fraction is biologically treated either through composting in Scenario 3A, or anaerobic digestion in Scenario 3B.
- The remaining non-recyclable MSW, the rejects, and residues from the treatments and post-treatment processes are directed to a large-scale mixed MSW sanitary landfill.

The depiction of the three scenarios is presented in Fig. 1.

3.2 Assessment methods

As stated previously, the "Estimation Tool for Greenhouse Gas from Municipal Solid Waste Management in a Life Cycle Perspective" was used to estimate the GHG emissions of the waste treatment and final disposal processes. The tool is a simple spreadsheet simulation that helps in estimating GHG emission from waste management methods. The adapted life cycle approach used in the simulation can be applicable to quantify GHG emissions from individual treatment technologies as well as from integrated systems. It includes the analysis of eight main waste handling and treatment options: transportation, mix waste landfilling, composting, anaerobic digestion, Mechanical Biological Treatment (MBT), recycling, incineration and open burning. Once the input of location-specific parameters is concluded, the user is able to assess the results of both direct emissions and GHG savings. Subsequently, the results can support the decision-making process for local governments on the selection of appropriate technologies for GHG mitigation, the evaluation of the progress, and to contribute to a bottom-up approach for national GHG inventory report (Menikpura and Sang-Arun 2013).

Regarding the cost assessment, the total cost per year is the sum of the capital costs and the O&M. Capital costs include costs such as the cost of the land, design, construction and equipment, whereas O&M costs include costs such as for labour, taxes, administration,
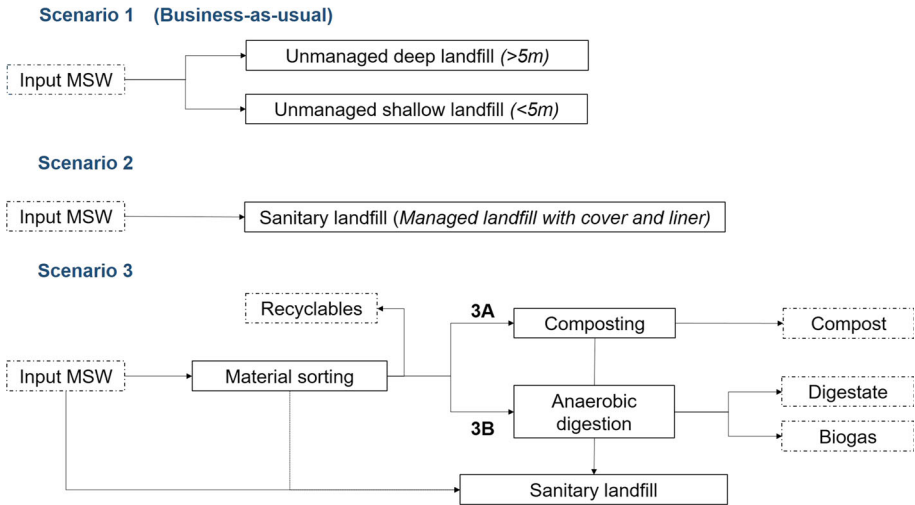
Fig. 1 Diagrams of MSW flow for the assessed scenarios

indirect costs, fuel, electricity and maintenance cost. The total cost per year is then calculated according to Eq. (1) (Nishtala and Solano-mora 1997).

$$\text{Annual cost} = \text{CRF} \times \text{Capital cost} + \text{O\&M cost} \qquad (1)$$

In the equation, the capital recovery factor (CRF)—unit of 1/year enables the conversion of the capital costs into annual terms. CRF is a function of the facility or equipment life (lifetime), and an appropriate interest rate. For a discount rate different than zero, CRF is calculated as shown in Eq. (2) (Nishtala and Solano-mora 1997).

$$\text{CRF} = \frac{\text{Discount rate} \times (1 + \text{Discount rate})^{\text{lifetime}}}{(1 + \text{Discount rate})^{\text{lifetime}} - 1} \qquad (2)$$

The cost assessment is based on the following assumptions:

- An interest rate of 10%.
- A lifetime of waste treatment facilities (for material recovery, composting and anaerobic digestion) of 15 years.
- A lifetime for the sanitary landfill of 20 years.
- A lifetime for the unmanaged landfills of 40 years.
- The cost values are converted to the American dollars, corresponding to the market price of 2017 (Table 3).

Table 3 Applied currency conversion. Values for 2017, *Source*: OANDA Corporation (2017)

Country	Currency	Amount for $1 US dollar
P.R. China	Yuan—CNY	0.1457
European Zone	Euro—EUR	1.0912

A description of the considerations for each treatment and final disposal processes is presented in the following sections, and the applied values are summarised in Table 4.

3.2.1 Anaerobic digestion

The IGES model for this process is based on the *Waste Volume of the 2006 Intergovernmental Panel on Climate Change [IPCC] Guidelines for National Greenhouse Gas Inventories*, with the usage of recommended average default values. To quantify the overall GHG emissions from anaerobic digestion, users are required to input data on the amount of organic waste used in the process, the fossil fuel and electricity necessary for operational activities, the approximate moisture content of the influent and the type of output (electricity or thermal energy) (Menikpura and Sang-Arun 2013).

