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Repercussions of clinical waste co-incineration in municipal solid waste incinerator during COVID-19 pandemic

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ABSTRACT

During coronavirus disease 2019 pandemic, the exponential increase in clinical waste (CW) generation has caused immense burden to CW treatment facilities. Co-incineration of CW in municipal solid waste incinerator (MSWI) is an emergency treatment method. A material flow model was developed to estimate the change in feedstock characteristics and resulting acid gas emission under different CW co-incineration ratios. The ash contents and lower heating values of the feedstocks, as well as HCl concentrations in flue gas showed an upward trend. Subsequently, 72 incineration residue samples were collected from a MSWI performing co-incineration (CW ratio <10 wt%) in Wuhan city, China, followed by 20 incineration residues samples from waste that were not co-incineration. The results showed that the contents of major elements and non-volatile heavy metals in the air pollution control residues increased during co-incineration but were within the reported ranges, whereas those in the bottom ashes revealed no significant changes. The impact of CW co-incineration at a ratio <10 wt% on the distribution of elements in the incineration residues was not significant. However, increase in alkali metals and HCl in flue gas may cause potential boiler corrosion. These results provide valuable insights into pollution control in MSWI during pandemic.

1. Introduction

Coronavirus disease 2019 (COVID-19) is an acute respiratory disease that emerged in late 2019, resulting in a pandemic (Zhou and Shi, 2021). According to the World Health Organization report, COVID-19 has affected 223 countries with around 174 million confirmed infected cases and 3.7 million deaths until June 11, 2021 (World Health Organization, 2020). During such an outbreak, a substantial amount of clinical waste (CW) has been generated (Prata et al., 2020). Due to the vast consumption of clinical resources and personal protective equipment to mitigate the pandemic, the CW generation rates of confirmed and suspected cases have been predicted to be 3.2 and 1.8 kg/(bed × day), respectively, which are 4- and 2-fold higher than those of patients with other diseases (0.8 kg/(bed × day)) (Wang et al., 2021). For example, the daily generation of CW in Wuhan city in March 2020 was 5-fold higher than that in March 2019 (You et al., 2020). In addition, domestic waste produced by emergency medical centers and quarantine points are often processed as CW owing to their potential infection hazard (Yang et al., 2021). A previous investigation indicated the strong stability of coronavirus on the surfaces of plastics and stainless steel for up to 72 h, and on cardboard surfaces for up to 48 h (van Doremalen et al., 2020). Therefore, environment-friendly treatment and disposal of CW is essential to protect human health during COVID-19 outbreak.

The sharp increase in CW generation threatens to overwhelm local treatment capacity. To address the overflow of CW, co-incineration in municipal solid waste incinerator (MSWI) is an emergency option and a feasible choice because grate incinerator with high heat transfer rate and operating temperature (> 850 °C) guarantees reliable destruction of pathogens (Neuwahl et al., 2010). At the same time, the decline in the generation of municipal solid waste (MSW) during the pandemic provides sufficient CW treatment capacity.

Nevertheless, information about the risks arising from co-incineration, such as production and emission of secondary pollutants is limited. The nature and composition of CW significantly vary from those of MSW, and specific CW often contains materials with high lower heating values (LHV) and ash contents (e.g., metals) (Neuwahl et al., 2010). As the nature and composition of CW significantly vary from those of MSW, and specific CW often contains materials with high lower heating values (LHV) and ash contents (e.g., metals) (Neuwahl et al., 2010). As the nature and composition of CW significantly vary from those of MSW, and specific CW often contains materials with high lower heating values (LHV) and ash contents (e.g., metals) (Neuwahl et al., 2010). As the nature and composition of CW significantly vary from those of MSW, and specific CW often contains materials with high lower heating values (LHV) and ash contents (e.g., metals) (Neuwahl et al., 2010).
incineration ratios were calculated as follows: 

\[ w_{\text{wet}} = \left(1 - \frac{R_{\text{w}}}{100}\right) \times \sum_{i} \left(\frac{M_{\text{w}}^{\text{w}}}{100} \times F_{\text{w}}^{i}\right) + \frac{R_{\text{w}}}{100} \times \sum_{z} \left(\frac{C_{\text{w}}^{\text{w}}}{100} \times F_{\text{w}}^{z}\right) \]  

