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RESEARCH ARTICLE

Environmental impact assessment of municipal solid waste management options using life cycle assessment: a case study

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Abstract The goal of this study is to use life cycle assessment (LCA) tool to assess possible environmental impacts of different municipal solid waste management (MSWM) scenarios on various impact categories for the study area Dhanbad City, India. The scenarios included in the present study are collection and transportation (denoted as S1); baseline scenario consisting of recycling, open burning, open dumping, and finally unsanitary landfilling without energy recovery (denoted by S2); composting and landfilling (denoted by S3); and recycling and composting followed by landfilling of inert waste without energy recovery (denoted by S4). One ton of municipal solid waste (MSW) was selected as the functional unit. The primary data were collected through sampling, surveys, and literatures. Background data were obtained from Eco-invent data of SimaPro 8.1 libraries. The scenarios were compared using the CML 2 baseline 2000 method, and the results indicated that the scenario S1 had the highest impact on marine aquatic ecotoxicity (1.86E + 04 kg 1,4-DB eq.) and abiotic depletion (2.09E + 02 kg Sb eq.). S2 had the highest impact on global warming potential (9.42E + 03 kg CO_2 eq.), acidification (1.15E + 01 kg SO_2 eq.), eutrophication (2.63E + 00 kg PO_4^{3-} eq.), photochemical oxidation (2.12E + 00 kg C_2H_4 eq.), and human toxicity (2.25E + 01 kg 1,4-DB eq.). However, S3 had the highest impact on abiotic depletion (fossil fuels) (2.71E + 02 MJ), fresh water aquatic ecotoxicity (6.54E + 00 kg 1,4-DB eq.), terrestrial ecotoxicity (3.36E − 02 kg 1,4-DB eq.), and ozone layer depletion (2.73E − 06 kg CFC-11 eq.). But S4 did not have the highest impact on any of the environmental impact categories due to recycling of packaging waste and landfilling of inert waste. Landfilling without energy recovery of mixed solid waste was found as the worst disposal alternative. The scenario S4 was found as the most environmentally suitable technology for the study area and recommended that S4 should be considered for strategic planning of MSWM for the study area.

Keywords Collection · Composting · Landfilling · Life cycle assessment · Open burning · Open dumping · Recycling · Transportation

Introduction

India is the second most populated country of the world. It consists of 29 states and 7 union territories, covering 640 districts, 5767 tehsils, 7933 towns, and more than 0.6 million villages (Census 2011, Govt. of India). The cities with population of 4 million and above are recognized as mega cities in India. At present, there are total eight megacities in India. Moreover, 46 cities have population more than 1 million, 388 cities with population in between 100,000 and 1 million, and 2493 cities have a population less than 100,000 (POC 2017). The population growth rate in India is approximately 1.3% (Census 2011, Govt. of India). The reason for high population growth in metro and mega cities may be due to rapid industrialization and migration of people from villages to the cities in search of better job prospects which has resulted in considerable increase in the municipal solid waste (MSW)

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generation rate (Sharma and Chandel 2017; Yadav and Samadder 2017a). With increase in MSW generation rate, the waste collection and disposal efficiency should also increase proportionately in these cities. But MSW collection efficiency in developing countries is only 41%, whereas for the developed countries the waste collection efficiency is around 90% (World Bank 2012). Uncontrolled open dumping, open burning, and unsanitary landfilling of MSW are common waste disposal practices in developing countries like India, Indonesia, Nepal, Bangladesh, Cameroon, Pakistan, Sri Lanka, etc. (Table 1) (Yadav and Samadder 2017b). Open dumping, open burning, and landfilling may lead to several environmental problems along with toxicity to human and animal health (UNEP 2011). The MSW disposal option with minimum environmental impact is one of the biggest developmental challenges for India and other developing countries.

Life cycle assessment (LCA) is one of the most widely used and accepted techniques for quantification of the environmental impacts of MSW management (MSWM) options (Miliute and Kazimieras Staniskis 2010). LCA methodology is widely used in European and Asian countries such as Italy (Buttol et al. 2007), Turkey (Banar et al. 2009; Yay 2015), China (Hong et al. 2006 and Zhao et al. 2009), Malaysia (Saheri et al. 2012), Thailand (Menikpura et al. 2013), Singapore (Tan and Khoo 2006), etc. In Austria, Beigl and Salhofer (2004) compared three scenarios of MSWM such as the recycling options using bring collection system, kerbside collection system with recycling, and non-recycling. Kerbside collection was environmentally better than the bring collection system, because the specific fuel consumption was lower for collection and transport in kerbside collection. Tan and Khoo (2006) reported recycling and biological treatments of MSW as better MSWM options than thermal treatments, and landfilling was the worst among all the options studied by them. Hong et al. (2006) found similar results, although recycling was included in each scenario. Zhao et al. (2009), Tarantini et al. (2009), and Karoneas and Nanaki (2012) worked on the LCA of MSWM and their results also showed recycling as the better waste management option due to the requirement of less natural resources and less environmental impact. Banar et al. (2009) compared five different scenarios (collection/transportation, material recovery facility (MRF)/recycling, composting, incineration, and landfilling) and concluded that composting was the most environmentally preferable MSWM option. Mendes et al. (2004), Hong et al. (2006), and Abduli et al. (2011) also reported studies of LCA on MSWM and considered composting and mechanical-biological treatment (MBT) and other management options. Their findings suggested composting and MBT as the better waste management options than incineration, landfilling, and others. However, composting was less preferable than source reduction of MSW. Ozeler et al. (2006), Zaman (2010), and Dong et al. (2014) compared landfilling with other MSWM options and concluded that landfilling had the highest global warming potential (GWP) than other waste management options due to higher methane and carbon dioxide emissions. Srivastava and Nema (2011) conducted a study on LCA of integrated solid waste management options in Delhi, India, to analyze the environmental impacts due to recycling, composting, incineration, and landfilling of MSW and found that recycling had minimum environmental impact among all the MSWM options. Srivastava and Nema (2011) also predicted the quantity and composition of solid waste generation in Delhi till the year 2024. Bohra et al. (2012) conducted another LCA study in Delhi City and compared the 12 different MSW treatment scenarios. These scenarios were designed by changing the percentage of the waste to be handled by incineration, refuse-derived fuel, biomethanation, composting, and sanitary landfill. The results showed that the scenario with maximum diversion from sanitary landfill had least GWP. Babu et al. (2014) conducted a LCA study on MSW of

Table 1 Municipal solid waste management practices in some of the developing countries

Countries	Waste generation (kg/c/day)	Composting (%)	Open dumping (%)	Unsanitary landfilling (%)	Recycling (%)	Incineration or WTE (%)	Other (%)
Malaysia	1.52	1	–	34.10	5.50	–	59.40
Indonesia	0.52	–	40.09	–	1.61	7.54	15.27
Iran	0.16	10	–	84	6	–	–
India	0.30–0.60	10	60	15	–	5	10
South Korea	1.24	–	–	36	49	14	–
Nepal	0.32	–	70	–	–	–	30
Pakistan	0.65	5	80	5	–	–	10
Sri Lanka	0.20–0.90	5	85	–	–	–	10
Philippines	0.50	–	–	–	–	–	–
Bangladesh	0.41	–	50	–	15	–	35

Source: (Yadav and Samadder 2017b)

Bangalore City, in which four scenarios (open dumps, landfill without gas recovery, landfill with gas recovery, and bioreactor landfill) were compared and found that the bioreactor landfill option was the best among all the scenarios. In Mumbai City of India, Sharma and Chandel (2017) compared seven scenarios, such as open dumping and bioreactor landfill; MRF and sanitary landfill; MRF, composting and sanitary landfill; MRF, anaerobic digestion, and sanitary landfill; MRF, composting, anaerobic digestion, and sanitary landfill; MRF, composting and incineration and MRF and incineration. The study found that MRF, composting, anaerobic digestion, and sanitary landfill had the lowest environmental impact. All these studies were done in the metro cities (Delhi, Bangalore, and Mumbai) of India, but there is not a single LCA study present for non-metro cities of India. The present study has been carried out in Dhanbad (Fig. 1), a non-metro city of India. This is a representative study of many small cities of India and other developing countries that are facing problems related to MSWM (Fig. 2a). Furthermore, previous studies showed that environmental impacts of different technologies would vary from one county to another and also from one city to another due to differences in waste composition as well as environmental conditions.

The important aspects of MSWM planning are identification of the suitable waste management system that can reduce volume of waste and the environmental impacts (Banar et al. 2009). In developing countries, the MSWM strategies should aim for minimization of the final amount and volume of waste intended for landfilling and to reduce the pollution caused due to MSW treatment, collection, and transportation activities (Cherubini et al. 2008). Actually, there is no single MSW treatment option which is the most suitable for all waste fractions in different topographical regions (Liamsanguan and Gheewala 2008).

At present Dhanbad City is facing problems due to the continuous increase in MSW generation and limited resources for MSWM. Also, no scientific waste management practices are followed in the study area (Fig. 2b) for waste collection, transportation, and disposal process. The MSW are currently disposed in two unsanitary landfill sites located in the study area (Baniya heer and Matkuria).

The present study quantified the environmental impacts due to open dumping, open burning, and unsanitary landfilling in the study area. These are the hazardous MSWM practices in several developing and under developed countries (Ray et al. 2005). The present study focused on the management of overall MSW of the study area. Bio-waste and inert waste constitutes 68.82% in the study area. Windrow composting and inert waste management were not considered in previous LCA studies. The aim of this study is to use the