The final objective of the anaerobic digestion process in Scenario 3 is assumed to be biogas production. The anaerobic digestion will occur in a bioreactor plant with a dry mesophilic process (30–40°C) as operating technology, with an electricity consumption of 0.038 MWh per tonne of input waste and a production of 0.69 t of digestate per tonne of input waste (Bjarnadóttir et al. 2002). Fossil fuel consumption is considered null, and the resulting digestate is used in soil conditioning. Cost estimation is based on a French anaerobic digestion facility with a capacity of 72,000 t, being the capital cost $19.1 (EUR 17) per tonne of input waste and the O&M cost $59.6 (EUR 53) (Hogg and Eunomia Research and Consulting 2002).

3.2.2 Composting

Only methane (CH_4) and nitrous oxide (N_2O) emissions are taken into account for the GHG emissions calculation, because of the biogenic origin of the CO_2 emissions from composting—the emitted CO_2 is regarded to be greenhouse gas neutral (Bjarnadóttir et al. 2002). The average default emission factors recommended by IPCC, used in IGES model are, 4 kg CH_4 per tonne of organic waste in wet basis and 0.3 kg N_2O per tonne of organic waste in wet basis (Menikpura and Sang-Arun 2013). A manual processing string technology is assumed, with 0.5 t of pure compost being generated per tonne of input waste (Bjarnadóttir et al. 2002) and then used for agricultural purposes. Furthermore, the overall fuel consumption demand is 3 L of diesel per tonne of input waste (Barton et al. 2008). The total cost of composting is set as $12.5 per tonne of compost, of which 24% ($3.0) corresponds to capital costs and the remaining 76% ($9.5), corresponds to O&M costs (Dulac 2001).

3.2.3 Mixed MSW landfilling

The sanitary landfill option is adopted in all scenarios as the treatment and ultimate MSW disposal method. For landfilling, the IGES model is also adapted from the 2006 IPCC guidelines, where the first-order decay (FOD) method is strongly suggested, because it reflects the degradation rate of disposal sites more accurately. To calculate the emissions from a landfill or open dump site, data on the total amount of mixed waste landfilled, and the fossil fuel consumption for operational activities, are required. In addition, the selection of the landfill type under analysis is also required, that is: if managed, deep unmanaged (MSW heights > 5 m) or shallow unmanaged (MSW heights < 5 m) landfills (IPCC 2006; Menikpura and Sang-Arun 2013). In Scenario 1, two landfill (open dumps) types are

Table 4 MSW amounts and costs inventory. *Source:* [a]dos Muchangos (2012), [b]Wilson et al. (2015), [c]National Statistics Institute (INE) of Mozambique (2015), [d]Zhao et al. (2011), [e]Pressley et al. (2015), [f]Dulac (2001), [g]Hogg and Eunomia Research and Consulting (2002)

Scenarios	Waste treatment	Input MSW amount (t/year)	Capital cost ($/MSW tonne)	O&M cost ($/MSW tonne)	Cost of inaction—$20 per capita[b]
Scenario 1	Deep unmanaged landfill	278,832	0.0[a]	5.0[b]	Population in the urban, sub-urban and semi-urban areas—1,198,435 in 2014[c]
	Shallow unmanaged landfill	158,498			Population in the Island of Inhaca and the municipal district of KaTembe—27,432 in 2014[c]
Scenario 2	Mixed MSW landfill	437,330	5.2[d]	1.6[d]	Not applicable
Scenario 3A	Recycling	91,621	18.1[e]	6.9[e]	
	Composting	220,414	3.0[f]	9.5[f]	
	Mixed MSW landfill	244,664	5.2[d]	1.6[d]	
Scenario 3B	Recycling	91,621	18.1[e]	6.9[e]	
	Anaerobic digestion	220,414	19.1[g]	59.6[g]	
	Mixed MSW landfill	202,785	5.2[d]	1.6[d]	

considered. In one hand, the deep unmanaged landfill reflects the situation in Maputo City's official dumpsite, where the waste is deposited in a massive open dump with heights that vary from 5 m to 17 m, and, on the other hand, the shallow unmanaged landfill corresponds to the portion of waste deposited in scattered locations around the city, both legally and illegally (dos Muchangos et al. 2017).

Several default values are required, and their accuracy highly influences the result on the amount of methane generation. Those include the degradable organic carbon (DOC), methane generation rate constant (k), methane oxidation on landfill cover (OX) and methane correction factor (MCF) (IPCC 2006; Menikpura and Sang-Arun 2013). The operational activities are assumed to yield a diesel consumption of 1 L per tonne of waste landfilled (Barton et al. 2008).

As for the capital costs for the unmanaged landfills, it was considered to be null. Open dumps are typically unplanned, particularly with respect to siting considerations. Although those are usually government-owned property, they are for the the most part, randomly selected, in a process that lack consideration for standard sanitary landfill sitting criterias (UNEP 2005a). Similarly, final disposal in the official dumpsite of Maputo City started without planning and consideration for future population growth and expansion of settlements in the periphery areas. This unplanned circumstance also apply to the other smaller open dump sites in the city (dos Muchangos et al. 2014).

An additional cost corresponding to the negative externalities of pollution and waste—the cost of inaction—was considered. Within the *Waste Management Outlook* (2015), several examples of the cost of inaction are presented, matching the situation in Maputo City. The cost of inaction is estimated as being at least between the $20 and $50 per capita. It accounts for health impacts, (diarrhoea, gastroenteritis, respiratory diseases and dioxin poisoning, infectious outbreaks and spread of vector-borne diseases, flooding, risks to animals feeding and hazardous substances entering the food chain, and health impacts from uncontrolled hazardous waste disposal), environmental pollution (surface, groundwater and marine contamination, GHG emissions, impacts on fisheries and agriculture, loss of biodiversity and amenity losses to residents) and impacts on tourism (Wilson et al. 2015).

