Effect of waste incineration and gasification processes on heavy metal distribution

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ABSTRACT

This study investigates the distribution of heavy metals during incineration and gasification. Because heavy metals (Cr, Pb, Cd) become volatile at high temperatures, their concentrations in fly ash increase as the operating temperature of incineration/gasification increases. By contrast, the efficiency of the bed-material mixing and the contact between the bed material and the heavy metals inside the fluidized bed increase when the operating-gas velocity increases; hence, the heavy-metal content in the bed material also increases. Our results show that the heavy-metal content in the fly ash increases with the bed-material particle size. The probability of contact between the heavy metals and the bed material is reduced at higher bed-material particle sizes; hence, the emission concentration of heavy metals increases. The heavy-metal concentration in the fly ash after gasification is approximately 1/10–1/30 of that after incineration. However, the proportion of the heavy metals trapped in the bottom ash after gasification is higher than that after incineration under any of the operating conditions, regardless of the volatility of the heavy metals. Overall, gasification lowers the emissions of heavy metals, but traps more heavy metals in the bottom ash than incineration.

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1. Introduction

Incineration is the most widely used waste-treatment method in Taiwan at present. Incineration is defined as a fast oxidation reaction between waste and air at high temperature, completely combusting the organic matter and converting the incombustible material into stable inorganic matter. Incineration technology is characterized by volume reduction, stabilization, pathogen elimination, and recycling. Because the reaction time is short, the waste can be treated rapidly. Although the heat generated after combustion can be recycled in the course of incineration operations, the recovery efficiency at present is relatively low, and the storage of the generated energy is difficult. Therefore, it has limited applicability [1,2].

In contrast, gasification is based on the reaction of biomass under a limited oxygen supply to generate biomass energy in the form of a combustible gas. The mixed gas after gasification (synthesis gas) is composed of CO, CH$_4$, and H$_2$, and it needs to be purified before use. After purification, the synthesis gas can be simply burned off. A better alternative is to use the synthesis gas for the production of industrial chemicals, or for power generation in gas engines and turbines. Therefore, gasification also reduces the volume of waste, while generating fuel gases that are convenient for storage and use, providing a means to replace the incineration process [3].

Besides combustible constituents, the waste also contains heavy metals. Many researchers [4–8] have indicated that the major heavy metals in waste are Zn, Cr, Cu, Pb, Ni, Hg, and Cd, etc. in the incineration process. These heavy metals can be released into the environment after high-temperature incineration or gasification. Numerous studies have discussed the migration of heavy metals during combustion. Upon heating, the heavy metals in the waste form volatile metallic vapor or particles, which enter the environment together with flue gas, or fly/bottom ash. The migration behavior depends on the physical and chemical properties of the heavy metals or their compounds formed during the treatment. Reimann [9] indicated that the heavy-metal distribution during the incineration process was related to the characteristics of compounds and boiling points of heavy metals. High-boiling metals such as Cr mostly exist in the bottom ash, while highly volatile heavy metals such as Cd mostly form vapor and leave the combustion zone with the flue gas. Obernberger et al. [10] also indicated that more volatile heavy metals such as Zn and Cd would tend to vaporize and recondense to form fine particles.

Barton et al. [11] showed that less volatile heavy metals only partially participated in chemical reactions in the combustion environment, and were not volatilized; instead, they became part of the bottom ash. By contrast, highly volatile heavy metals reacted with other materials in the incineration system to produce high-boiling species (e.g., metal oxides), which also remain in the bottom ash. In addition, some of these volatile heavy metals or metal compounds formed vapors at high temperature, whereas the high-boiling heavy metals may have reacted with other species to form highly volatile compounds (e.g., metal chlorides), leaving the combustion zone in flue gases.

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Besides the characteristics of heavy metals, factors such as the physical and chemical compositions of waste and chlorine-containing species, and operating factors such as combustion temperature, operating-gas flow, feed load, moisture levels and flue-gas-treatment technique are other major factors influencing the heavy-metal distribution. For operating parameters, Hiraoka and Takeda [12] and Gerstle and Albrinck [13] demonstrated that the combustion temperature could influence the distribution ratio of heavy metals in the bottom ash, and the abundance of Zn, Pb, and Cd in the bottom ash decreased as the temperature increased, whereas that of As, Cd, Hg, Zn, and Pb in the flue gas increased. Additionally, the moisture in the waste can affect the elemental distribution of heavy metal. Meng, et al. [14] indicated that the volatilizations of Pb, Zn, and Cd were enhanced by increasing the moisture in MSW. Separately, Li et al. [15] found that increased moisture content increased the Pb, Zn, Cu, and Cd concentrations in the bottom ash.

In a further study on operating parameters, Wey et al. [16] showed that the bed material of a fluidized bed trapped some heavy metals during the combustion process, and the relative concentration of heavy metals in the bed material was in the order Pb > Cr > Cd. The elemental distribution of various mixtures can be also influenced by other factors such as the incineration-operating temperature, the presence of other elements (O, Cl, S), and air quantity, besides the characteristics of heavy metals. Therefore, the existence of chloride and sulfide will significantly affect the elemental distribution of heavy metals during combustion. Wang et al. [17] demonstrated that plastic and food residues in urban waste influenced the amount of metal chlorides discharged after incineration. Li et al. [15] also illustrated that the volatility of Pb, Zn, and Cu decreased as the flue gas moisture increased in the presence of chlorine, because of the transformation of the more volatile metal chlorides into less volatile metal oxides. For Cd, the existence of chlorine promotes its volatility. Additionally, Zhang et al. [18,19] showed the existence of elemental sulfur or sulfide will increase the retention of Cd on bottom ash, because of formation of CdS in the reactor.