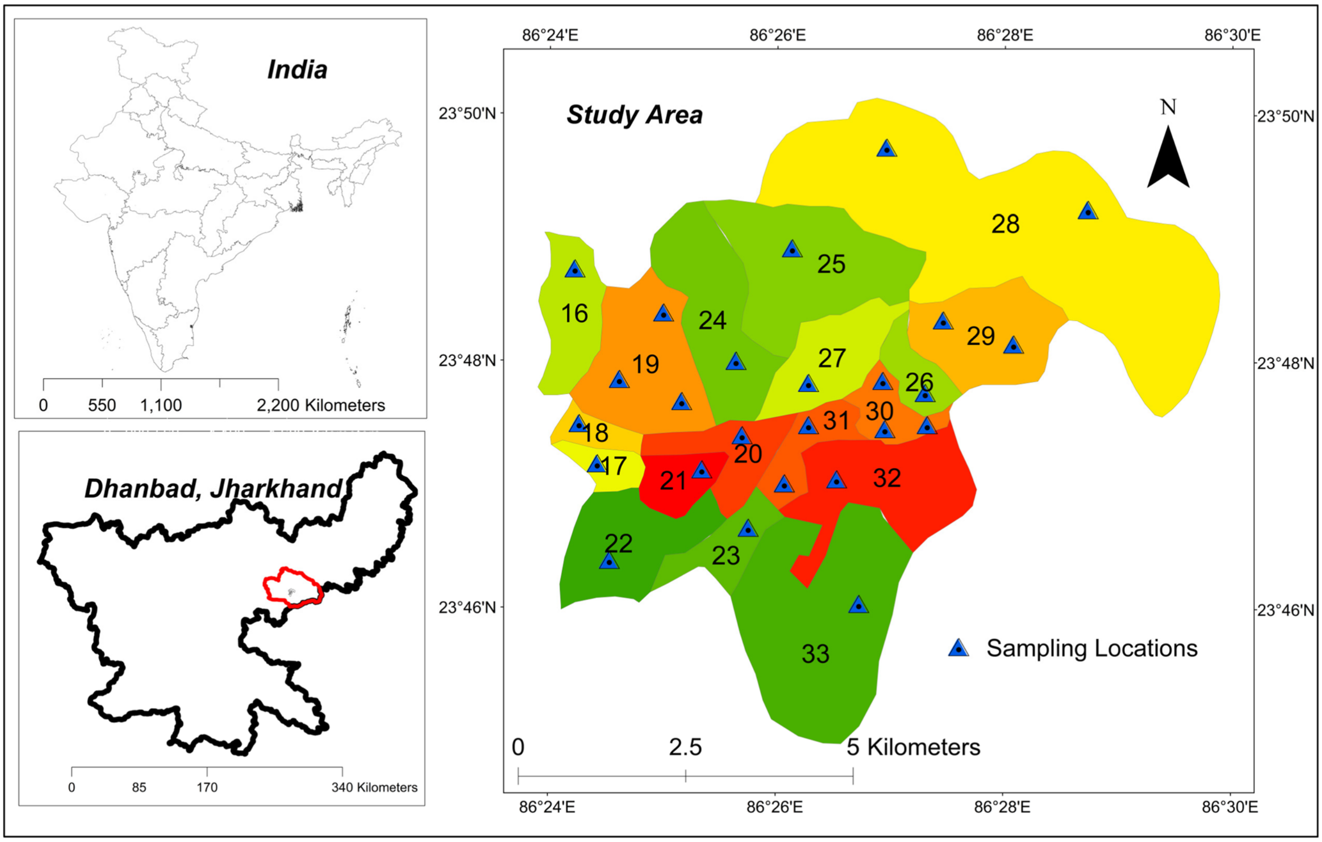

Fig. 1 Location of the study area

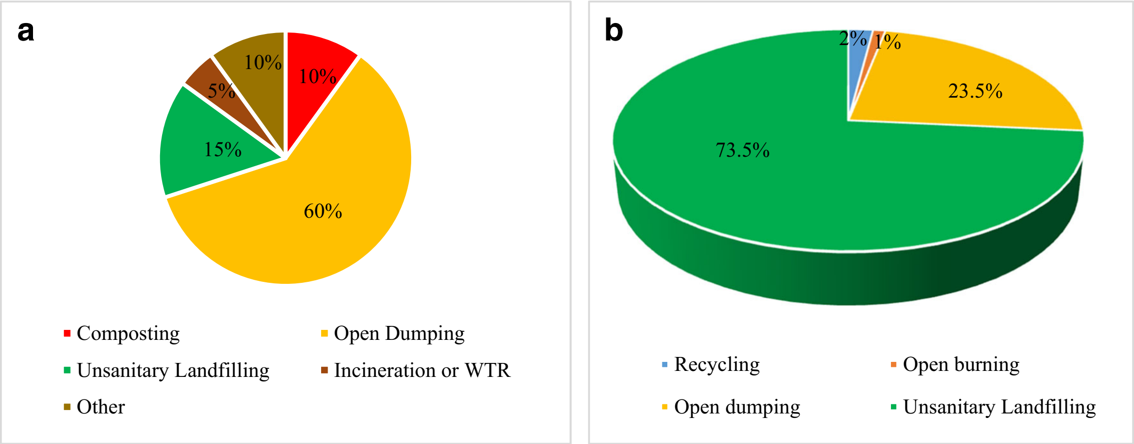

Fig. 2 Percentage distribution of MSW disposal practices **a** in India, **b** in the study area

LCA tool to compare and assess the environmental impacts of four different MSWM scenarios and determine the most feasible management scenario (based on minimum environmental impact in all the impact categories) for the study area. Another objective of the study was to quantify the impacts of existing MSWM scenario of study area. The findings of the present study can be applied to the other cities of India and the world with similar social and climatic conditions.

Materials and methods

Study area

The present study area comes under Dhanbad District, which is known as the coal capital of the India due to the presence of one of the largest coal reserves in India. Dhanbad has an average elevation of 227 m (745 ft.) above mean sea level and comes under the Chota Nagpur Plateau. The study area is located between 85°45′ E to 86°30′ E longitudes and between 23°32′ N to 24°5′ N latitudes. According to the Census 2011, Dhanbad District has a population of more than 1 million. The present study area covers the main municipal region of the city within the Dhanbad Municipal Corporation (DMC), consisting of 18 administrative wards as shown in Fig. 1. Total area of the study area is about 120 km^2 and supports a population of about 0.36 million as per the 2011 Census of India. The population generates approximately 147.06 ton/day of MSW with an average per capita waste generation rate of 0.41 kg/c/day.

MSW characterization

At the beginning of this study, a survey was conducted in various administrative wards of the city (within the jurisdiction of DMC) to get primary data related to the number of households, demography, population, education level, number of shops, markets, and bin locations. The primary data was used for the categorization of the administrative wards into different zones such as residential area, commercial area (shops, markets, hotels, and restaurants), and other areas (dumping sites). Based on the survey, it was decided that 100 solid waste samples in each season (summer, winter, autumn, and monsoon) (thus total 400 samples) are sufficient for representing the whole study area. For sample collection, 25 different sampling locations were identified that can represent the overall population of the study area. The sampling procedure for solid waste collection from the residential area which covers all the socioeconomic groups of the study area was followed as recommended by Khan et al. (2016); and for the commercial areas was followed as recommended by Sharma and McBean (2007), and Al-Khatib et al. (2010). The solid waste samples were collected from the dumping sites following the procedure mentioned in MSW manual, Government of India CPHEEO (2016). First of all, the contents of individual waste collection bin (5 m^3) were emptied and thoroughly mixed at the sampling location itself. Then, 10 kg of representative solid wastes samples was taken out from the thoroughly mixed waste using quadrate method. The representative waste samples (10 kg) were placed on a plastic sheet, then the wastes were segregated manually into different components and categorized into recyclable, compostable, incinerable and inert wastes on the basis of their compositions as defined in previous literature (CPHEEO 2016) and presented in Fig. 3a, b. After the physical characterization, the wastes were again mixed thoroughly with a trowel and a cone shape was made using the solid waste (Peavy et al. 1985; Tchobanoglous et al. 1993). The sample cone was then divided into four parts and discarded two opposite slices of the cone. The remaining two parts of the cone were mixed thoroughly and again a similar cone was made. This procedure was repeated until the 10 kg of solid wastes

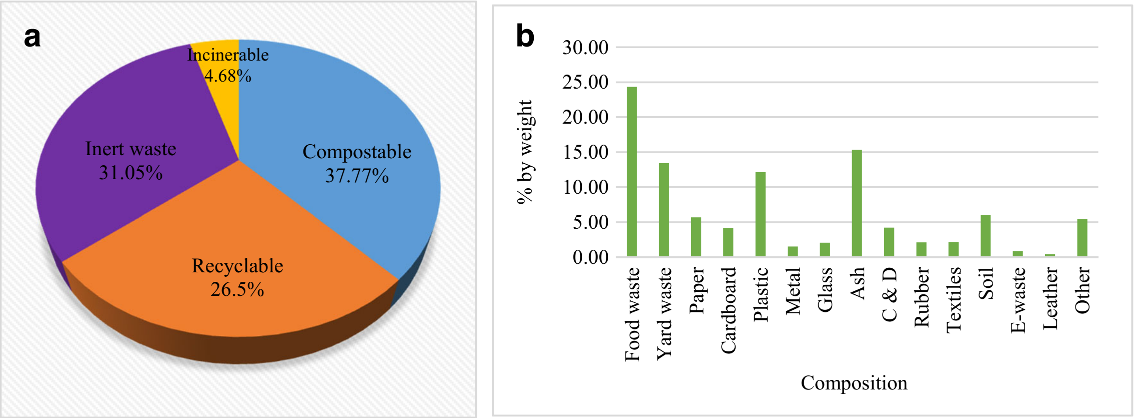

Fig. 3 **a** Percentage distribution of the different MSW categories. **b** Percentage composition of MSW in the study area

sample was reduced to 1.25 kg. The reduced sample size (1.25 kg) was brought to the laboratory for further analysis. The same procedure was repeated for all the samples collected from 25 sampling locations. Thus, the total sample size considered in this study for physical characterization was 4 ton (four seasons), which was further reduced to 500 kg (125 kg in each season) for laboratory analysis. Sixty samples were collected from the residential area (wards 16, 17, 18, 19, 23, 24, 25, 26, 27, 28, 29, 30, 31, 32, and 33); 28 samples were taken from commercial area (wards 20, 21, 22, 28, 29, 30, and 31); and 12 samples were taken from dumping sites (wards 19 and 30, respectively) (Fig. 1). MSW classification was done as per ASTM D5231-92 (2003). The components of MSW along with their composition (in % by weight) is shown in Fig. 3b.

Goal and scope of the study

The goal of the present study is to quantify and compare the environmental impacts of four different MSWM scenarios and to select the best waste management option based on minimum emission. The first scenario that was selected in the present study was collection and transportation that is an important component of MSWM. The second scenario was the existing MSWM practices of study area. Third scenario was selected on the basis of composition, moisture content, and physico-chemical properties of MSW. Selection of fourth scenario was dependent on the fractions of MSW, physico-chemical properties of waste, and the MSWM hierarchy. The selected functional unit was 1 ton for comparison of the four different scenarios of MSWM in the study area.

System boundary

System boundary is an essential part of LCA methodology, the emission levels from different waste management options are dependent on system boundary setting. The system boundary created for the present study is an attributional system boundary. As shown in Fig. 4, the system boundary covers all the inputs from natural resources and techno sphere as well as the outputs into the air, water, soil, and residues or product into the nature. It starts from the collection of MSW and ends with the landfilling of residual materials. The intermediate phases are transportation, base line scenario (Fig. 5) (open dumping and open burning), recycling, and composting.

Scenario 1 (S1): collection and transportation

In this study, scenario 1 represents the existing collection and transportation part of MSWM in study area. The waste collection efficiency during the study period was 73.5% in the study area and rest of the waste remained uncollected. The daily collected waste from community dustbins (108 ton) are transported to the nearby disposal landfill site (Fig. 5) using tractors and trucks. The distance of landfill is approximately 10 km away from the center of the main city.