Lastly, the average O&M cost for an open dump is estimated as $5.0 per tonne of MSW (Wilson et al. 2015). The average costs for sanitary landfilling (without landfill gas recovery) in Scenarios 2 and 3 are based on a Chinese landfill plant in Tianjin City, with for $5.2 (CYN 34.5) per tonne of MSW for capital cost and $1.6 (CYN 10.8) per tonne of MSW for O&M cost (Zhao et al. 2011).

3.2.4 Material recovery

The recyclable materials—paper, cardboard, metals, glass, and plastic—were considered to be sorted by a single-stream process, and forwarded to recycling. The single-stream process flow is designed to retrieve fibre, glass, metals, and plastic from a commingled recyclables stream, assuming a separation efficiency of 90% (Pressley et al. 2015). The basis for the inventory data in the IGES tool is Thailand's specific, and emissions are calculated based on CO_2 emissions from fossil fuel and utilisation of electricity to operate machines at the sorting phase. The required input data include the total amount and composition of recyclable materials (Menikpura and Sang-Arun 2013). In addition, the average capital cost of a single-stream material recovery facility (MRF) is $18.1 per tonne of MSW and $6.9 per tonne of MSW for O&M cost (Pressley et al. 2015).

4 Results and discussion

4.1 GHG emissions assessment

The results from total GHG emissions for each scenario are illustrated in Fig. 2, where the worst-case scenarios are Scenario 2 with 260,621 t CO_2-eq per year, followed by the Scenario 1 with 201,112 t CO_2-eq per year (Fig. 3).

Scenario 1 presents 30% fewer emissions than Scenario 2, due to the differences during the waste decomposition process under aerobic conditions, which is directly related to the waste final disposal approach and the facility characteristics. A larger fraction of waste decomposes aerobically in the top layer of unmanaged waste disposal facilities; thus, it is
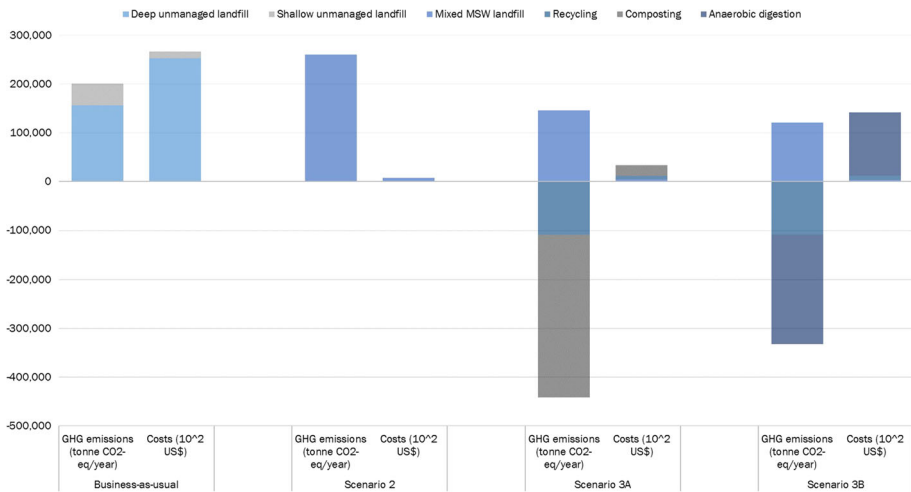
Fig. 2 GHG emissions and cost requirements for each scenario
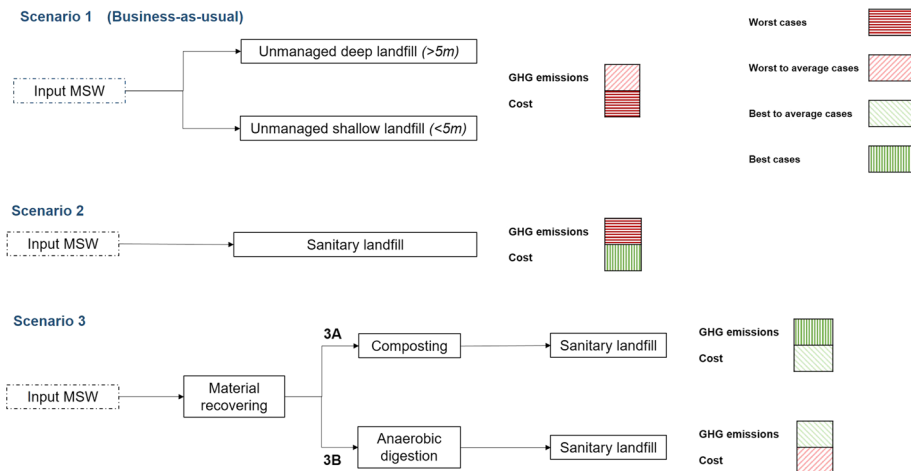
Fig. 3 Overview of the best- and worst-case scenarios

regarded that CH_4 generation is inherently less than in anaerobic managed disposal facilities, such as sanitary landfills. Besides, the same applies in the case of shallow and deep unmanaged facilities, where in deep facilities (and/or facilities with high water table) the fraction of waste that degrades aerobically is smaller than in shallow facilities. Hence, the different MFC default values among those three types of facilities—0.4 for shallow unmanaged landfills, 0.8 for deep unmanaged landfills and 1.0 for anaerobic sanitary landfills (IPCC 2006; UNEP 2005a).

