ISSN 1001-0742
CN 11-2629/X

JES

JOURNAL OF ENVIRONMENTAL SCIENCES

April 1, 2015 Volume 30
www.jesc.ac.cn

MBR in Wastewater Reclamation

Science Press

Sponsored by
Research Center for Eco-Environmental Sciences
Chinese Academy of Sciences

Journal of Environmental Sciences Volume 30 2015

www.jesc.ac.cn

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Environmental impacts of a large-scale incinerator with mixed MSW of high water content from a LCA perspective

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ARTICLE INFO

Article history:
Received 12 June 2014
Revised 13 October 2014
Accepted 29 October 2014
Available online 29 January 2015

Keywords:
Large scale incinerator
Life cycle assessment
Environmental impact
MSW
Sorting system
China

ABSTRACT

Large-scale incinerators are applied widely as a result of the heavy burden of municipal solid waste (MSW) generated, while strong opposition is arising from the public living nearby. A large-scale working incineration plant of 1500 ton/day was chosen for evaluation using life cycle assessment. It was found that the corresponding human toxicity impacts *via* soil (HTs), human toxicity impacts *via* water (HTw) and human toxicity impacts *via* air (HTa) categories are 0.213, 2.171, and 0.012 personal equivalents (PE), and global warming (GW100) and nutrient enrichment (NE) impacts are 0.002 and 0.001 PE per ton of waste burned for this plant. Heavy metals in flue gas, such as Hg and Pb, are the two dominant contributors to the toxicity impact categories, and energy recovery could reduce the GW100 and NE greatly. The corresponding HTs, HTw and HTa decrease to 0.087, 0.911 and 0.008 PE, and GW100 turns into savings of −0.007 PE due to the increase of the heating value from 3935 to 5811 kJ/kg, if a trommel screener of 40 mm mesh size is used to pre-separate MSW. MSW sorting and the reduction of water content by physical pressure might be two promising pre-treatment methods to improve the combustion performance, and the application of stricter standards for leachate discharge and the flue gas purification process are two critical factors for improvement of the environmental profile identified in this work.

Introduction

Municipal solid waste (MSW) disposal leads to a significant environmental burden due to the huge amounts of pollutant emissions. Incineration is regarded as one of the effective ways to minimize waste mass and volume, and has increased from 2.5% (2001) to 19.8% (2011) of total MSW disposal due to the heavy burden of MSW generated as a result of the rapidly increasing urban population and the improvement of people's lifestyles in China. Around 31 million tons of MSW collected was burned in 109 incineration plants in China, with a corresponding total treatment capacity of 94,114 ton/day (National Bureau of statistics of China, 2011).

Large scale incinerator plants are an attractive way to deal with the sharp increase of MSW, and grate firing has been demonstrated to be the most promising type of furnace for non-classified MSW (Shi et al., 2008; Xi et al., 2003). Currently, the technology for large scale incinerators is imported from

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http://dx.doi.org/10.1016/j.jes.2014.10.004
1001-0742/

the developed countries, such as the EU, USA, and Japan, which are designed based on burning of classified refuse with high heating value. Putrescible biodegradable materials predominate in the non-classified refuse in MSW due to the particular lifestyles in some Asian countries, which is characterized as "three high and one low", *i.e.*, high mixture, high inorganic matter content, high percentage of putrescible waste (more than 55% with consequent high moisture) and low calorific heating value (<4000 kJ/kg) (Sun et al., 2008). The application of imported incinerator technology faces an increasingly complex set of environmental and social pressure in China.

Incinerators have created many concerns in the past decade regarding the safeguarding of public health and environmental safety from toxic and cancer-causing emissions, as well as concerns about financial costs (Monni, 2012; Vermeulen et al., 2012), and the concerned people frequently oppose the construction of incineration plants, such as the Nanjing Jiangbei Incineration Plant, Guangdong Panyu Incineration Plant (Zheng, 2009), and Zhejiang Hanzhou Jiufeng Incineration Plant in China. Since the relative environmental impact is still not clear, it is necessary to assess the environmental performance of large scale incineration plants both in qualitative and quantitative terms.