Scenario 2 (S2): baseline scenario

Baseline scenario (S2) corresponds to the existing MSWM scenario of the study area that has been described in Fig. 5. Currently, DMC is practicing unscientific method for MSW disposal. Recycling activities in the study area included only collection and separation of mixed plastics, metals, and glass products from the waste stream. Only 2% (2.95 ton) of total generated waste was recycled in the study area by the informal recyclers and rag pickers (Fig. 5). The emissions caused by the segregated recyclable materials that are sent outside the city for further processing were excluded from the study. One percent (1.48 ton) of total generated waste was open burnt by the waste owners or by local people at the generation and collection points. in The study area, 23.5% (34.58 ton) of waste that remains uncollected was considered as open dumping in the present study. The remaining 73.5% (108 ton) wastes were collected from various localities and was directly disposed in unsanitary landfill sites (Fig. 5).

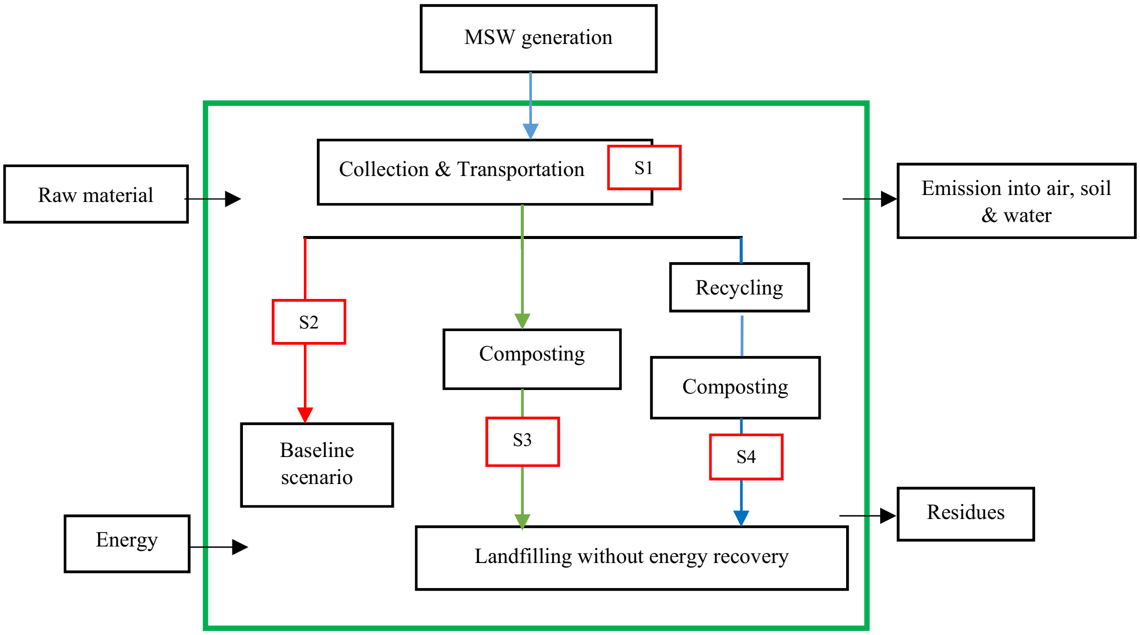

Fig. 4 System boundary of the present study

Scenario 3 (S3): composting and landfilling without energy recovery

The simplest approach in future would be to convert the open dumps and unsanitary landfill into sanitary landfills. The scenario (S3) in the present study assumed that the food and yard waste (37.77%) will transported for the windrow composting (aerobic), due to high moisture content in food and yard waste. However, the rest of the wastes (62.23%) was assumed transported to the sanitary landfill without energy recovery to the unsanitary landfill sites of the present study area. It was also assumed that the wastes are suitable for landfilling without energy recovery due to the presence of high amount of inert wastes. However, landfill gas is only produced from the biodegradable fraction of the MSW (Rajcoomar and Ramjeawon 2017).

Scenario 4 (S4): recycling, composting, and landfilling without energy recovery

Scenario 4 (S4) was chosen to minimize the environmental impacts of MSW disposal by recycling, composting, and

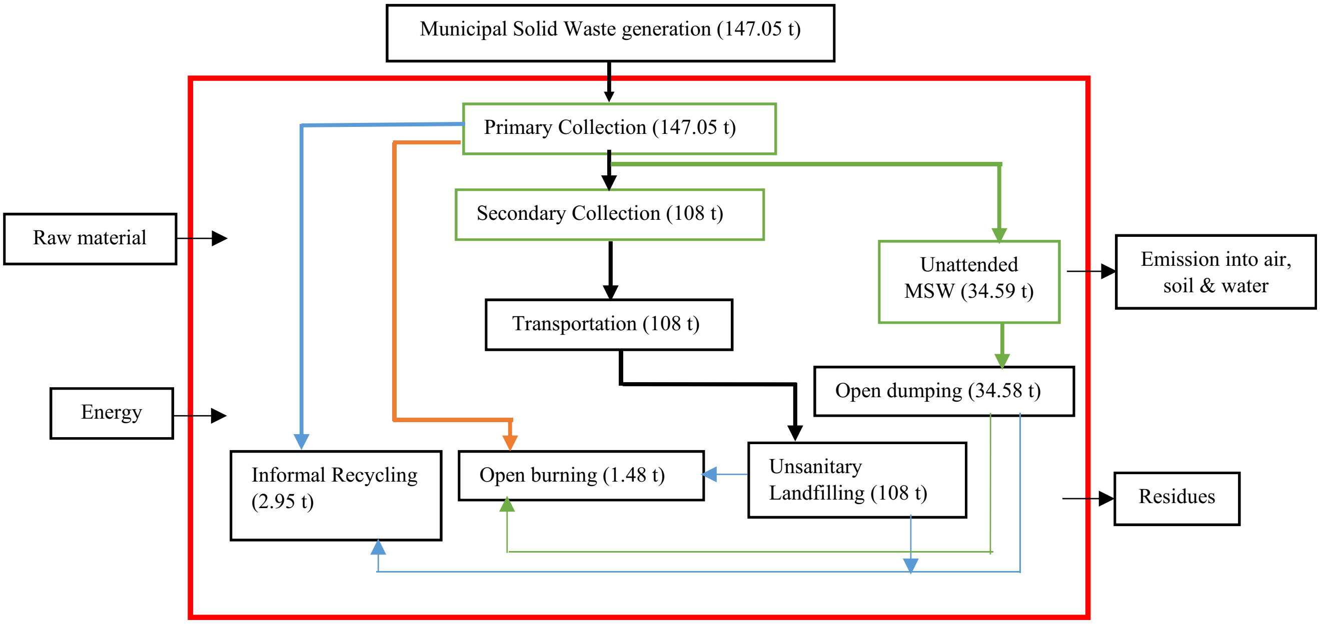

Fig. 5 Baseline scenario of the MSW management in the study area

landfilling. In S4, plastic, paper, card board, metal, glass, e-waste, and recyclable textile products (28.67%) were recycled, food and garden wastes (37.77%) were sent for composting, while the remaining wastes (only inert wastes) (residues of recycling and composting, construction and demolition waste, ash, soil, and rubber) were disposed in landfill without energy recovery. For this reason, it was assumed that the emissions from landfill site will be zero as it received only inert wastes.

Life cycle inventory

The life cycle inventory data used in present study were generated from the laboratory analysis of MSW samples of the study area, on-site investigations, previously published literatures, and the database of SimaPro 8.0.1.

Collection and transportation of wastes

In scenario 1, at present, 100% waste collection is the biggest challenge for the DMC. DMC spent almost 80 to 90% of their waste management budget for waste collection and transportation only. Despite such high spending, waste collection efficiency is only 73.5% (108 ton) in the study area. While most of the developed countries use less than 10% of their municipal budget on waste collection and transportation and rest for waste disposal process (Yadav and Samadder 2017b). The primary data related to the collection and transportation of wastes were collected by the physical survey of the study area and from DMC office. At present, there is no primary waste collection system, the local people dispose their waste by their own means in street dustbins provided by the DMC. The wastes from residential areas, commercial areas, public places like markets, bus stands, etc. were collected regularly by DMC. Local authorities (DMC and other local bodies) provided dustbins (stationary container system) at various locations within the study area. In addition, there were 20 open waste collection points across the study area. Dustbins were provided mainly on major roads and important localities (residential and commercial) of the city. The places where dustbins were not provided, people dump their wastes at a point from where the local authority collects the waste. The average distance between the dustbins varies from 1500 to 2000 m. There was no intermediate collection or transit points and the collected solid wastes were directly transported to the disposal site from bins by DMC vehicles. Waste was collected from dustbins manually and loaded into the transportation vehicles. At some places, collection was done with the help of tricycles (Fig. 5). Total amount of fuel used for collection purpose was considered as the emission from the collection process. The emissions caused due to burning of diesel (in g/L) for 1 ton of waste collection and transportation were included in the study and provided in Table 2 (Kebin et al. 2010; Babu et al. 2014). The local authorities of DMC allocated one vehicle to each administrative ward for waste collection. Thus, total of 18 vehicles were used for the waste collection and transportation from the study area (18 wards). Vehicles of 2.5 ton capacity (tractors and trucks) with 1 driver and four to five labors were used for the transportation of solid waste to the disposal site. The amount of diesel consumed by the tractors for the transportation of 1 ton of waste to the disposal sites was taken as 2 L and was verified from the DMC office. EURO 2 standards were used for choosing tractor emissions (diesel-driven vehicles) from SimaPro 8.1.

The average distance between the waste collection points and the present landfill location is approximately 10 km. Therefore, the distance for transporting the waste was assumed as 10 km in scenarios S3 and S4 despite the fact that some waste treatment facilities can be farther away. In the present study, it was assumed that the sorting and recycling plant, the composting plant, and the landfill site are at the same site. So, the transport emissions for scenario 3 and scenario 4 are expected to be similar.

Open burning

In this study, open burning was considered as the burning of MSW (such as yards waste, paper, plastic, rubber, and textiles waste without using incinerators) in outdoor areas, which resulted into release of pollutants in the atmosphere. Open burning of waste material is an illegal disposal method in India. It was recognized as a major source of carbon monoxide (38.56 kg/ton), a colorless and odorless gas, formed from the incomplete combustion of fuel, which also contributes to the greenhouse effect, smog, and acidification (other emissions are explained in Table 3) (EPA 1997; EPA 1995; Babu et al. 2014). Open burning of MSW also leads to the release of particulate matters (PM_{10}—17.24 kg/ton, $PM_{2.5}$—15.43 kg/ton) into the air, along with the dust, dirt, soot, smoke, and liquid droplets (Cabaraban et al. 2008).