Despite the fact the GHG emissions from Scenario 1 are lesser than those in Scenario 2, the unmanaged landfill (open dump) option is a source of significant GHG emissions, which combined with open burning of waste, releases smoke, particulates, and gaseous contaminants into the atmosphere. Open dumping also causes long-term soil and water (waterways and groundwater reservoirs) contamination, due to leachate and landfill gas (LFG) migration, thus, being considered the most impactful waste treatment disposal method (UNEP 2005b; Mavropoulos et al. 2016). Contrariwise, sanitary landfilling drastically reduces the contact between wastes and the environment, as wastes are concentrated in a well-defined area, enabling appropriate management of gas and leachate emissions as well as limited access of vectors. Moreover, depending on specific conditions and technologies, LFG emissions can further be reduced (UNEP 2005a). For example, ensuring semi-aerated landfill conditions and/or introducing specific LFG management systems to use it as an energy source are some of the alternatives to reduce gas emissions (World Bank 2004; Tanaka et al. 2005). On landfill operation under semi-aerobic conditions, the Fukuoka method is a remarkable type of landfill system that utilises natural decomposition processes under aerobic conditions, to increase microbial activity, resulting in faster stabilisation of waste. In addition, the leachate and gas are constantly removed from the waste mass through leachate collection and gas venting systems. Moreover, it is intrinsically cheaper to operate, comparing with the anaerobic systems (Chong et al. 2005; Tanaka et al. 2005; SPREP and JICA 2010; Ministry of the Environment of Japan 2012). Concerning gas management systems, other than simply collecting and venting the gas, it is also possible to flare it, or recover it for energy use (UNEP 2005a). For example, a proposal by Chang (2004) to develop a stove that runs on a LFG system for Guatemala City's poor demographics, exemplifies the potential of LFG collection and recovery schemes in low-income contexts, to simultaneously address air pollution reduction, natural resource conservation, public health protection, and poverty reduction.

The best-case scenarios are, Scenario 3A with net emissions of $-296,008$ t CO_2-eq per year, and Scenario 3B with net emissions of $-211,603$ t CO_2-eq per year, which is the result from total GHG emissions avoided, via the combined processes included in both scenarios. The main contributors for the negative values of emissions are the biological treatment processes ($-333,287$ t CO_2-eq per year for composting and $-223,925$ t CO_2-eq per year for anaerobic digestion), followed by the material recovery through recycling activities ($-108,528$ t CO_2-eq per year). That is because, compost production yields significant more GHG saving than biogas production, ~ 160 kg of CO_2-eq per year organic waste, and ~ 800 kg of CO_2-eq per year, respectively. The actual emissions are caused by waste final disposal in mixed MSW sanitary landfills, which corresponds to 145,807 t CO_2-eq per year in Scenario 3A, and 120,849 t CO_2-eq per year in Scenario 3B. Here, it is clear that the option of integrating material recovery and biological treatment has potential to yield acceptable results. Yet, there is an identified need for investment in creation of markets, awareness and expertise in developing countries, particularly, for the biological treatment options (Hoornweg and Bhada-Tata 2012). Waste managers must select the technologies, based on factual viability and practical aspects, in order to avoid the

extensively reported failures in such treatment systems (Hazra and Goel 2009; ABRELPE and ISWA 2013; Boonrod et al. 2015).

Furthermore, among all scenarios, the introduction of waste reduction and source separation, and the expansion of material recovery schemes must be emphasised: firstly, by recognising the worth of material recovery, coupled with implementation of direct and simple informative campaigns and studies to secure the understating and adherence of waste generators; combined with the strategies to facilitate the flourishing of recyclable markets, which will finally result in further emission reductions, cost-savings, job creation, public engagement and landfill space conservation (Olar 2003; Zen et al. 2013; Mbiba 2014; Zen et al. 2014; Mavropoulos et al. 2016). Waste reduction and source separation can also benefit the collection and transportation capacity and efficiency (Tai et al. 2011). However, because those activities are dependent on factors such as characteristics of vehicles, fuel consumption and haul distances, which in turn influence GHG emissions (Chen and Lin 2008; Wang et al. 2009), it is imperative to carefully assess and select the least impactful collection and transportation options.

4.2 Cost assessment

Opposite to the GHG emission results, Scenario 2 is the best-case scenario with overall costs under US$ 1 million per year, followed by Scenario 3A, which demands little less than US$ 3.5 million per year. The worst cases are Scenario 3B with about US$ 14 million per year, preceded by Scenario 1 with around US$ 27 million per year.

These results substantiate the premise that open dumps have low initial cost and high long-term cost, while sanitary landfills have an increased initial cost and medium long-term costs (UNEP 2005a). In Maputo City, the share of the cost of inaction from the total cost in Scenario 1 is about US$ 24 million per year. Meaning that if the situation remains the same, it may lead to the rise of several costs related to the environment remediation and public health protection. However, since the available data regarding the cost of inaction per capita is still limited (Wilson et al. 2015), there is a need to conduct a thorough investigation on the precise number and extension of impacts, associated with open dumping.

Concerning the high cost of Scenario 3B, because large-scale or centralised facilities require costly mechanisation and have limited commercial past performance records for MSW (Barton et al. 2008; Matthews 2012), to reduce the resulting high costs, setting-up small-scale and/or localised anaerobic digestion facilities, is an option that can be evaluated.