(2)

where \( E_{\text{w}}^{i} \) and \( E_{\text{w}}^{z} \) are the contents of element \( x \) in organic component \( f \) on a wet basis and dry basis, respectively (%); \( M_{\text{w}}^{\text{w}} \) is the moisture content of organic component \( f \) (%); \( x \) refers to C, H, O, N, S, Cl; \( f \) refers to the organic component \( y \) in MSW (food waste, wood waste, paper, textile, plastic, and rubber) or the organic component \( z \) in CW (plastic, paper, textile, and food waste); \( F_{\text{w}}^{i} \) is the content of element \( x \) in CIWF on a wet basis (%); \( R_{\text{w}} \) is the ratio of CW co-incineration, i.e., 0%, 5%, 10%, 15%, and 20%; \( M_{\text{w}}^{\text{w}} \) is the percentage of organic component \( y \) in MSW (food waste, wood waste, paper, textile, plastic, and rubber) in MSW (%); and \( C_{\text{w}}^{\text{w}} \) is the percentage of organic component \( z \) in CW (plastic, paper, textile, and food waste) (%). Glass, metals, and other incombustible substances (Lin et al., 2015) were treated as ash in CIWF.

Similar calculations were conducted for determining the moisture and ash contents, as well as LHV of CIWF based on the physical compositions of MSW and CW. LHV of CIWF were calculated based on the elemental compositions using Dulong formula (see Supporting Information (SI)).

Subsequently, the source strengths of acid gas in flue gas from CIWF incineration were deduced, as shown in Eqs. (3)–(12). Flue gas is mainly composed of \( N_{2}, CO_{2}, H_{2}O, O_{2}, SO_{2}, \) and HCl (Peng et al., 2016). The excess air ratio was set as 2.1, which is often used for full-scale incinerators.

\[
V(A_{0}) = \frac{E_{\text{w}}^{i} - E_{\text{w}}^{i}}{35.5} \times 22.4 \times \frac{V_{\text{CO}_{2}}}{12} \times 100
\]

(3)

\[
V(CO_{2}) = \frac{E_{\text{w}}^{i} \times 22.4}{32} \times 100
\]

(4)

\[
V(SO_{2}) = \frac{E_{\text{w}}^{i} \times 22.4}{35.5} \times r
\]

(5)

\[
V(HCl) = \frac{E_{\text{w}}^{i} \times 22.4}{5.5} \times 100
\]

(6)

\[
V(H_{2}O) = \frac{E_{\text{w}}^{i} - E_{\text{w}}^{i}}{2} \times 100 + \frac{M_{\text{w}}^{\text{w}}}{18} \times 22.4
\]

(7)

\[
V(O_{2}) = V(A_{0}) \times 1.1 \times 0.21
\]

(8)

\[
V(N_{2}) = V(A_{0}) \times 2.1 \times 0.79 + \frac{E_{\text{w}}^{i} \times 22.4}{28} \times 100
\]

(9)

\[
V(G) = V(CO_{2}) + V(SO_{2}) + V(HCl) + V(H_{2}O) + V(O_{2}) + V(N_{2})
\]

(10)

\[
M(SO_{2}) = \frac{E_{\text{w}}^{i} \times 64}{32} \times 100
\]

(11)

\[
M(HCl) = \frac{E_{\text{w}}^{i} \times 36.5}{35.5} \times 100
\]

(12)

\[
C(j) = \frac{M(j)}{V(G)} \times 10^{6}
\]

(13)

where \( V(A_{0}) \) is the theoretical air demand for incineration of 1 kg of CIWF (Nm\(^{3}\)/kg CIWF); \( V(CO_{2}) \), \( V(SO_{2}) \), \( V(HCl) \), \( V(H_{2}O) \), \( V(O_{2}) \), and \( V(N_{2}) \) are the normalized volume of \( CO_{2}, SO_{2}, HCl, H_{2}O, O_{2}, \) and \( N_{2} \), respectively.

The main aim of this study was to quantitatively evaluate the potential impacts of CW co-incineration in MSWI. First, the various compositions and properties of CW and MSW, as well as the co-incineration ratios of CW treated in MSWI were examined. The fluctuation of LHV and ash contents of the co-incinerated waste feedstock (CIWF) and potential boiler corrosion were determined and compared. The policies and experience of CW management in Wuhan city have been summarized in previous studies (Yang et al., 2021; Yu et al., 2020; Wang et al., 2020). Before the COVID-19 outbreak, CW had been predominantly treated in centralized disinfection facilities and CW incinerators which can handle steady-state conditions (Yang et al., 2021; Purnomo et al., 2021). However, during COVID-19 outbreak, co-incineration had to be performed to manage the dramatic increase in CW. Therefore, the contents of major elements and heavy metals in the incineration residues were determined and compared. Furthermore, the effects of CW co-incineration on the fate of major elements and heavy metals as well as potential boiler corrosion were examined by statistical analysis. The findings of this study provide further insights into the risk assessment of CW treated in MSWI, thereby serving as a guide to safe application.