Table 2 Gaseous emissions into air from 1 L of diesel oil for the transportation of MSW

Gaseous emissions from diesel	Unit (g/L) diesel
CO_2	2663
CO	11.95
HC	1.75
NO_X	2.36
$PM_{2.5}$	0.62

Sources: (Kebin et al. 2010; Babu et al. 2014)

Table 3 Emission of different pollutants from open burning of 1 ton of MSW

Pollutants	Emissions (kg/t)
SOx	0.46
CO	38.56
CH_4	5.89
Nitrogen oxide	2.73
VOC	3.88
PM_{10}	17.24
$PM_{2.5}$	15.43
Chlorobenzenes	3.849×10^{-4}
Benzene	1.125
Acetone	0.853
Styrene	0.67
Phenol	0.127
Dichlorobenzene	1.45×10^{-4}
Trichlorobenzens	9.97×10^{-5}
Tetrachloro benzene	6.71×10^{-5}
Penta chlorobenzene	4.8×10^{-5}
Hexa chlorobenzene	1.99×10^{-5}
Total polycyclic aromatic hydrocarbons (PAH)	5.99×10^{-2}
Acenaphthylene	9.9×10^{-3}
Naphthalene	1.64×10^{-2}
Phenantherne	6.63×10^{-3}
Total polychloribenyn (PCB)	5.72×10^{-3}
Hydrogen chloride (HCL)	0.568
Hydrogen cyanide (HCN)	0.936

Source: (Babu et al. 2014; EPA 1997; EPA 1995)

Open dumping

The uncollected waste in the study area was approximately 23.5% of the total generated waste, and it was considered as open dumping. Open dumping of MSW includes the unattended areas by the DMC that are used for unsanitary dumping of untreated and unsegregated solid wastes. The open dumps are the places which do not have any liner systems and are temporarily used as a waste disposal sites (Babu et al. 2014). There is no initial costs incurred in this method but the environmental consequences are very high as the leachate may pollute the soil and groundwater as well as the emissions could lead to air pollution (Babu et al. 2014). The air emissions from an open dump were calculated with the help of chemical formula of biodegradable MSW of the study area. The chemical formula ($C_{86}H_{197}O_{64}N$) was used after ultimate analysis of the biodegradable wastes samples of the study area. The emissions (CO_2, CH_4, and NH_3) due to anaerobic degradation of biodegradable wastes was calculated and found as 705.1 kg/ ton of CO_2, 361.7 kg/ton of CH_4, and 7.50 kg/ton of NH_3. Other air emissions from open dump were retrieved from Babu et al. (2014) (CO = 3×10^{-2} kg/ton, HC = 5.17×10^{-3} kg/ton, $NO_X = 1.4 \times 10^{-3}$ kg/ton, and P.M $_{2.5} = 2.217 \times 10^{-4}$ kg/ton). The emissions into water due to open dumping was not considered in this study.

Recycling

Recycling can reduce direct as well as indirect greenhouse gas (GHG) emissions by reducing the amount of virgin material being processed and avoiding emissions of CO_2 and CH_4 (Saft and Elsinga 2006). Direct emissions can be reduced when waste is not disposed at landfills or not treated in unsanitary way (e.g., direct combustion). Indirect emissions can be cut down by decreasing the energy consumption both in acquiring and producing raw materials and also in manufacturing the product itself (Korhonen and Dahlbo 2007). Baseline scenario considered zero emissions from the recycling because the collection and separation process in the study area were found to be carried out manually without use of any machinery. The fuel was found to be used for the transportation of recyclable products to some other city for further processing, since the study area do not have any recyclable products processing facility. In scenario 4, recycling was assumed for MSW fractions: paper (5.69%), cardboard (4.19%), plastic (12.15%), glass-(2.07%), metal (1.54%), and textiles (2.17%). The amount of electricity and diesel required in machineries operation for the MRF was 3.2 kWh and 3.21 L/ton of waste, respectively (Rajaeifar et al. 2015). The indirect environmental burdens associated with the electricity and diesel consumption during the treatment process have been taken from the Ecoinvent 3 database (SimaPro 2014). The recycling rate of 40% has been assumed for the study area. The rejects and residues from recycling processes were assumed to be landfilled.

Composting

Composting is the third preferred choice in the integrated solid waste management hierarchy (CPHEEO 2016). Aerobic composting is carried out in presence of air and this process yields humus rich compost (organic manure) along with macronutrients and micronutrients for the plants. The high moisture content of biodegradable waste is responsible for lower heating value (LHV) of MSW, which reduces its combustion efficiency. Therefore, windrow composting of bio-waste (food and yard waste) has been considered in scenarios 3 and 4. The assumptions for composting process were based on the inventory data taken from the literature and the waste composition analysis of the study area. The N, P, and K contents of the compost product were taken from literature that were calculated using mass percentages of N, P, and K nutrients at 0.83, 0.2, and 0.99%, respectively (Song et al. 2013). Air emissions from the windrow composting were estimated using the chemical formula ($C_{86}H_{197}O_{64}N$) of the bio-waste which was calculated using the fractional mass composition of the MSW of the study area (Tchobanoglous et al. 1993). CO_2 and CH_4 are found the highest in the center of the windrows, where O_2 is absent (Hao et al. 2001). Jackel et al. (2005) reported that 46–98% of the CH_4 produced in windrows is oxidized by methanotrophic bacteria before it escapes the windrow. The optimum temperatures for methane oxidation in the windrow range from 45 to 55 °C. It was assumed that CH_4 was oxidized by the microorganisms in the upper layer of the compost material (Andersen et al. 2010). According to the mass balance approach, 1 ton wet MSW is expected to produce approximately 0.14 ton of residues in a good quality MSW composting facility and approximately 0.80 ton of wet compost product (Komilis and Ham 2004). It was assumed that in dry seasons leachate can be recirculated into the windrow to reduce loss of nutrients and also to reduce pollution potential. In high rainfall areas, the windrows need to be covered either temporarily or permanently to control leachate generation (Komilis and Ham 2004).

Landfilling without energy recovery

In baseline scenario, landfill sites were not equipped with any liner or gas and leachate collection systems in the study area. Organic materials under anaerobic condition produce landfill gas (LFG). The amount of LFG release from landfill site was determined by biodegradable organic content of the MSW (Babu et al. 2014). The landfill sites produce significant amounts of methane (59.67 kg/ton), biogenic carbon dioxide (25 kg/ton), non-methane volatile organic compounds (0.388 kg/ton) as well as the smaller amounts of nitrous oxide (1.47 g/ton) and carbon monoxide (3 g/ton) (Babu et al. 2014; Mboowa et al. 2017). The emissions into water from landfill site and heavy metal concentrations were taken from Samadder et al. (2017) as mentioned in Table 4. In scenario 3, high amount of inert wastes placed in the landfill site that usually has short contaminant transport potential and it is chemically inert. If no gas collection occurs then no electricity is required for gas pumps (Yay 2015). Scenarios 3 and 4 ensure the necessities of an engineered landfill site for waste disposal. However S3 needs the gas recovery system and leachate collection system, but S4 does not need because scenario S4 assumed that only inert waste will go for the landfill. Therefore, the methane and carbon dioxide emissions from the landfill site were considered zero due to inert nature of MSW.

Life cycle impact assessment of MSW disposal

Life cycle impact assessment (LCIA) was conducted using the method Centre for Environmental Studies (CML) 2 baseline 2000. The method was developed by the Center of Environmental Science of Leiden University in the Netherland, which supports the 11 impact categories [i.e.,

Table 4 Emissions from landfill site into air, water and soil from 1 ton of MSW

Emission into soil		Emission into water		Emissions into air	
Name of the pollutant	Quantity of emission (g/t)	Name of the pollutant	Quantity of emission (g/t)	Name of the pollutant	Quantity of emission (g/t)
Na	101.79	NO^{-3}	5.9619	Methane	59,670
K	162.86	F^{-}	3.8293	CO_2	25,000
N	51.50	Na	20.2220	CO	3
Mg	591.86	K	0.7503	HC	388
Ca	585.86	TN	1.6207	NOx	1.47
P	45.26	TP	5.5500	$PM_{2.5}$	0.233
SAR	3.16	Fe	0.4425		
Zn	2.25	As	0.1099		
Fe	17.42	Pb	2.84×10^{-2}		
Ni	0.54	Hg	9.1×10^{-3}		
Fe	17.42	Zn	0.2285		
Mn	102.25	Ni	3.79×10^{-2}		
		pH	7.47		
		EC (μs/cm)	102.59		
		CEP (cmol (+) /kg)	8.92		

Source: (Mboowa et al. 2017; Samadder et al. 2017; Babu et al. 2014)

CEC cation exchange capacity, *EC* electrical conductivity

Table 5 Emissions of MSWM scenarios on different impact categories (based on LCA characterization)

S. N	Impact category	Unit	MSWM scenarios			
			S1	S2	S3	S4
1	Abiotic depletion	kg Sb eq.	4.20E − 05	4.46E − 07	7.20E − 06	7.15E − 06
2	Abiotic depletion (fossil fuels)	MJ	2.09E + 02	6.32E + 01	2.71E + 02	2.52E + 02
3	Global warming (GWP100a)	kg CO$_2$ eq.	1.24E + 01	9.42E + 03	4.92E + 03	3.43E + 03
4	Ozone layer depletion (ODP)	kg CFC-11 eq.	2.24E − 06	7.60E − 07	2.73E − 06	2.47E − 06
5	Human toxicity	kg 1,4-DB eq.	4.76E + 00	2.25E + 01	5.61E + 00	5.50E + 00
6	Fresh water aquatic ecotoxicity	kg 1,4-DB eq.	2.61E + 00	1.57E + 00	6.54E + 00	5.39E + 00
7	Marine aquatic ecotoxicity	kg 1,4-DB eq.	1.86E + 04	1.23E + 03	1.86E + 04	1.81E + 04
8	Terrestrial ecotoxicity	kg 1,4-DB eq.	2.42E − 02	7.87E − 03	3.36E − 02	3.19E − 02
9	Photochemical oxidation	kg C$_2$H$_4$ eq.	4.13E − 03	2.12E + 00	1.20E + 00	8.72E − 01
10	Acidification	kg SO$_2$ eq.	6.60E − 02	1.15E + 01	4.63E + 00	4.63E + 00
11	Eutrophication	kg PO$_4^{3-}$ eq.	1.54E − 02	2.63E + 00	2.08E + 00	1.99E + 00

abiotic depletion, abiotic depletion (fossil fuels), GWP100, ozone layer depletion (ODP), human toxicity potentials (HTP), fresh water aquatic ecotoxicity (FWAE), marine aquatic ecotoxicity (MWAE), terrestrial ecotoxicity (TE), photochemical oxidation potential (POP), acidification potential (AP), and eutrophication potential (EP)] and all of them were used in this study. The abiotic depletion is related to the extraction of minerals due to inputs in the system. The abiotic depletion factor is determined for extraction of minerals (kg antimony equivalents/kg extraction) based on concentration reserves and rate of deaccumulation (Goedkoop et al. 2004). AD of fossil fuels is related to the LHV expressed in megajoule per kilogram of cubic meter fossil fuel. The reason for taking the LHV is that fossil fuels are considered to be fully substitutable. Global warming characterization model as developed by the Intergovernmental Panel on Climate Change (IPCC) was selected for development of characterization factors. Factors are expressed as GWP for time horizon of 100 years (GWP100) in kilogram CO$_2$ equivalent/kilogram emission. ODP (steady state) characterization model was developed by the World Meteorological Organization to define ODP of different gases (kg CFC-11 equivalent/kg emission). Human toxicity, FWAE, MWAE, and TE characterization factors, expressed as HTP's are calculated using LCA for describing fate, exposure, and effects of toxic substances for an infinite time horizon. For each of the toxic substances, HTP is expressed as 1,4-dichlorobenzene equivalents/kilogram emission. Photochemical oxidation (high NOx) model is expressed in kilogram ethylene equivalents/kilogram emission. AP is expressed in kilogram SO$_2$ equivalents/kilogram emission. EP was developed by Huijbregts et al. (2003) and expressed in kilogram PO$_4^{3-}$ equivalents/kilogram emission (PRé, Consultants 2008).