As all alternative scenarios include sanitary landfilling, for additional cost reduction, the possibility of operating a semi-aerobic facility and/or pursuing LFG recovery, as previously described, cannot be disregarded. While the capital cost might increase, semi-aerobic landfills are cheaper to operate and manage as well as costs can be reduced by usage of the locally available or wasted materials (SPREP and JICA 2010). On the other hand, with the LFG recovery, there is a potential to create revenue through the trade and transfer of emission reduction credits (World Bank 2004; UNEP 2005a).

Lastly, although this assessment did not address the role of informal waste street pickers, itinerant buyers, municipal collection crews and the scavengers in the open dump, who are active in Maputo City MSWM system (Ferrão 2006; Allen and Jossias 2011; Mertanen et al. 2013), the process of deliberating on waste treatment and final disposal alternatives, must include legitimation and integration of those stakeholders, as they minimise capital expenditures, required time and maximise hand power (Olar 2003;

Wilson et al. 2006; Scheinberg et al. 2010). In addition, it has been proven that integration of the informal actors, who are usually part of the urban poor, generates substantial benefits related to social inclusion and secure income creation (Wilson et al. 2006; Dias 2011; Premakumara et al. 2014; Polzer et al. 2015). Also, being that collection and transportation process are commonly cost-intensive activities (Hazra and Goel 2009; Masood et al. 2014), those must also be the object through investigation, to allow the selection of cost-effective solutions.
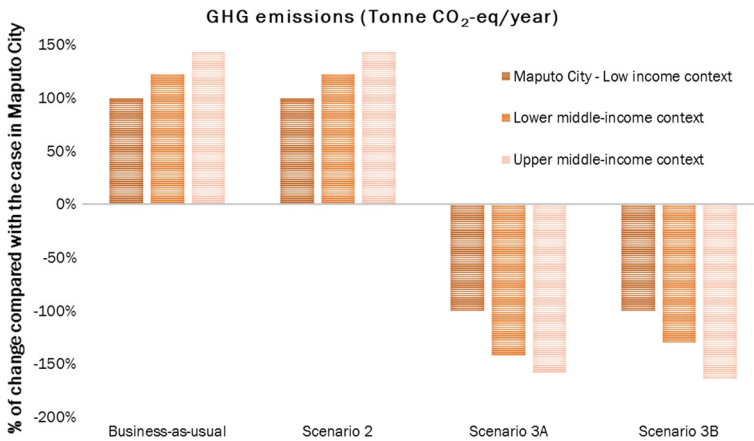
4.3 Sensitivity to waste composition changes

The physical characteristics of waste such as density, moisture content and calorific value are affected by the waste composition, thus, affecting the waste management schemes, i.e., the technology for collection, treatment and the 3Rs (Reduce, Reuse, and Recycle) (Wilson et al. 2015). Since waste composition changes over time, according to the variations in consumption patterns, an analysis is required. However, that is a costly activity for an authority to carry out (Barton et al. 2008). Given that, this section examines the consequences of the changes in waste composition, in relation to the potential future development of the socio-economic context in Maputo City. That is, a change from the current context (low-income), to lower middle income and subsequently to an upper middle income. The average values of waste composition for each income level considered are presented in Table 5.

The sensitivity analysis results are presented in Fig. 4, where the current economic status of Maputo City (low-income) is set at 100% and the other cases are shown relative to it. For the business-as-usual scenario and Scenario 2, there is an increasing trend in total GHG emissions in equal proportion. Increases are due to the lower moisture content (higher fractions of degradable carbon) along the years, mainly caused by the increase in paper and cardboard content, which is reflected in the value of DOC that in the current context is 0.1008, and increasing to 0.1235 for the lower-middle-income and to 0.145 for the upper-middle-income context. Thus, in relation to the low-income context, GHG emissions in the business-as-usual scenario and scenario 2 for a lower-middle-income context are 23% higher, and for the upper middle income are 44% higher. In Scenarios 3A and 3B, the low-income context presents negative GHG emissions values, and in the case of lower-middle-income context, those further decrease 42% in Scenario 3A and 30% in Scenario 3B; for upper-middle-income context, GHG reductions are around 58% in

Table 5 Waste composition for different economic context. *Source*: [a]Stretz (2012), [b]World Bank (2017) and [c]Wilson et al. (2015)

Waste components	Maputo City— low income ($1045 or less)[a]	Lower middle income (more than $1046 less than $4125)[b,c]	Upper middle income (more than $4126 less than $12,735)[b,c]
Paper and cardboard	6	11	19
Organic matter	50	53	46
Plastic	8	9	12
Metal	2	3	4
Glass	5	3	5
Other, residue, inert waste	29	21	14

Fig. 4 Sensitivity to waste composition on GHG emissions

Scenario 3A and 54% Scenario 3B. This decrease is mainly prompted by recycling activities, since the portions of recyclable materials increased in both alternative income levels.

The sensitive analysis showed that the GHG estimations are influenced by the waste composition and the changes are in conformity with results obtained from each scenario. These results highlight the fact that with higher income levels, the possibility for GHG emissions increases if MSW treatment and final disposal remains the same, or in case sanitary landfilling without LFG management is established. However, in case, integrated schemes focused on material recovery are preferred, with the increase in recyclable materials generation, net negative GHG emissions will also significantly increase, thus emphasising the relevance of investing in waste material recovery and sound final disposal, as a way to improve the environmental performance of the system.