The environmental profiles of grate firing incinerators and fluidized bed incinerators have been evaluated and compared (Chen and Christensen, 2010), and grate firing incinerators were found to result in more savings in terms of global warming potential than fluidized bed incinerators due to their higher net power generation from the combustion of MSW. However, only the energy consumption in the pretreatment process of MSW was considered, and the leachate generation/treatment and mass minimization were missing from the calculation process. It was also found that circulating fluidized bed incineration with auxiliary coal of 700 ton/day was beneficial for mitigating global warming with the addition of sufficient coal (Zhao et al., 2012). The environmental performance of sludge and medical waste incineration was also evaluated, and the results were found to be influenced greatly by the type of furnace and auxiliary resources (Assamoi and Lawryshyn, 2012; Chen and Christensen, 2010). It is necessary and urgent to assess the environmental performance of large scale-incineration plants for mixed MSW with high water content and organic matter as the number of incineration plants applied in China continues to increase.

In this work, the environmental impact of a full scale grate firing incinerator with three lines was examined and assessed using life cycle assessment (LCA). The specific objectives were to answer the following questions: (1) What are the environmental burdens associated with the current large-scale MSW incineration plant? (2) How do the combustion performance and the environment impact vary after application of some feasible supplemental measures, such as the pretreatment of waste and use of an advanced pollution control system?

1. LCA process

1.1. EASEWASTE introduced briefly

The EASEWASTE model (2008 version) Technical University of Denmark, has been developed with a database including waste technologies, recovery and disposal options, as well as external processes that might be included either upstream or downstream in a solid waste management system. The relative waste specific mass flows, resource consumption and environmental emission are considered in the environmental assessment of an incineration plant (Riber et al., 2008). A graphical overview of how the waste sector is modeled in EASEWASTE can be found in Christensen et al. (2007). All the relative environmental impacts

Table 1 – Environmental normalized potential impacts reference in China.

Potential impact category	Normalization reference	Physical basis	References
Global warming	8700	Global	(J.H. Li et al. (2007)
(kg CO_2-eq./person/year)	36	Regional	(J.H. Li et al. (2007)
Acidification	0.20	Global	(J.H. Li et al. (2007)
(kg SO_2-eq./person/ year)	62	Regional	(J.H. Li et al. (2007)
Ozone depletion	0.65	Regional	(J.H. Li et al. (2007)
(kg CFC-11-eq./person/ year)	358	Regional	(J.H. Li et al. (2007)
Nutrient enrichment (kg NO_3^-eq./person/ year)	3.52×10^5	Regional	Wenzel et al. (1997)
Photo-chemical ozone formation (kg C_2H_4-eq./person/ year)	9.64×10^5	Regional	Wenzel et al. (1997)
Human toxicity, soil (m^3 soil/person/year)	5×10^4	Regional	Wenzel et al. (1997)
Ecotoxicity, water chronic (m^3 water/person/year)	6.09×10^{10}	Regional	Wenzel et al. (1997)
Ecotoxicity, soil (m^3 soil/person/year)	140	Local	Wenzel et al. (1997)
Human toxicity, water (m^3 water/person/year)			
Human toxicity, air (m^3 air/person/year)			
Spoiled groundwater resources (m^3 water/person/year)			

and resource consumptions are normalized into the same units according to the modified EDIP standard references (Wenzel et al., 1997), *i.e.*, global data for the global warming impact, Chinese normalization references for the standard impact categories, European normalization references for the toxic categories, and Danish normalization reference for contaminated groundwater. Detailed normalization factors are shown in Table 1.

Five related outputs of the waste/process are included in the incineration module, *i.e.*, bottom ash, fly ash, sludge, iron scrap and wastewater (leachate), with a corresponding generation rate of 20%, 1.4%, 0.3%, 0.5% and 24% of the burned MSW, respectively, according to the one-year statistics data for the working incineration plant. The process-specific emissions were defined separately on the basis of per ton MSW combusted.

The transfer co-efficient of the incineration plant was calculated based on the mass balance according to the practical statistics data. Emissions connected with fly ash, bottom ash and wastewater discharged from the system were defined and calculated by the corresponding transfer coefficient, as shown in Table 2. Bottom ash generated could be routed to construction materials or landfill. Fly ash and sludge are disposed in the hazardous waste landfill, and wastewater is introduced into in-site or ex-site wastewater treatment system.