Table 6 Emissions of MSWM scenarios on different impact categories (based on LCA normalization)

S. N	Impact category	MSWM scenarios			
		S1	S2	S3	S4
1	Abiotic depletion	2.01E − 13	2.13E − 15	3.44E − 14	3.42E − 14
2	Abiotic depletion (fossil fuels)	5.49E − 13	1.66E − 13	7.16E − 13	6.64E − 13
3	Global warming (GWP100a)	2.97E − 13	2.25E − 10	2.26E − 10	8.19E − 11
4	Ozone layer depletion (ODP)	9.89E − 15	3.35E − 15	1.21E − 14	1.09E − 14
5	Human toxicity	1.85E − 12	8.72E − 12	2.36E − 12	2.13E − 12
6	Fresh water aquatic ecotoxicity	1.10E − 12	6.65E − 13	2.78E − 12	2.28E − 12
7	Marine aquatic ecotoxicity	9.60E − 11	6.33E − 12	9.60E − 11	9.34E − 11
8	Terrestrial ecotoxicity	2.21E − 14	7.20E − 15	3.08E − 14	2.92E − 14
9	Photochemical oxidation	1.12E − 13	5.76E − 11	6.03E − 11	2.37E − 11
10	Acidification	2.77E − 13	4.81E − 11	5.07E − 11	1.94E − 11
11	Eutrophication	9.72E − 14	1.66E − 11	2.35E − 11	1.25E − 11
	Total	1.01E − 10	3.63E − 10	4.62E − 10	2.36E − 10

Results and discussion

The results of the MSW composition analysis have been presented in Fig. 3b as an average. As seen in Fig. 3b, food waste (24.34%) (kitchen and vegetables and fruit peels) was found to be maximum in the study area. The other components of MSW in the waste stream were yard waste (13.43%), plastic (12.15%), ash (15.35%), soil (6.01%), paper (5.69%), cardboard (4.19%), and glass (2.07%) that were also found in significant amount. The bulky wastes like furniture (wood and plastics) were absent in the study area. These bulky wastes were reused by waste generators or scrap dealers.

The moisture contents of food wastes and yard wastes were 79.98 and 51%, respectively. In the study area, the average moisture content (65.49%) values of biodegradable waste were found higher than that values of China (61%) (Zhenshan et al. 2009) and Turkey (57%) (Yay 2015). The compostable waste was found to be the maximum (37.77%), while inert waste was 31.05%, recyclable waste was 26.5%, and incinerable waste 4.68% (Fig. 3a).

The LCA characterization (Table 5) and normalization (Table 6) results for each impact category of all the waste management scenarios have been presented in Figs. 6, 7, 8, 9, and 10. As reported in the Table 5 and Fig. 9, the results were explored for each environmental impact category. As shown in Fig. 6a, collection and transportation [S1 (4.20E − 05 kg Sb eq.)] had the higher impact than scenario S2 (4.46E − 07 kg Sb eq.), S3 (7.20E − 06 kg Sb eq.), and S4 (7.15E − 06 kg Sb eq.) on AD due to the use of fossil fuels such as diesel oil. GWP 100 for a time horizon of 100 years is measured in kilogram carbon dioxide/kilogram emission (Goedkoop et al. 2004). Open dumping and landfilling were

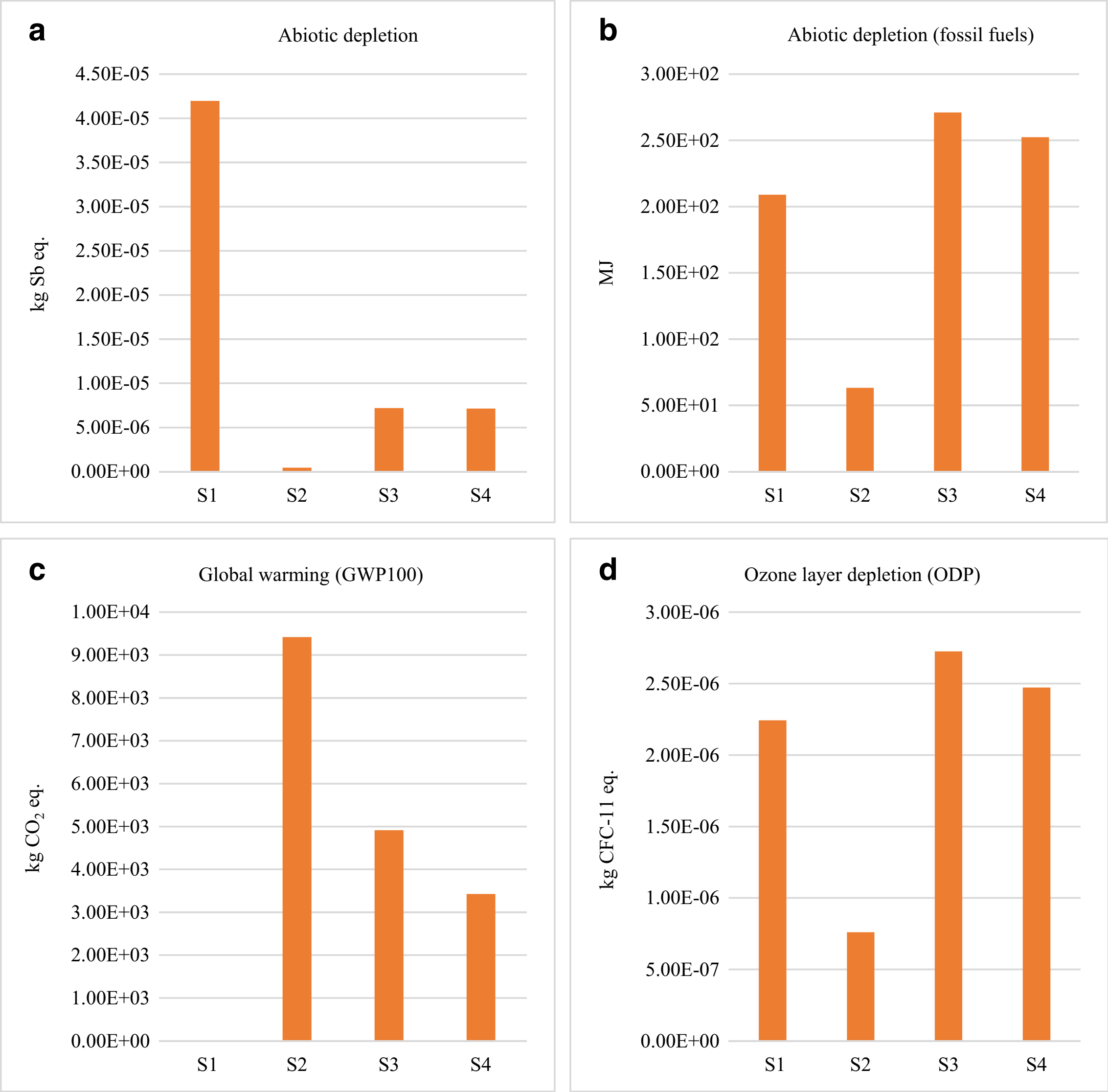

Fig. 6 Environmental impacts of different MSW management scenarios on **a** abiotic depletion, **b** abiotic depletion (fossil fuels), **c** global warming (GWP 100a), and **d** ozone layer depletion (ODP)

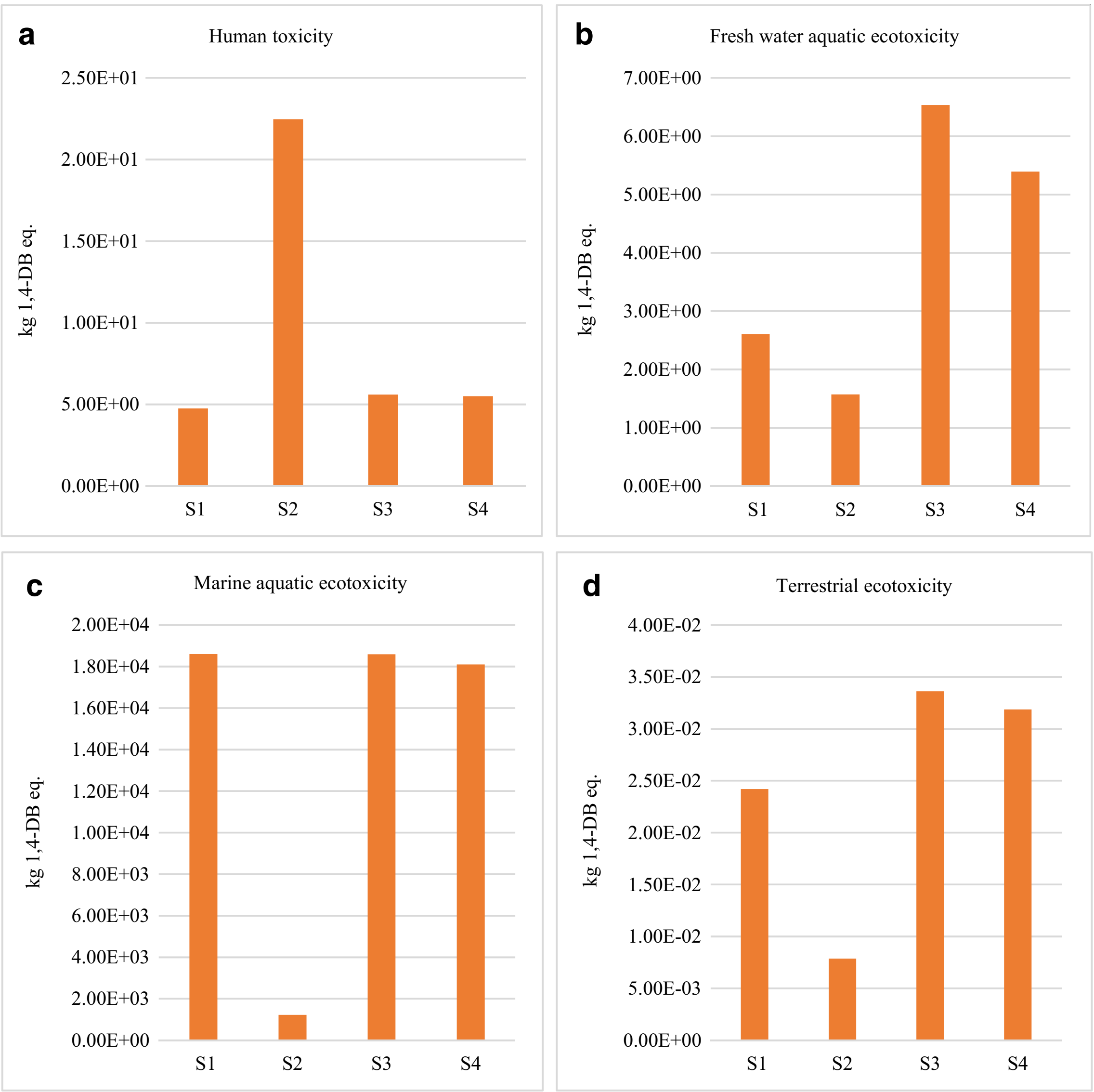

Fig. 7 Environmental impacts of different MSW management scenarios on **a** human toxicity, **b** fresh water aquatic ecotoxicity, **c** marine aquatic ecotoxicity, and **d** terrestrial ecotoxicity