5 Conclusions

An environmental and cost assessment of MSW treatment and final disposal alternatives were completed in a case study for Maputo City. The principles of life cycle analysis were adopted to estimate the GHG emissions and the total costs of such alternatives.

Three scenarios were assessed: Scenario 1, which corresponds to the business-as-usual scenario, where the majority of MSW is finally disposed of in open dumps; Scenario 2, with MSW being disposed of in a mixed MSW sanitary landfill, and Scenario 3, with the inclusion of material recovery via recycling, biological treatment (3A—composting or 3B—anaerobic digestion) and of sanitary landfilling as the final disposal method.

The worst-case scenarios concerning GHG emissions were Scenario 2 and 1. Whereas, Scenario 3A and 3B, showed net negative emissions. On the other hand, for Scenario 2, the potential to reduce LFG emissions from sanitary landfilling was acknowledged, providing that it operates in a particular set of conditions to manage and reduce the LFG generated. In the cost assessment, Scenario 2 followed by Scenario 3A presented the best-case alternatives. Results also showed that Scenario 1 represents the worst-case alternative, due to the cost of inaction. Furthermore, due to the high cost associated with the mechanisation of

large-scale or centralised facilities and the limited availability of performance records for anaerobic digestion, Scenario 3B presented the second highest total costs, suggesting the relevance of examining the performance of small-scale or localised anaerobic digestion facilities in developing countries. The results for the sensitivity analysis, considering changes in the waste composition caused by an assumed future increase in the income per capita, suggest that GHG emissions will increase for scenarios 1 and 2. Conversely, in case either Scenario 3A or 3B is in place, the GHG emissions will reduce and the portion of material that can be recycled will considerably increase.

This study provided an overview and comparison of waste treatment and final disposal in options in Maputo City. Nonetheless, there are limitations that should be addressed in future studies. For instance, necessity of expand the boundaries to include the assessment of the impacts due to handling and processing activities at the source, the contribution of transportation processes, and the evaluation of the impacts deriving from the integration of the informal sector.

References

Achillas, C., Moussiopoulos, N., Karagiannidis, A., Banias, G., & Perkoulidis, G. (2013). The use of multi-criteria decision analysis to tackle waste management problems: A literature review. *Waste Management and Research, 31*(2), 115–129.

Allen, C., & Jossias, E. (2011). *WIEGO organizing brief no.6: Mapping of the policy context and Catadores Organizations in Maputo, Mozambique*. Manchester: Women in Informal Employment: Globalizing and Organizing (WIEGO).

Associação Brasileira de Empresas de Limpeza Pública e Resíduos Especiais (ABRELPE), & International Solid Waste Association (ISWA). (2013). *Solid waste: Guidelines for successful planning*. Sao Paulo: ABRELPE & ISWA.

Barton, J. R., Issaias, I., & Stentiford, E. I. (2008). Carbon-making the right choice for waste management in developing countries. *Waste Management, 28*(4), 690–698.

Bjarnadóttir, H. J., Friðriksson, G. B., Johnsen, T., & Sletsen, H. (2002). *Guidelines for the use of LCA in the waste management sector*. Espoo. http://www.nordtest.org/register/techn/tlibrary/tec517/tec517.pdf

Boonrod, K., Towprayoon, S., Bonnet, S., & Tripetchkul, S. (2015). Enhancing organic waste separation at the source behavior: A case study of the application of motivation mechanisms in communities in Thailand. *Resources, Conservation and Recycling, 95,* 77–90.

Chang, J. (2004). Guatemala city landfill gas burning stove. http://www.its.caltech.edu/~kpickar/e105/papers/papers.htm

Chen, T.-C., & Lin, C.-F. (2008). Greenhouse gases emissions from waste management practices using Life Cycle Inventory model. *Journal of Hazardous Materials, 155*(1–2), 23–31.

Chong, T. L., Matsufuji, Y., & Hassan, M. N. (2005). Implementation of the semi-aerobic landfill system (Fukuoka method) in developing countries: A Malaysia cost analysis. *Waste Management, 25*(7), 702–711.

Couth, R., & Trois, C. (2010). Carbon emissions reduction strategies in Africa from improved waste management: A review. *Waste Management, 30*(11), 2336–2346.

Couth, R., & Trois, C. (2011). Waste management activities and carbon emissions in Africa. *Waste Management, 31*(1), 131–137.

Dahlman, K. (2009). *Modeling sanitation scenarios in developing countries. A case study in Kumasi, Ghana*. Swedish University of Agricultural Sciences. Retrieved from http://www.w-program.nu/filer/exjobb/Kristina_Dahlman.pdf

Dávila, J. D., Kyrou, E., Nuñez, T., & Sumich, J. (2008). *Urbanisation and municipal development in Mozambique: Urban poverty and rural-urban linkages*. London. http://siteresources.worldbank.org/INTDEBTDEPT/Resources/468980-1218567884549/5289593-1224797529767/Mozambique200808_01.pdf

Dias, S. (2011). Recycling in Belo Horizonte, Brazil—An overview of inclusive programming recycling of materials from domestic waste. *WIEGO Policy Brief (Urban Policies), 1*(3), 1–8.

dos Muchangos, L. S. (2012). *A study of MSWM system in Maputo City, Mozambique—Learning experiences of Chongqing, China*. Unpublished master's thesis. Chongqing University, Chongqing.

dos Muchangos, L. S., Liu, Y., & Li, B. (2014). Comparative study on municipal solid waste management systems of Maputo City, Mozambique and Chongqing City, China. *African Journal of Science, Technology, Innovation and Development, 6*(4), 323–331.

dos Muchangos, L. S., Tokai, A., & Hanashima, A. (2017). Application of material flow analysis to municipal solid waste in Maputo City, Mozambique. *Waste Management and Research, 35*(3), 253–266.