Electricity generation is the main revenue for the incineration plants, since district heating is usually not available in most southern cities in China. Nowadays, 82% of electricity is generated from coal combustion, and the energy substituted is chosen according to the coal energy production process (Statistics and Information Department of China Electricity Council, 2008). The avoided emissions from external energy production are subtracted from the emissions occurring at the local coal energy power plant using LCA-modeling.

1.2. LCA system boundary for incineration plant

The module was established based on the practical data collected from the working incineration plant in Shanghai, China. The system boundary starts from the point of MSW arrival and ends when the byproducts of bottom ash/fly ash/leachate leave the incineration plant, as shown in Fig. 1. Both the directly and indirectly emitted pollutants and the avoided impacts are considered. The diesel oil, activated carbon and chemical compounds used are specified per ton MSW burned. Emissions associated with the manufacture of the incineration plant are excluded from this analysis. The electricity consumption in the supplemental processes, *i.e.*, waste separation process using trommel screener, de-watering system, and leachate treatment system in the municipal wastewater treatment plant (MWWTP), are considered here.

Table 2 – Transfer coefficient of waste burning in incineration plant.

	Air emission	Bottom ash	Fly ash Activated particle carbon	Waste water	Iron scraps
H_2O	75.95%			24.05%	
Ash		91.31%	8.69%		
VS		35.97%	51.72%	12.31%	
Ca		91.31%	8.69%		
As		73.37%	26.48%	0.15%	
Cd	0.03%	50.71%	49.19%	0.07%	
Cr		59.35%	40.63%	0.02%	
Cu		82.07%	12.15%	0.01%	5.77%
Hg	3%	3.44%	93.5%	0.059%	
Mg		91.31%	8.69%		
Mn		83.5%	7.3%	0.08%	9.122%
Ni		91.31%	8.69%		
Pb	0.02%	66.65%	33.21%	0.12%	
Zn		82.62%	17.36%	0.023%	

All data were calculated based on the regular monitoring data on a working incineration plant in Shanghai, China.

1.3. Waste composition

Waste was collected from the waste storage center in the local resident community in the autumn season in Shanghai, and the corresponding fractions and waste compositions after trommel separation are shown in Table 3. This enables the calculation of emission factors specific to the waste flow in question.

1.4. Basic operation data for the incineration plant

1.4.1. Resource and energy consumption

The consumed items during the incineration process include diesel fuel, electricity and chemicals. Electricity is applied in the pollution control and feeding systems, which could be self-service. The amounts of auxiliary resources/energy used were 0.2628 kg/ton MSW of diesel oil for the burner system, 5.79 kg/ton MSW of $Ca(OH)_2$, 18.4 kg/ton MSW of CaO, 0.208 kg/ton MSW of activated carbon for the flue gas cleaning system, 0.28 m^2/ton MSW of membrane and 0.712 kg/ton MSW of flocculant for the leachate treatment system, respectively. Energy is exploited as the steam used in the district electricity production, with a range of 210–310 kWh/ton MSW generated depending on the waste composition and operation conditions.

1.4.2. Flue gas generation and upgrading process

Semi-dry spraying method/activated carbon adsorption/bag filter scrubber systems are used in the flue gas cleaning system. About 3748–4286 Nm^3 of gas emission is released per ton MSW burned, and the corresponding NO_x, SO_2, HF, HCl, CO, Dust and CO_2 concentrations in final outlet flue gas are shown in Table 4. The average concentration values were used in the life cycle inventory model for the controlled stack emissions (NO_x, SO_2, *etc.*), and the SO_2 concentration was two times higher than that of HCl and HF. Usually, the exhaust gas cleaning system does not influence CO_2 emissions (Heron and Søren, 2007), and it is therefore common to differentiate CO_2 emissions based on waste composition only, with 0.40–0.68 ton CO_2 generated per ton waste incinerated.