The combustion system used for gasification is an oxygen-deficient environment; therefore, the distribution and incineration of heavy metals may be different from that of gasification, which is usually conducted in an excess of air. However, previous studies have seldom compared the effects of the operating conditions of incineration with those of gasification on heavy-metal distribution. Therefore, this study investigates the influence of operating parameters for incineration and gasification (temperature, gas velocity, and bed-material particle size) on the emission vs. bed-material retention of heavy metals. The direct comparison of the two treatment processes is intended to facilitate the best choice of future waste-disposal technology.

2. Materials and methods

2.1. Materials

The simulated waste used in this study consisted of wood chips and polypropylene (PP), covered with polyethylene (PE) plastic bags. A solution (1 mL) of $Cr(NO_3)_2$, $Pb(NO_3)_2$, and $Cd(NO_3)_2$ each (0.025 g heavy metal/mL) was added. Cr, Pb, and Cd were chosen to represent heavy metals of low, moderate, and high volatility, respectively. The ultimate, proximate, and heat value analyses of these materials are shown in Table 1, and the simulated waste feed rate in the experiment is shown in Table 2.

2.2. Instruments

A bubbling fluidized-bed reactor was employed, as shown in Fig. 1. The reactor was made of stainless steel (AISI 310), and it had a main chamber measuring 120 cm in height and 10 cm in inner diameter. The bottom of the chamber was equipped with a stainless-steel dispersion plate (open area, 15.2%). An electrical heating system, covered with ceramic fiber for heat insulation, was mounted outside the stainless steel

Table 1
Ultimate analysis, proximate analysis, and heat value analysis of artificial waste.

Species	Wood chips	Polypropylene (PP)	Polyethylene (PE)
Ultimate analysis (wt.%)			
C	43.12	86.16	85.71
H	5.80	12.20	13.04
O	46.07	0.52	0.39
N	5.01	1.12	0.86
Proximate analysis (wt.%)			
Moisture	0.74	0.01	0.01
Volatile matter	81.6	99.99	99.99
Fixed carbon	16.6	0	0
Ash	1.07	0	0
LHV (MJ/kg)	15.10	45.63	44.92

tube. The outermost shell was fixed by a stainless steel casing, and a temperature feedback control system (PID), and three thermocouples were used for system temperature control. A bivalve feed orifice was used to prevent the synthesis gas that was generated in the chamber from leaking during the feeding process and to prevent air from entering the chamber. Finally, a cyclone collector at the air outlet was connected in parallel with a bag filter-type collector to collect the fly ash.

2.3. Experimental procedures

Silica sand (particle size, 550 μm; density, 2600 kg/m³) was used as the bed material. The minimum fluidization velocity was measured before the experiment, as described by Lin et al. [20]. When the sand bed was heated and the preset temperature stabilized, the blower was switched on to feed the air, and the air-flow was regulated using a float-type flow meter. The simulated waste was fed into the chamber at a feeding rate of 1 pack/20 s through the feed inlet. The sample gas was allowed to flow through a filter device made of glass fibers to trap the fly ash in the GF/A filter paper (Cat. No. 1820 047). Thereafter, the gas was passed through a low-temperature water tank charged with a heavy-metal-absorption liquid (100 mL), containing H_2O_2 (SHOWA, 30%, reagent grade; 33 mL) and HNO_3 (Scharlau, 65%, reagent grade; 7.67 mL) in deionized water (total volume, 100 mL), loaded in an impinger to absorb the volatile heavy metals. The samples were taken at 4-min intervals, and the operating conditions for each experiment are shown in Table 2. Upon completion of the experiment, the bottom ash was cooled, collected, and analyzed using an ASTM standard sieve (Nos. 30, 35, 40, and 45) to evaluate the change in the bed-material particle size.

To analyze the heavy-metal concentration, microwave digestion (CEM MARS Xpress) was used to pre-treat both the bottom and fly ashes. The US EPA method 3052 was employed to set the microwave oven digestion program. Firstly, the ash sample (0.5 g) or GF/A filter paper (fly ash), 9 mL concentrated nitric acid and 3 mL concentrated

Table 2
The composition of artificial waste in incineration/gasification processes and operating conditions for each experiment.

Incineration: Total weight (2.33 g)
Wood chips (1.26 g), PP (0.09 g), PE (0.24 g), water solution (1 mL)

RUN	Temperature (°C)	Gas velocity (m/s)	Sand material size (mm)
1, 2, 3	700, 800, 900	0.083	0.550
4, 5	800	0.070, 0.096	0.550
6, 7, 8	800	0.083	0.390, 0.460, 0.650

Gasification: Total weight (8.53 g)
Wood chips (6.50 g), PP (0.60 g), PE (0.43 g), water solution (1 mL)

	Temperature (°C)	Gas velocity (m/s) (ER = 0.3)	Sand material size (mm)
9, 10, 11	700, 800, 900	0.083	0.550
12, 13	800	0.070, 0.096	0.550
14, 15, 16	800	0.083	0.390, 0.460, 0.650