Dulac, N. (2001). The organic waste flow in integrated sustainable waste management. In A. Scheinberg, (Ed.) *Tools for decision-makers, experiences from the urban waste expertise programme (1995–2001), Gouda*.

Elagroudy, S., Elkady, T., & Ghobrial, F. (2011). Comparative cost benefit analysis of different solid waste management scenarios in Basrah, Iraq. *Journal of Environmental Protection, 2*(5), 555–563.

Ferrão, D. A. G. (2006). *Avaliação da Remoção e Eliminação dos Resíduos Sólidos na Cidade de Maputo*. Moçambique: University of Cape Town.

Hazra, T., & Goel, S. (2009). Solid waste management in Kolkata, India: Practices and challenges. *Waste Management, 29*(1), 470–478.

Hedrick-Wong, Y., & Angelopulo, G. (2011). *The challenges of urbanization in sub-Saharan Africa: A tale of three cities*. MasterCard Worldwide.

Hogg, D., & Eunomia Research and Consulting. (2002). *Costs for municipal waste management in the EU. Final report to Directorate General Environment, European Commission*. http://ec.europa.eu/environment/waste/studies/eucostwaste_management.htm

Hoornweg, D., & Bhada-Tata, P. (2012). *What a waste*. World Bank: A global review of solid waste management.

Ikhlayel, M., Higano, Y., Yabar, H., & Mizunoya, T. (2016). Introducing an integrated municipal solid waste management system: Assessment in Jordan. *Journal of Sustainable Development, 9*(2), 43.

International Solid Waste Association (ISWA) (2014), Waste-to-energy research and technology council (WTERT), Regional Solid Waste Exchange of Information and Expertise Network (SWEEPNET), University of Leeds, and Solid Waste Management in Asia and Pacific Islands (SWAPI). (2014). *Waste Atlas. The world's 50 biggest dumpsites*. 2014 Report.

Jenkins, P. (2000). City profile: Maputo. *Cities, 17*(3), 207–218.

Laurent, A., Bakas, I., Clavreul, J., Bernstad, A., Niero, M., Gentil, E., et al. (2014). Review of LCA studies of solid waste management systems—Part I: Lessons learned and perspectives. *Waste Management, 34*(3), 573–588.

Maputo Municipal Council. (2008). *Plano Director—Gestão de resíduos sólidos urbanos na Cidade de Maputo (Master plan of municipal solid waste management in Maputo City)*. Maputo: Conselho Municipal da Cidade de Maputo. **(in Portuguese)**.

Masood, M., Barlow, C. Y., & Wilson, D. C. (2014). An assessment of the current municipal solid waste management system in Lahore, Pakistan. *Waste Management and Research, 32*(9), 834–847.

Matthews, E. (2012). *Good news from the dump methane emissions from solid waste: Current conditions and future prospects*. Boston: Clean Air Task Force.

Mavropoulos, A., Cohen, P., Greedy, D., Plimakis, S., Marinheiro, L., Law, J., & Loureiro, A. (2016). *A roadmap for closing waste dumpsites the world's most polluted places*. Vienna: ISWA.

Mbiba, B. (2014). Urban solid waste characteristics and household appetite for separation at source in Eastern and Southern Africa. *Habitat International, 43,* 1–11.

Menikpura, N., & Sang-Arun, J. (2013). *User manual estimation tool for greenhouse gas (GHG) emissions from municipal solid waste (MSW) management in a life cycle perspective*.

Mertanen, S. T., Langa, J. M., & Ferrari, K. (2013). *Catadores de Lixo de Maputo. Quem sao e como trabalham.*

Ministry of the Environment of Japan. (2012). *Solid waste management and recycling technology of Japan—Toward a sustainable society*. Tokyo.

Morrissey, A. J., & Browne, J. (2004). Waste management models and their application to sustainable waste management. *Waste Management, 24*(3), 297–308.

National Statistics Institute. (2015). *Population projections for 2007 to 2014*. Government of Mozambique: http://www.ine.gov.mz/estatisticas/esta-tisticas-demograficas-e-indicadores-sociais/projeccoes-da-populacao/populacao-projectada-por-distritos-maputo-cidade-2007_2040.xls/view. Accessed June 13, 2015.

Nishtala, S., & Solano-mora, E. (1997). *Description of the material recovery facilities process model design, cost, and life-cycle inventory*. Research Triangle Institute, North Carolina State University.

Olar, Z. (2003). *Urban solid waste management: Waste reduction in developing nations*. Houghton, MI: Michigan Technological University.

OANDA Corporation. (2017). Historial rates. http://www.oanda.com/currency/historical-rates/. Accessed August 10, 2017.

Polzer, V. R., & Pisani, M. A. J. (2015). The challenge of solid waste collection in precarious settlements in São Paulo. *Brazil, 1,* 1–9. doi:10.5348/C01-2015-1-RA-1.

Premakumara, D. G. J., Canete, A. M. L., Nagaishi, M., & Kurniawan, T. A. (2014). Policy implementation of the Republic Act (RA) No. 9003 in the Philippines: A case study of Cebu City. *Waste Management, 34*(6), 971–979.