1.4.3. Leachate effluents

MSW is stored in the storage bunker for 3 days before feeding into the furnace, and around 20%–30% of total MSW (weight (wt)) converts into leachate, with more leachate generated in the summer season due to the waste fruits included (Zhang et al., 2010). Leachate is not allowed to spray back into the burning system, as is the practice in the developed countries,

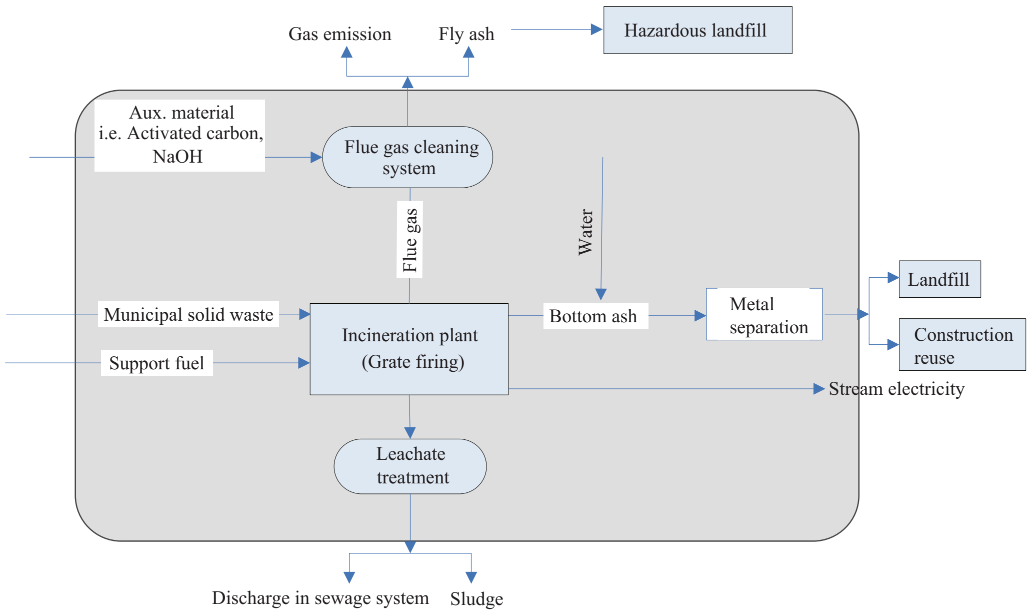

Fig. 1 – System boundary of the modeling incineration plant.

due to the lower heating value of MSW, and use of an on-site leachate treatment system and input to the municipal wastewater collection system are the two feasible options for leachate. In this incineration plant, a leachate treatment system of Anoxic–Oxic-Membrane Bioreactor followed by ultra-filtration is applied, and around 1.93 kWh/ton leachate of electricity and some chemicals, *i.e.*, 0.712 kg/ton leachate of flocculant and 0.105 kg/ton leachate of $Ca(OH)_2$ are consumed. The compounds emitted in effluent are expressed as kg/ton MSW, as following (unit; g/ton MSW) CODcr 179, $BOD_5$15, SS 3.3, TN 21.9, NH_3–N10.2, TP 0.4, Pb 0.1, Zn non detection, Cr 0.2, Cu 0.1, Cd 0.3, Hg non detection, Ni 0.1, Cl 1198.2, SO_4^{2-} 523.1, and TOC 21.8; and the total pollutants discharged are calculated at the point when leachate is discharged from the incineration plant.

1.4.4. *Bottom ash and fly ash*

Fly ash is another predominant pollutant, although only 1.42%–2.53% (wt.%/wt.%) of MSW is collected as fly ash in this incineration plant, and the bottom ash is around 25%–30% of total MSW burned. The composition of bottom ash including the following parts: Ceramics (7.33%–7.36%), glass (6.81%–7.37%), molten slag (64.09%–66.42%), stone (14.02%–16.62%), organic matter (0.78%–1.24%), metals (3.67%–4.10%) and others (0.08%–0.11%). The leaching toxicity of heavy metals from fly ash and bottom ash is higher, compared to the limitation value in the Identification standard for hazardous wastes—Identification for extraction toxicity GB5085.3-2007. Fly ashes are eventually disposed in hazardous waste landfill. Bottom ash was formerly disposed of in the normal landfill, and now is expected to be recycled in municipal pavement materials.

Table 3 – Distribution of waste composition in raw MSW and the different size ranges.

Composition	Raw waste	>80 mm	40–80 mm	>40 mm
Percentage	100%	24.8%	20.0%	45.3%
Organic matter	70.6%	27.51%	63.06%	44.07%
Plastic	12.8%	34.90%	15.97%	25.84%
Paper	7.3%	12.55%	15.36%	13.78%
Textile	3.2%	9.21%	4.85%	6.43%
Resident	2.4%	6.32%	3.14%	4.28%
Wood	0.1%	0.30%	0.19%	0.23%
Metals	0.3%	4.61%	0.15%	2.50%
Rubber	0.2%	0.00%	0.02%	0.01%
Soil	0.1%	0.00%	0.45%	0.17%
Glass	3%	4.61%	0.63%	2.71%

Table 4 – Average amounts of flue gas emission per ton incinerating MSW.