found as the main contributors of global warming due to methane emission. Therefore, scenario S2 (9.42E + 03 kg CO_2 eq.) had the higher impact on GWP than S1 (1.24E + 01 kg CO_2 eq.), S3 (4.92E + 03 kg CO_2 eq.), and S4 (3.43E + 03 kg CO_2 eq.) as shown in Fig. 6c. The absence of an LFG control system in unsanitary landfill is the reason for the uncontrolled release of methane into the atmosphere. Methane bromotrifluoro-Halon 1301 is a product of crude oil, petroleum, and natural gas production that is the main cause of ozone layer depletion (Yay 2015). Scenario S3 (2.73E − 06 kg CFC-11 eq.), which included composting and sanitary landfilling had the higher impact on ODP than S1 (2.24E − 06 kg CFC-11 eq.), S2 (7.60E − 07 kg CFC-11 eq.), and S4 (2.47E − 06 kg CFC-11 eq.) scenarios in the present study (Table 5 and Fig. 6d).

HTP describe the effect of toxic substances (1,4-dichlorobenzene equivalents/kg emission) for infinite time (Goedkoop et al. 2004; Yay 2015). In scenario S2, open burning was the main contributor of 1,4-dichlorobenzene that caused the human toxicity. Transportation (fuel consumption) and heavy metals were other reasons for HTP. Scenario S2 (2.25E + 01 kg 1,4-DB eq.) had higher impact on HTP than S1 (4.76E + 00 kg 1,4-DB eq.), S3 (5.61E + 00 kg 1,4-DB eq.), and S4 (5.50E + 00 kg 1,4-DB eq.) scenarios due to open burning and landfilling (Table 5 and Fig. 7a). Nickel, arsenic, lead, zinc, mercury, and barium are the pollutants discharged during landfilling, composting, and open burning processes and cause FWAE, MWAE, and terrestrial ecotoxicity. As shown in Fig. 7b, the scenario S3 (6.54E + 00 kg 1,4-DB eq.) had

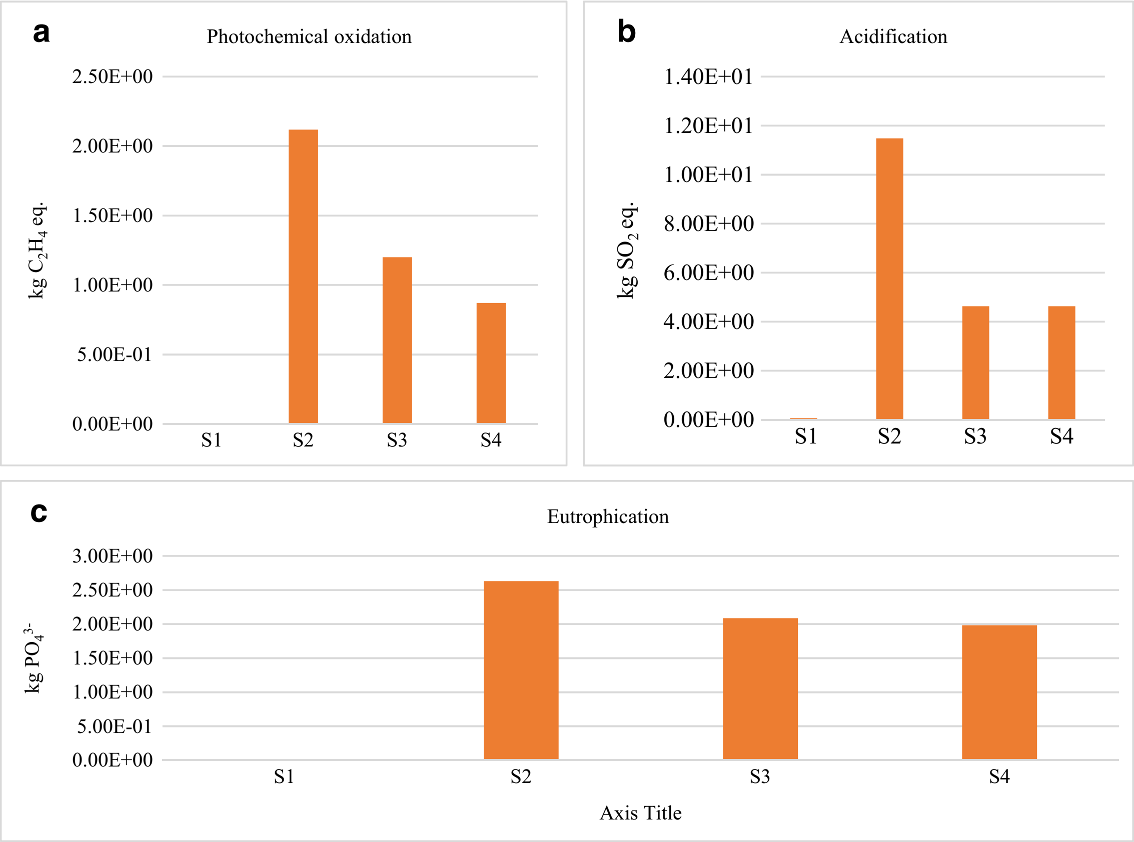

Fig. 8 Environmental impacts of different MSW management scenarios on **a** photochemical oxidation, **b** acidification, and **c** eutrophication

the higher impact on FWAE than S1 (2.61E + 00 kg 1,4-DB eq.), S2 (1.57E + 00 kg 1,4-DB eq.), and S4 (5.39E + 00 kg 1,4-DB eq.) scenarios.

Volatile organic compounds (VOCs) are the main contributors of photochemical ozone formation that contributes to photochemical oxidation (Hauschild and Wenzel 1998). Open dumping and landfilling were the main sources of methane that cause adverse impact on photochemical oxidation, particulate matter emissions, and other hydrocarbon emissions. Sulfur dioxide emission produced by the transportation, open burning of MSW, and the usage of electricity for running the equipment also creates adverse repercussion of POP in scenario S2. As presented in Table 5 and Fig. 8a, scenario S1 (4.13E − 03 kg C_2H_4 eq.) had the lowest impacts on POP while S2 had the highest (2.12E + 00 kg C_2H_4 eq.) impacts on POP among all the scenarios.

The AP is expressed as the number of H^+ ions formed per kilogram of substance relative to SO_2 (Bauman and Tillman 2004). The most important acidifying pollutants were NOx, HCl, SO_2, and NH_3 in the present study that caused the maximum impact on acidification in scenario S2 (1.15E + 01 kg SO_2 eq.) (Fig. 8b). Scenario S3 (1.86E + 04 kg 1,4-DB eq.) and S1 (1.86E + 04 kg 1,4-DB eq.) had the maximum impact on MWAE (Fig. 7c) and S3 (3.36E-02 kg 1,4-DB eq.) had the highest impact on TE (Fig. 7d). Composting of MSW can increase heavy metals burden of soil (Ramos and López-Acevedo 2004). That is the reason that S3 had higher impacts in MWAE, FWAE, AD (fossil fuel), and TE impact categories as shown in Fig. 6b, d and Fig. 7b–d. Scenario S2 had the highest impact on EP (2.63E + 00 kg PO_4^{3-} eq.) among all the scenarios [S1 (1.54E − 02 kg PO_4^{3-} eq.), S3 (2.08E + 00 kg PO_4^{3-}eq.), and S4 (1.99E + 00 kg PO_4^{3-}eq.)] (Fig. 8c). As presented in Fig. 9 (three MSWM practicing scenarios S2, S3, and S4), S4 had the lowest environmental impacts on GWP (3.43E + 03 kg CO_2 eq.), HTP (5.50E + 00 kg 1,4-DB eq.), POP (8.72E-01 kg C_2H_4 eq.), AP (4.63E + 00 kg SO_2 eq.), and EP (1.99E + 00 kg PO_4^{3-} eq.).

The environmental impacts due to landfilling of MSW (wet weight) on AP, GWP, and abiotic depletion potential (fossil fuel) in different countries have been reported by various authors [(such as by Arena et al. (2003) in Italy, Eriksson et al. (2005) in Sweden, Banar et al. (2009) in Turkey, Mendes et al. (2004) in Brazil, and Hong et al. (2010) in China)]. Arena et al. (2003) reported that the impacts of landfilling of MSW on AP, GWP, and AD (fossil fuel) were − 0.44 kg SO_2 eq./ton, 490 kg CO_2 eq./ton, and − 0.67 GJ/ton, respectively. Eriksson et al. (2005) reported that the environmental impacts on AP and GWP categories were 0.99 kg SO_2 eq./ton and 580 kg CO_2 eq./ton, respectively. Banar et al. (2009) reported that