Pressley, P. N., Levis, J. W., Damgaard, A., Barlaz, M. A., & DeCarolis, J. F. (2015). Analysis of material recovery facilities for use in life-cycle assessment. *Waste Management, 35,* 307–317. doi:10.1016/j.wasman.2014.09.012.

Scarlat, N., Motola, V., Dallemand, J. F., Monforti-Ferrario, F., & Mofor, L. (2015). Evaluation of energy potential of municipal solid waste from African urban areas. *Renewable and Sustainable Energy Reviews, 50,* 1269–1286.

Scheinberg, A., Simpson, M., Gupt, Y., Anschütz, J., Haenen, I., Tasheva, E., et al. (2010). *Economic aspects of the informal sector in solid waste management*. Eschborn: GTZ (German Technical Cooperation).

Secretariat of the Pacific Regional Environment Programme (SPREP), & Japan International Cooperation Agency (JICA). (2010). *A Practical Guide to Landfill Management in Pacific Island Countries and Territories*. Apia: SPREP.

Segala, K., Opressa, I., & Palalane, J. (2008). Gestão de Resíduos Sólidos. *Urbanização e Desenvolvimento Municipal em Moçambique*. Rio de Janeiro, Brazil.

Stretz, J. (2012). Economic Instruments in Solid Waste Management. In E. Gunsilius & GIZ (Eds.), *Case Study Maputo, Mozambique*. Maputo: Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) GmbH Concepts for Sustainable Waste Management.

Tai, J., Zhang, W., Che, Y., & Feng, D. (2011). Municipal solid waste source-separated collection in China: A comparative analysis. *Waste Management, 31*(8), 1673–1682.

Tanaka, N., Tojo, Y., & Matsuto, T. (2005). Past, present, and future of MSW landfills in Japan. *Journal of Material Cycles and Waste Management, 7*(2), 104–111.

Tas, A., & Belon, A. (2014). *A Comprehensive review of the municipal solid waste sector in Mozambique: Background documentation for the formulation of nationally appropriate mitigation actions in the waste sector in Mozambique*. Nairobi, Kenya; Maputo. www.carbonafrica.co.ke

The Intergovernmental Panel on Climate Change (IPCC). (2006). In S. Eggleston, L. Buendia, K. Miwa, T. Ngara, & K. Tanabe (Eds.) *2006 IPCC guidelines for national greenhouse gas inventories, waste*. (Vol. 5). Hayama: Institute for Global Environmental Strategies (IGES) on behalf of the IPCC. http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol5.html

United Nations Environment Programme (UNEP). (2005a). *Training module: Closing an open dumpsite and shifting from open dumping to controlled dumping and to sanitary land filling*. Osaka/Shiga: United Nations Environment Programme.

United Nations Environment Programme (UNEP). (2005b). *Solid Waste Management*. Osaka/Shiga: United Nations Environment Programme.

United Nations Human Settlements Programme (UN-HABITAT). (2010). *Mozambique Cities Profile. Maputo, Nacala and Manica*. Nairobi: United Nations Human Settlements Programme.

Wang, L., Hu, G., Gong, X., & Bao, L. (2009). Emission reductions potential for energy from municipal solid waste incineration in Chongqing. *Renewable Energy, 34*(9), 2074–2079.

Wilson, D., Rodic, L., Modak, P., Soos, R., Rogero, A. C., Velis, C., et al. (2015). In T. Cannon (Ed.), *Global waste management outlook*. United Nations Environment Programme (UNEP). http://www.unep.org/ietc/Portals/136/Publications/Waste-Management/GWMO-report/GWMO-full-report.pdf

Wilson, D. C., Velis, C., & Cheeseman, C. (2006). Role of informal sector recycling in waste management in developing countries. *Habitat International, 30*(4), 797–808.

World Bank. (2004). *Handbook for the Preparation of Landfill Gas to Energy Projects in Latin America and the Caribbean The World Bank—ESMAP* (Vol. 19399). Waterloo, Ontario: The International Bank for Reconstruction and Development/The World Bank.

World Bank. (2017). *World Bank Country and lending groups.* https://datahelpdesk.worldbank.org/knowledgebase/articles/906519-world-bank-country-and-lending-groups. Accessed January 19, 2017

Yadav, P., & Samadder, S. R. (2014). Life cycle assessment of solid waste management options: A review. *Recent Research in Science and Technology, 6*(1), 113–116.

Yay, A. S. E. (2015). Application of life cycle assessment (LCA) for municipal solid waste management: A case study of Sakarya. *Journal of Cleaner Production, 94,* 284–293.

Zen, I. S., Ahamad, R., & Omar, W. (2013). No plastic bag campaign day in Malaysia and the policy implication. *Environment, Development and Sustainability, 15*(5), 1259–1269.

Zen, I. S., Noor, Z. Z., & Yusuf, R. O. (2014). The profiles of household solid waste recyclers and non-recyclers in Kuala Lumpur, Malaysia. *Habitat International, 42,* 83–89.

Zhao, W., Huppes, G., & van der Voet, E. (2011). Eco-efficiency for greenhouse gas emissions mitigation of municipal solid waste management: A case study of Tianjin, China. *Waste Management, 31*(6), 1407–1415.

Zurbrügg, C., Caniato, M., & Vaccari, M. (2014). How assessment methods can support solid waste management in developing countries—A critical review. *Sustainability, 6,* 545–570.