	Average amounts per ton MSW	GB18485-2001
CO_2 [a]	0.40–0.68 ton/ton	–
NO_x	1.09–1.34 kg/ton	400 mg/m^3
SO_2	0.30–0.44 kg/ton	260 mg/m^3
HF	0.37–0.60 g/ton	–
Dust	10.11–17.17 g/ton	80 mg/m^3
HCl	0.14–0.18 kg/ton	75 mg/m^3
CO	0.03–0.63 kg/ton	150 mg/m^3
Hg	1.00 g/ton	0.20 mg/m^3
Cd	0.50 g/ton	0.10 mg/m^3
Pb	8.00 g/ton	1.60 mg/m^3
Dioxin [b]	1.59×10^{-9} kg/ton	1.0 ng TEQ/Nm^3

[a] The fossil-CO_2 in the calculation process was assumed to be 0.20 ton/ton MSW, and the total CO_2 was 0.56 ton/ton MSW, thus the biological CO_2 was 0.36 ton/ton MSW.
[b] The average dioxin concentration from 16 survey incineration plant, with the value of 0.432 ng TEQ/Nm^3 (Ni et al., 2009).
[c] Standard for pollution control on the municipal solid waste incineration GB18485-2001.

2. Results and discussion

2.1. Inventory and normalization process

The environmental impact from the incineration plant is the result of a complex calculation of all of the related anthropogenic activities and is strongly related to the type of environmental effect considered. Amounts of green house gas emissions (the main contributors are CO_2, hydrocarbons (HC) and CO) released from the incineration plant, with a value of 314.1 kg CO_2-eq. per ton waste, and electricity recovery are assumed to substitute the electricity generated from coal in China, which could save around 293.2 kg CO_2-eq. per ton waste.

As for nutrient enrichment (NE), total nitrogen (TN) and P to fresh water are the main contributors, with the values of 0.090 and 0.045 kg NO_3^--eq. For acidification (AC), HCl to air and NH_3–N are the main sources, at the rate of 0.125 and 0.047 kg SO_2-eq. per ton waste. In terms of the toxicity categories, Pb and Hg to air are the two main contributors for human toxicity, air (HTa) impact, with the values of 7.679×10^8 and 6.567×10^6 m^3 air per ton of waste according to the life cycle inventory. The discharge of heavy metals significantly contributed to human toxicity, water (HTw) simultaneously, *i.e.*, Hg to air and Cd to fresh water at 1.1×10^5 and 733.4 m^3 water per ton waste. Hg and Pb to air emission are the main sources for human toxicity, soil (HTs) and ecotoxicity, soil (ETs), with the values of 80.050 and 0.635 m^3 soil per ton waste and 5.248 and 0.082 m^3 soil per ton waste, respectively.

The normalization potential impacts were calculated and the illustrated results are shown in Fig. 2. HTw, HTs and HTa contribute to the damage to human health from the incineration plant, with the value of 2.171, 0.213 and 0.012 personal equivalents (PE), and heavy metals in flue gas, such as Hg and Pb, are the dominant contributors. GW100 and NE are the two other main impacts of concern, with the values of 0.002 and 0.001 PE per ton MSW, respectively, because the putrescible food waste predominates in the waste composition and most of it is composed of biological carbon. For AC, a negative impact of −0.037 PE is found due to the reduction of SO_2 through the recovery of electricity generation. 0.116 PE of ETwc is obtained due to the release of dioxin and heavy metals, and the removal of heavy metal and dioxin are the two main targets for the reduction of the toxicity environmental impact. Flue gas purification and energy recovery in incinerators are considered two key parameters for the improvement of the environmental impacts and should be considered seriously.