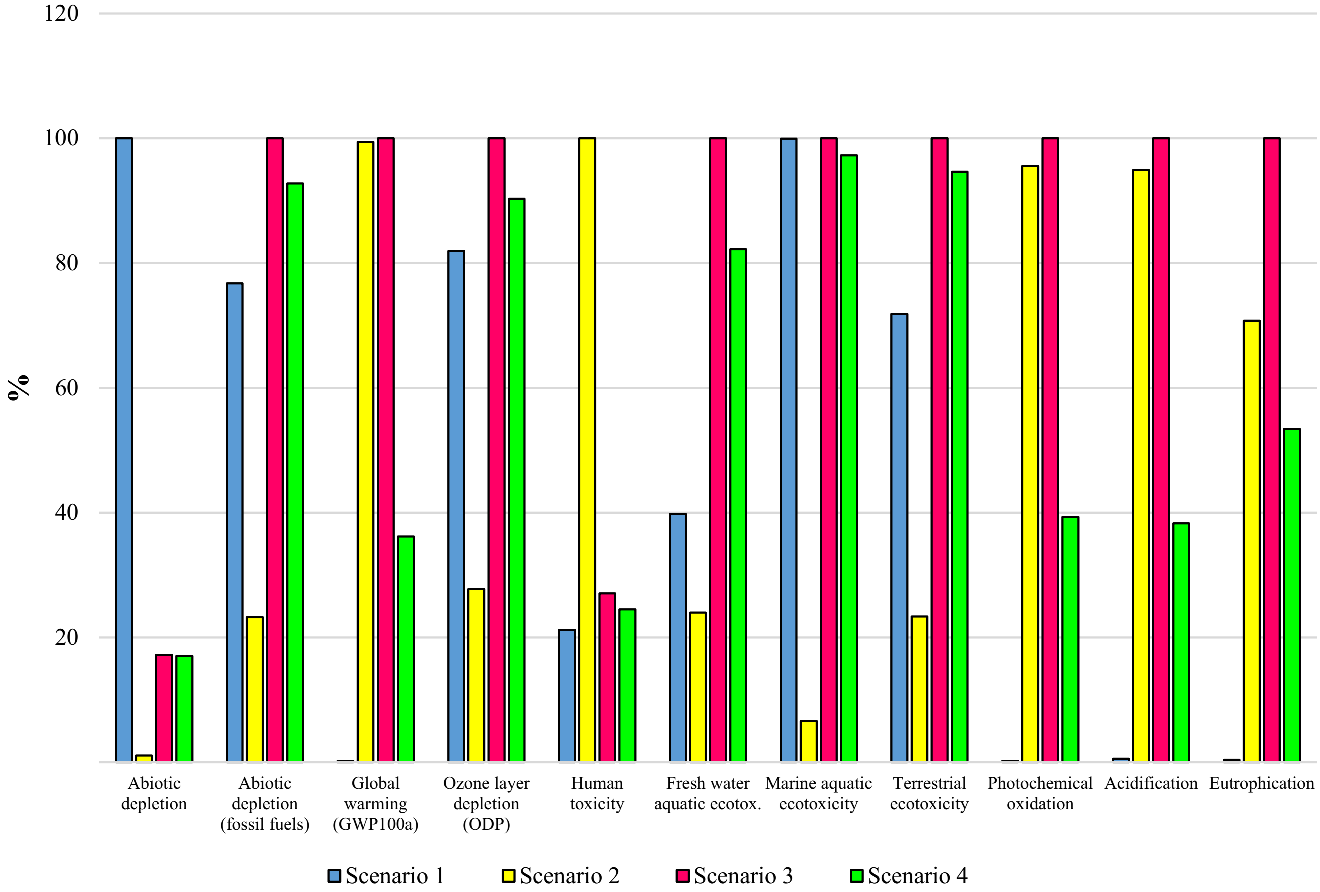

Fig. 9 Comparison of environmental impacts of the different MSWM scenarios on various impact categories based on the LCA characterization results

environmental impacts on AP and GWP categories were 43.6 kg SO_2 eq./ton and 6990 kg CO_2 eq./ton, respectively. Mendes et al. (2004) reported that the impacts on AP, GWP, and AD (fossil fuel) categories were 0.30 kg SO_2 eq./ton, 900 kg CO_2 eq./ton, and 1.83 GJ/ton, respectively. Hong et al. (2010) reported that the impacts on AP, GWP, and AD (fossil fuel) categories were 0.89 kg SO_2 eq./ton, 625 kg CO_2 eq./ton, and − 0.07 GJ/ton, respectively. In the present study, CML 2 baseline 2000 life cycle impact results showed that the impacts of landfilling of MSW (wet weight) on AP, GWP, and AD (fossil fuel) categories were 0.007 kg SO_2 eq./ton, 1319 kg CO_2 eq./ton, and 63.24 MJ/ton, respectively. However, in the present study, GWP was found higher than the previously published studies of Arena et al. (2003), Eriksson et al. (2005), Mendes et al. (2004), and Hong et al. (2010), but lower than the Banar et al. (2009). The difference in results was found due to the difference in composition of the wastes and climatic conditions of the different study areas. In the present study, GWP impact was higher mainly due to the absence of LFG collection system. It is worth to note that electricity recovery from methane gas trapped in a landfill can significantly reduce the GWP, thereby reducing the overall environmental impacts of landfilling. In the present study, CML 2 baseline 2000 method was used considering 100-year time horizon (25 kg CO_2 eq./kg CH_4). Whereas, Hong et al. (2010) used IMPACT 2002+ method considering 500-year time horizon in global warming category, in such case the GWP of CH_4 is 7 kg CO_2 eq./kg CH_4. The difference in the results also depends on the selection of LCIA methods. However, in this study, the values of aquatic acidification and abiotic depletion potential impacts of landfill process were lower than the values mentioned in the previous studies, mainly due to lower consumption of fossil fuels (petroleum products) in the study area.

Figure 10 and Table 6 present the normalization results of the scenarios S1, S2, S3, and S4 on the basis of different environmental impact categories. Normalization is an optional step of LCA according to ISO 14040/44 to rank the impacts of different MSWM options and scenarios (Aymard and Botta-Genoulaz 2017). As presented in Fig. 10, GWP had the highest impact among all the 11 environmental impact categories for all the scenarios except scenario S1 on the basis of normalization factor. MWAE ranked second in all the environmental impact categories for all the scenarios except scenario S2 (Fig. 10 and Table 6). Photochemical oxidation, acidification, and eutrophication ranked third, fourth, and fifth, respectively. HTP ranked seventh out of all the impact categories. While, all the scenarios had negligible impacts on

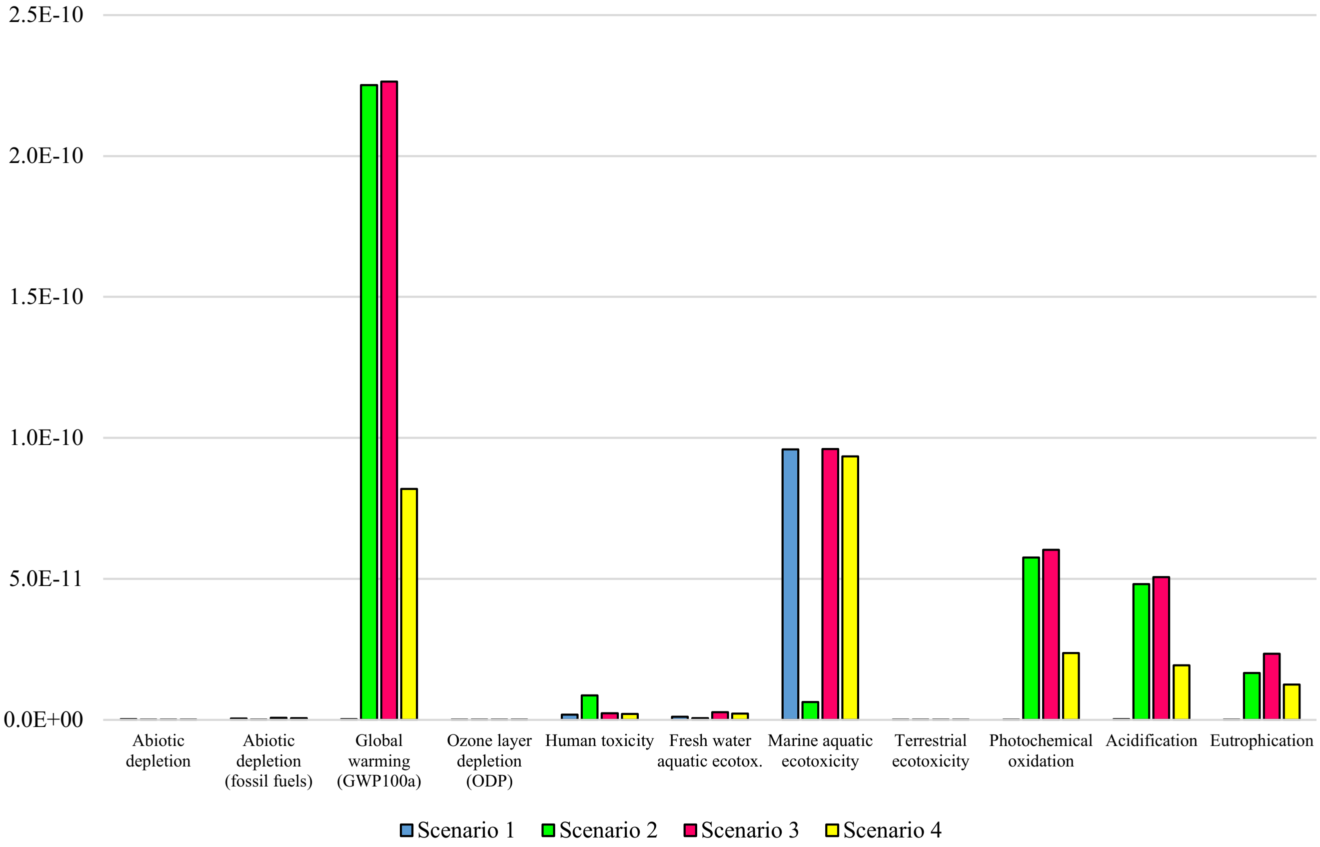

Fig. 10 Comparison of environmental impacts of the different MSWM scenarios on various impact categories based on LCA normalization results

FWAE, ODP, abiotic depletion, and abiotic depletion (fossil fuels). GWP had the highest impact in all the environmental impact categories considered in this study except scenario S1.

After the life cycle analysis, the potential impact was observed more in composting and landfilling (S3) in the present study, but it could be reduced when composting and landfilling were done along with MRF/recycling (S4). The environmental impacts due to incineration of waste in the study area can be avoided with the recycling of packaging products (paper and plastic) and composting of yard and food wastes. Scenario S4 covered all the components of the MSW in the study area and it was found as the most suitable MSWM scenario for the study area due to its minimum environmental impacts with compared to all the other scenarios considered in this study.

Conclusions

The study compared environmental impact of four different waste management scenarios. The existing waste management practice in Dhanbad (modeled as baseline scenario) was estimated to cause maximum impact on GWP, photochemical oxidation, eutrophication, and HTP. The results indicated that increase in recycling rate of packaging waste would reduce the environmental impacts considerably due to very low inputs from technosphere. Waste recycling activities should be enhanced by promoting separation at source of generation through education and awareness among the community. According to the LCA results of this study, the highest environmental impact was observed from landfilling without energy recovery, open dumping, and open burning of mixed waste in scenario 2. The most suitable waste management scenario was S4 for the study area on the basis of GWP, photochemical oxidation, acidification, and eutrophication potential. The existing practice of municipal solid waste management in Dhanbad is not appropriate as environmental impacts on the major impact categories were high with compared to the proposed scenarios.