It can be seen that the normalization results are different from those in the report by Chen and Christensen (2010). The feed waste composition, transfer co-efficient and operation conditions in the incineration plants all might influence the LCA results. The waste composition might be one of the key factors, since food waste made up around 79.6% of total waste in this scenario, while only 22.1%–37.4% of the waste composition was food waste in the report of Chen and Christensen (2010). Generally, food waste is the main composition in mixed collected municipal solid waste in Shanghai, China. Thus the corresponding heating value is totally different, which resulted in a different burning performance in the incineration plant. Chen and Christensen (2010) claimed that leachate could be sprayed back to the furnace for evaporation as an alternative method for leachate treatment without major changes in the environmental profile, while this will destroy the burning system in this scenario. Waste composition is the critical factor for the burning performance through its effect on low heating value (LHV), and the effect of the waste composition on the environmental performance of the incineration plant should be assessed. Meanwhile, it has been claimed that a fluid bed incinerator with sufficient coal presents a significant benefit in mitigating global warming, whereas the incineration with a mass of coal can avoid more impacts to acidification, photochemical ozone and nutrient enrichment because of the increased electricity substitution and reduced emission from coal power plants (Zhao et al., 2012). However, it is also reported that the electricity generated from coal combustion in a fluid bed incinerator plant has a larger environmental load than that avoided from large dedicated coal-fired power plants, since the energy efficiency in the incinerator plant is lower than that from coal-fired power plants (Chen and Christensen, 2010), although more electricity could be generated from a fluid bed incinerator with the addition of more coal. Therefore, supplemental coal or oil is not considered to generate more electricity recovery in grate firing incinerators.

2.2. Potential improvement process of environmental profiles

The low heating value in MSW is the main barrier to good combustion performance in the incineration plant, and the lack of a qualified pollutant control system, such as for leachate and flue gas treatment, also leads to a high environmental load.

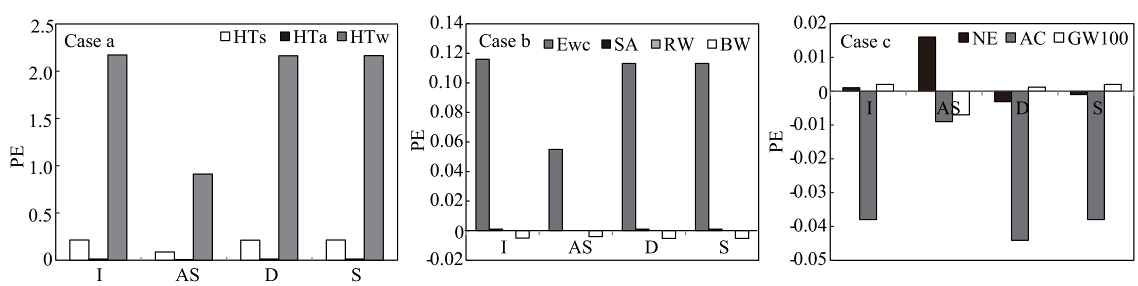

Fig. 2 – Environmental impact of incineration plant with three different cases. "I" means the case of the real working incineration plant; "AS" means the case after sorting; "D" means the case of dewatering; "S" means the case of the application of the stricter pollution control.

To maintain the clarity and pertinence of the assessment, three scenarios with potentially critical assumptions were considered, including increasing the heating values of waste by sorting, the dewatering process of the waste, and the reduction of pollutants released from the incinerator. It should be pointed out that the first two pretreatments are still not part of current practice at real incinerators, while they might be considered in the near future, with the implementation of the stricter Standard for pollution control on the municipal solid waste incineration GB18485-2014. The environmental profiles are also shown in Fig. 2.

2.2.1. Scenario-I: application of waste sorting system

Collection of non-classified refuse is currently practiced in China, and the LHV of the mixed MSW is lower than the basic requirement of 4000 kJ/kg (Sun et al., 2008). To improve the heating value of MSW, a sorting process might be the method of choice for the improvement of the combustion waste according to the test conditions.

A light fraction of high calorific value can be achieved by rotating trommel screens with the screen sizes of 40 and 80 mm according to our practical experience (B. Li et al., 2007). Three fractions with different size ranges and waste compositions, namely, the fractions of >80, >40 and <40 mm, were obtained according to our previous work (as shown in Table 3). Around 24.8% of the total MSW could be present in the fraction with a mesh size >80 mm, and the fraction with the mesh size >40 mm makes up around 45.3% of the total MSW. The fraction corresponding to the 40 mm screen underflow is assumed to be landfilled. The application of >40 mm fractionation might be the most feasible sorting operation process from the economy and efficiency perspective. Some inner materials, *i.e.*, the glass and metals, could also be removed simultaneously, with a percentage of 11.6% (B. Li et al., 2007). The residues of <40 mm made up about 43.2% of the total MSW, and over 80% of the biodegradable fraction was found in the 40 mm screen underflow, while the percentage of organic matter decreased from 70.6% to 45.5% after sorting on the 40 mm screen overflow, and the percentage of the main high heating value contributors, plastic and paper, increased from 12.8% to 25.8%, and 7.3% to 13.8% (Chen et al., 2007). The estimated heating values in raw waste, >40 mm fraction and >80 mm fraction are 3935, 5811 and 7284 kJ/kg, respectively, and thus the introduction of the larger size fraction might result in better combustion performance.

The environmental impacts before (all MSW treated by incineration) and after sorting (MSW on the 40 mm screen overflow treated by incinerating and that on the 40 mm screen underflow by landfilling) are shown in Fig. 2. GW100 impact changed from load (0.002 PE) into saving (−0.007 PE) as the waste size decreased from the raw MSW to the size fraction >40 mm. CH_4, CO_2 and CO are the main contributors, with the value of 57.28, 24.21 and 0.34 CO_2-eq., since some bio-carbon in MSW with size <40 mm converts to CH_4 in the landfill, and emission controls for CH_4 in landfill will result in significant benefit to the environment. NE impact increased from 0.001 to 0.016 PE per ton MSW, with NH_3 to the marine water and TN to the fresh water being the two main sources, with 1.452 and 0.036 kg NO_3-eq., respectively.

AC credit decreases from 0.038 to 0.010 PE per ton MSW, and NH_3, H_2S, and HCl are the first three contributors, with 0.750, 0.250, and 0.053 kg SO_2-eq. For toxicity categories, there is a decrease from 0.012 to 0.008 PE, 2.171 to 0.911 PE, and 0.213 to 0.087 PE, in terms of HTa, HTw and HTs, respectively. The observed increase of NE and AC is due to the residues of <40 mm landfilling, and more NH_3 in the leachate goes to the groundwater and air in the scenario after sorting. Meanwhile, less impact is found in HTs, EWs, HTa, and GW100, since there is a better burning performance brought about by the increase of waste LHV after sorting.

2.2.2. Scenario-II: application of the dewatering system

A dewatering process is helpful in reducing the high water content in unclassified collection MSW, and 5% of the water content could be removed by physical pressure after anaerobic digestion in the storage tank according to our experience, such that the estimated heating value increases from 3936 to 4186 kJ/kg after dewatering. The electricity consumption for the dewatering system is assumed to be 1 kWh/ton, and the impact of GW100 decreases from 0.002 to 0.001 PE. CO_2, HC and CO are still the main sources for GW100, although the corresponding green house gas decreases greatly, with the value of 20.96, 0.29 and 0.03 kg CO_2-eq. per ton waste, while that in the raw MSW is 31.58, 0.29, and 0.03 kg CO_2-eq. per ton waste. AC credit decreases from −0.037 to −0.044 PE per ton MSW, and HCl and NH_4–N are the two greatest contributors, with the value of 0.125 and 0.010 kg SO_2-eq. With regard to NE, it changes from a load (0.001 PE) to savings (−0.003 PE), and TN to the fresh water and NH_3 are the predominant sources of 0.011 and 0.009 kg NO_3-eq. In terms of the toxicity categories, Human toxicity impacts are almost the same, since most of the toxic substances, such as Hg, Pb and Cd, are still present in MSW, and will be released in the flue gas during the burning.

2.2.3. Scenario III—the improvement of the pollution control process

The reduction of leachate effluent and dioxin concentration are the critical issues for the improvement of the pollution control system in the incineration plants. Leachate could be introduced into the local MWWTP through the sewage collection system nearby, as incineration plants usually are located in a high density population area, where a sewage collection system is available. The effluent from MWWTP is assumed to meet the sewage water national discharge standards (discharge standard of pollutants for municipal wastewater treatment plant GB 18918-2002), and the corresponding pollutant emissions are expressed as kg/ton MSW, as following (unit; g/ton MSW) CODcr 13.6, BOD_5 2.7, SS 2.7, TN 4.08, NH_3–N 2.17, TP 0.27, Pb 0.02, Cr 0.02, Cu 0.1, Cd 0.2, Hg non detection, Ni 0.01, Cl 1198.2, SO_4^{2-} 523.1, and TOC 2.8.