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References

Abduli MA, Naghib A, Yonesi M, Akbari A (2011) Life cycle assessment (LCA) of solid waste management strategies in Tehran: landfill and composting plus landfill. Environ Monit Assess 178:487–498

Al-Khatib IA, Monou M, Salam AF, Hafez AQ, Shaheen KD (2010) Solid waste characterization, quantification and management practices in developing countries. A case study: Nablus district-Palestine. J Enviro Manag 91:1131–1138

American Society of Testing and Materials (ASTM), 2003. Standard test methods for determination of the composition of unprocessed municipal solid waste. D 5231–92

Andersen JK, Boldrin A, Samuelsson J, Christensen TH, Scheutz C (2010) Quantification of greenhouse gas emissions from windrow composting of garden waste. J Environ Qual 39(2):713–724

Arena U, Mastellone ML, Perugini F (2003) The environmental performance of alternative solid waste management options: a life cycle assessment study. Chem Eng J 96:207–222

Aymard V, Botta-Genoulaz V (2017) Normalisation in life-cycle assessment: consequences of new European factors on decision-making. Supply Chain Forum: An International Journal 18(2):76–83

Babu GS, Lakshmikanthan P, Santhosh LG (2014) Life cycle analysis of municipal solid waste (MSW) land disposal options in Bangalore City. In ICSI 2014: Creating infrastructure for a sustainable world 795–806. Am Soc Civil Eng. https://doi.org/10.1061/9780784478745.075

Banar M, Cokaygil Z, Ozkan A (2009) Life cycle assessment of solid waste management options for Eskisehir Turkey. Waste Manag 29:54–62

Bauman H, Tillman A (2004) The hitch hiker's guide to LCA. Student literature AB, Stockholm

Beigl P, Salhofer S (2004) Comparison of ecological effects and costs of communal waste management systems. Resour Conserv Recycl 41:83–102

Bohra A, Nema AK, Ahluwalia P (2012) Global warming potential of waste management options: case study of Delhi. Int J Environ Technol Manag 15(3–6):346–362

Buttol P, Masoni P, Bonoli A, Goldoni S, Belladonna V, Cavazzuti C (2007) LCA of integrated MSW management systems: case study of the Bologna district. Waste Manag 27:1059–1070

Cabaraban MTI, Khire MV, Alocilja ECC (2008) Aerobic in-vessel composting versus bioreactor landfilling using life cycle inventory models. Clean Techn Environ Policy 10:1–39

Census of India (2011) Ministry of Home Affairs. Government of India, New Delhi <http://censusindia.gov.in/2011-common/censusdataonline.html> (accessed 03.02.2015)

Cherubini F, Bargigli S, Ulgiati S (2008) Life cycle assessment of urban waste management: energy performances and environmental impacts. The case of Rome, Italy. Waste Manag 28(12):2552–2564

CPHEEO (2016) Ministry of urban development, central public health and environmental engineering organization, ministry of urban development, Government of India, www.swachhbharaturban.gov.in. (assessed 16/08/ 2017)

Dong J, Chi Y, Zou D, Fu C, Huang Q, Ni M (2014) Comparison of municipal solid waste treatment technologies from a life cycle perspective in China. Waste Manag Res 32(1):13–23

EPA (1995) Compilation of air pollution emission factors, volume I: stationary point and area sources, fifth edition, AP-42 (GPO 055–000–00500-1). U.S. Environmental Protection Agency, Research Triangle Park, North Carolina

EPA (1997) Evaluation of emissions from the open burning of household waste in barrels. EPA-600/R-97-134a. U.S. Environmental Protection Agency, Control Technologies Center, Research Triangle Park, North Carolina

Eriksson O, Carlsson Reich M, Frostell B, Bjorklund A, Assefa G, Sundqvist JO (2005) Municipal solid waste management from a systems perspective. J Clean Prod 13:241–252

Goedkoop M, Oele M, Effting S (2004) Simapro Database Manual Methods Library. Pre Constultants, Netharlands

Hao X, Chang C, Larney FJ, Greg RT (2001) Greenhouse gas emissions during cattle feedlot manure composting. J Environ Qual 30:376–386

Hauschild M, Wenzel H (1998) Environmental assessment of products. In: scientific background, vol. 2. Chapman & Hall, Institute for product development, Technical University of Denmark

Hong RJ, Wang GF, Guo RZ, Cheng X, Liu Q, Zhang PJ, Qian GR (2006) Life cycle assessment of BMT-based integrated municipal solid waste management: case study in Pudong, China. Resour Conserv Recycl 49:129–146

Hong J, Li X, Zhaojie C (2010) Life cycle assessment of four municipal solid waste management scenarios in China. Waste Manag 30(11):2362–2369

Huijbregts MAJ, Breedveld L, Huppes G, de Koning A, van Oers L, Suh S (2003) Normalisation figures for environmental life-cycle assessment: The Netherlands (1997/1998), Western Europe (1995) and the world (1990 and 1995). J Clean Prod 11(7):737–748

Jackel U, Thummes K, Kämpfer P (2005) Thermophilic methane production and oxidation in compost. FEMS (Federation of European Microbiological Societies) Microbiol Ecol 52:175–184

Karoneas CJ, Nanaki EA (2012) Integrated solid waste management and energy production a life cycle assessment approach: the case study of the city of Thessaloniki. J Clean Prod 27:141–150

Kebin HE, Zhiliang Y, Zhang Y (2010) Characteristics of vehicle emissions in China based on portable emission measurement system. 19th Annual International Emission Inventory Conference "Emissions Inventories Informing Emerging Issues", San Antoni

Khan D, Kumar A, Samadder SR (2016) Impact of socioeconomic status on municipal solid waste generation rate. Waste Manag 49:15–25

Komilis DP, Ham RK (2004) Life-cycle inventory of municipal solid waste and yard waste windrow composting in the United States. J Environ Eng 130(11):1390–1400

Korhonen MR, Dahlbo H (2007) Reducing greenhouse gas emissions by recycling plastics and textiles into products. https://dspace3.hulib.helsinki.fi/bitstream/handle/10138/38425/FE_30_2007.pdf?sequence=3. Accessed 18 Oct 2017

Liamsanguan C, Gheewala SH (2008) LCA: a decision support tool for environmental assessment of MSW management systems. J Environ Manag 87(1):132–138

Mboowa D, Quereshi S, Bhattacharjee C, Tonny K, Dutta S (2017) Qualitative determination of energy potential and methane generation from municipal solid waste (MSW) in Dhanbad (India). Energy 123:386–391

Mendes MR, Aramaki T, Hanaki K (2004) Comparison of the environmental impact of incineration and landfilling in Sao Paulo City as determined by LCA. Resour Conserv Recycl 41:47–63

Menikpura SNM, Gheewala SH, Bonnet S, Chiemchaisri C (2013) Evaluation of the effect of recycling on sustainability of municipal solid waste management in Thailand. Waste Biomass Valoriz 4(2):237–257

Miliute J, Kazimieras Staniskis J (2010) Application of life-cycle assessment in optimization of municipal waste management systems: the case of Lithuania. Waste Manag Res 28:298–308

Ozeler D, Yetis U, Demirer GN (2006) Life cycle assessment of municipal solid waste management methods: Ankara case study. Environ Int 32:405–411

Peavy HS, Rowe DR, Techobanoglous G (1985) Environmental Engineering PP-579. Mc-Graw-Hill, International Edition, New York

POC "Population of Cities in" (2017) http://worldpopulationreview.com/countries/india-population/cities/. 3 may 2017

PRé, Consultants. "SimaPro Database Manual Methods Library." PRé Consultants (2008)

Rajaeifar MA, Tabatabaei M, Ghanavati H, Khoshnevisan B, Rafiee S (2015) Comparative life cycle assessment of different municipal solid waste management scenarios in Iran. Renew Sust Energ Rev 51:886–898

Rajcoomar A, Ramjeawon T (2017) Life cycle assessment of municipal solid waste management scenarios on the small island of Mauritius. Waste Manag Res 35(3):313–324

Ramos MC, López-Acevedo M (2004) Zinc levels in vineyard soils from the Alt Penedès-Anoia region (NE Spain) after compost application. Adv Environ Res 8(3):687–696

Ray MR, Roy Choudhury S, Mukherjee G, Roy S, Lahiri T (2005) Respiratory and general health impairments of workers employed in a municipal solid waste disposal at open landfill site in Delhi. Int J Hyg Environ Health 108(4):255–262

Saft RJ, Elsinga W (2006) Source separation, composting a win for greenhouse gas reduction. Bio Cycle 47(8):50–53

Saheri S, Mir MA, Basri NEA, Mahmood NZB, Begum RA (2012) Life cycle assessment for solid waste disposal options in Malaysia. Pol J Environ Stud 21:1377–1382

Samadder SR, Prabhakar R, Khan D, Kishan D, Chauhan MS (2017) Analysis of the contaminants released from municipal solid waste landfill site: a case study. Sci Total Environ 580:593–601

Sharma BK, Chandel MK (2017) Life cycle assessment of potential municipal solid waste management strategies for Mumbai, India. Waste Manag Res 35(1):79–91

Sharma M, McBean EA (2007) Methodology for solid waste characterization based on diminishing marginal returns. Waste Manag 27: 337–344

Song Q, Wang Z, Li J (2013) Environmental performance of municipal solid waste strategies based on LCA method: a case study of Macau. J Clean Prod 57:92–100

Srivastava AK, Nema AK (2011) Life cycle assessment of integrated solid waste management system of Delhi. In: Finkbeiner M (ed) Towards life cycle sustainability management. Springer, Dordrecht, pp 267–276

Tan RBH, Khoo HH (2006) Impact assessment of waste management options in Singapore. J Air Waste Manage Assoc 56:244–254

Tarantini M, Loprieno AD, Cucchi E, Frenquellucci F (2009) Life cycle assessment of waste management systems in Italian industrial areas: case study of 1st Macrolotto of Prato. Energy 34:613–622

Tchobanoglous G, Hilar T, Samuel AV (1993) Integrated solid waste management-engineering principles and management issues. Tata McGraw Hill International Edition, New Delhi, pp 382–383

UNEP (2011) Towards a green economy: pathways to sustainable development and poverty eradication. United Nations Environment Programme (UNEP), Nairobi

World Bank (2012) What a waste a global review of solid waste management. Urban Development Series Knowledge Paper No. 15, Washington, DC

Yadav P, Samadder SR (2017a) Assessment of applicability index for better management of municipal solid waste: a case study of Dhanbad, India. Environ Technol. https://doi.org/10.1080/09593330.2017.1332104

Yadav P, Samadder SR (2017b) A global prospective of income distribution and its effect on life cycle assessment of municipal solid waste management: a review. Environ Sci Pollu Res 24(10):9123–9141

Yay ASE (2015) Application of life cycle assessment (LCA) for municipal solid waste management: a case study of Sakarya. J Clean Prod 94:284–293

Zaman AU (2010) Comparative study of municipal solid waste treatment technologies using life cycle assessment method. Environ Sci Technol 7(2):225–234

Zhao Y, Wang H, Lu W, Damgaard A, Christensen TH (2009) Life-cycle assessment of the municipal solid waste management system in Hangzhou, China (EASEWASTE). Waste Manag Res 27:399–406

Zhen-Shan L, Lei Y, Xiao-Yan Q, Yu-Mei S (2009) Municipal solid waste management in Beijing City. Waste Manag 29(9):2596–2599