Dioxin in flue gas is another concern for the local residents. The average dioxin value is supposed to decrease from 0.43 ng TEQ/Nm^3 to the EU discharge standards, with the value of 0.1 ng TEQ/Nm^3 (Ni et al., 2009). The environmental performance before and after the upgrading of the exhausted flue gas and leachate effluent is shown in Fig. 2. It can be seen that AC and NE impact decrease greatly when the leachate effluent meets the discharge standard of GB 18918-2002, and the corresponding NE and AC decrease from 0.001 to −0.001 PE, and

−0.037 to −0.038 PE per ton MSW, because N and P concentrations in the MWWTP effluent are lower than that from the current Anoxic–Oxic-Membrane Bioreactor/ultra-filtration treatment system applied. The toxicity categories, *i.e.*, HTw, and ETwc, are also found to decrease from 2.171 to 2.165 PE and 0.116 to 0.113 PE, respectively, meaning that the increase of the dioxin removal efficiency is important for the incineration plant. Dioxin is therefore one of the critical gas emissions, especially for the toxicity categories, and should be monitored regularly. Meanwhile, categories that do not change are those that are negligible in general or are not affected by variations in flue gas emissions and leachate effluent. In general, dioxin emission variations (even within the legal lower limits) will influence the environmental profile greatly, and qualified flue gas scrubbing facilities should be ensured.

Generally, all of these three methods might improve the environmental impact of incineration plants, since both the waste sorting and dewatering processes could contribute to the increase of heating value for the input waste, which results in better burning performance. Both the introduction of leachate into MWWTP and the improvement of the exhaust gas cleaning system will reduce the pollutants released to the surrounding environment. Therefore, all of these measures combined could reduce the environmental impact greatly, and should be considered together in practical projects.

3. Conclusions

The environmental profile of a large scale grate firing incinerator was evaluated by the LCA model of EASEWASTE, and HTs, HTw and HTa categories were 0.213, 2.171, and 0.012 PE, and GW100 and NE impacts were 0.002 and 0.001 PE per ton waste burning. Potential improvement processes were also proposed for the mixed, non-classified MSW burning, which would result in good environmental performance. The introduction of a MSW sorting system using a trommel screener of 40 mm mesh size and the removal of water content by physical pressure might be two promising pre-treatment methods, although both are still not practiced due to the required cost investment or other reasons, and the environmental performance is sensitive to these processes. The application of the stricter leachate discharge effluent standard and the reduction of the dioxin concentration are two critical factors for the improvement of the environmental profile in the incineration plant, which have been implemented in the practical project, and the corresponding NE and AC decreased from 0.001 to −0.001 PE, and −0.037 to −0.038 PE per ton MSW. HTw and ETwc, were also found to decrease from 2.171 to 2.165 PE and 0.116 to 0.113 PE, respectively.

Acknowledgments

This work was supported by the National Natural Science Foundation of China (Nos. 41173108, 51278350), the Alexander von Humboldt Foundation (2010), Shanghai Rising-Star Program (14QA1402400), Key project of Science and Technology Commission of Shanghai Municipality (No. 13DZ0511600), and National Key Technology R&D Program (No. 2014BAL02B03-4).

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Journal of Environmental Sciences (Established in 1989)
Volume 30 2015

Supervised by	Chinese Academy of Sciences	**Published by**	Science Press, Beijing, China
Sponsored by	Research Center for Eco-Environmental		Elsevier Limited, The Netherlands
	Sciences, Chinese Academy of Sciences	**Distributed by**	
Edited by	Editorial Office of Journal of	Domestic	Science Press, 16 Donghuangchenggen
	Environmental Sciences		North Street, Beijing 100717, China
	P. O. Box 2871, Beijing 100085, China		Local Post Offices through China
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Editor-in-chief	X. Chris Le	**Printed by**	Beijing Beilin Printing House, 100083, China

CN 11-2629/X **Domestic postcode: 2-580** **Domestic price per issue RMB ¥ 110.00**

ISSN 1001-0742