2. Materials and methods

2.1. Prediction calculation using material flow model

2.1.1. Composition data of MSW and CW

The physical and elemental compositions of MSW and CW were obtained from the literature and filtered in accordance with the following principles: (i) physical compositions and proximate analysis data were based on wet basis and (ii) ultimate analysis and higher heating values data were based on dry basis.

In total, 62 and 31 sets of physical compositions of MSW (Table S1) and CW (Table S2) were obtained, respectively, with 62 \( \times \) 31 = 1922 permutations of co-incineration mode between MSW and CW with each co-incineration ratio. Tables S3 and S4 show the proximate and ultimate analysis data and higher heating value data of typical fractions for MSW and CW, respectively, of which, the mean values were used for prediction calculations.

2.1.2. Calculation methods using material flow model

For assessing the effects of CW co-incineration in MSW incinerator, the LHV and ash contents of CIWF and the concentrations of HCl and SO\(_2\) in flue gas from co-incineration were predicted as shown in Eqs. (1)–(13).

First, the ultimate analysis results were transformed from dry basis to wet basis, and the elemental compositions of CIWF with different CW co-incineration ratios were calculated as follows:

\[ E_{\text{w}}^{i} = E_{\text{dry}}^{i} \times \left(1 - \frac{M_{\text{w}}^{\text{w}}}{100}\right) \]  

(1)
respectively, generated by incineration of 1 kg of CIWF (Nm\(^3\)/kg CIWF); \(V(G)\) is the normalized volume of gas generated by incineration of 1 kg of CIWF (Nm\(^3\)/kg CIWF); \(E^c_{\text{wet}}, E^H_{\text{wet}}, E^O_{\text{wet}}, E^S_{\text{wet}}, E^Cl_{\text{wet}}, \text{and } E^N_{\text{wet}}\) are the contents of C, H, O, S, Cl, and N in CIWF on a wet basis, respectively (%); \(r\) is the transfer ratio of S from CIWF to gaseous sulfur compounds, considered as SO\(_2\), along with flue gas, and is set to 0.35 according to previous studies (Belevi and Moench, 2000; Huang et al., 2018; Zhang et al., 2019); \(M(\text{SO}_2)\) and \(M(\text{HCl})\) are the amounts of SO\(_2\) and HCl generated from incineration of 1 kg of CIWF, respectively (kg/kg CIWF); and \(C_{(i)}\) is the normalized concentration of \(i\) (HCl or SO\(_2\)) in the flue gas (mg/Nm\(^3\)). The entire Cl in the feedstock was considered to be discharged into flue gas as HCl.

2.1.3. Uncertainty analysis
The d-factor was adopted to quantify the uncertainty of the calculation results. In general, a higher d-factor for a parameter indicates that it has greater influence on the results (Ma et al., 2020; Talebizadeh and Moridnejad, 2011). All the physical composition values of plastic, paper, textile, and tissue in CW as parameters were assumed to fluctuate by ± 10%, and the changes in the predicted results were observed with CW...
CW were BA samples from the same MSWI without CW co-incineration, named as CWG, were collected. For comparison, 10 APC residues and 10 flow are summarized in Tables S5 and S6. To investigate the effect of CW co-incineration of CW included those from designated hospitals, shelter hospital, and isolation locations. The general information and material incineration amounts of CW per day during sampling are tabulated in NCWG, were also collected (June – July 2020). The specific co-incineration amounts of CW per day during sampling are tabulated in Table S6. It can be noted from the table that the co-incineration ratios of CW were < 10 wt% of the incinerated MSW. All the incineration residues samples were dried at 105 C for 24 h.

The relative contents of major elements in the incineration residues were determined by X-ray fluorescence spectrometer (XRF-1800, Shimadzu, Japan). The contents of heavy metals in the APC residues were measured by inductively coupled plasma optical emission spectroscopy (ICP-OES) (5100-OES, Agilent, USA) after acid digestion according to EPA Method 3050B (United States Environmental Protection Agency, 1996). All the chemicals and reagents utilized in this study were purchased from Sinopharm Chemical Reagent Co., Ltd. (China), including concentrated HCl, HNO3, HF, HClO4, etc., and all the solutions employed for the experiments were prepared using Milli-Q water.

2.3. Multivariate data analysis

Descriptive statistics and correlation analysis were performed to evaluate the calculated and analytical data using Python (version 3.8.3) and R studio (version 4.0.2) software.

3. Results and discussion

3.1. Effect of CW co-incineration on CIWF properties and acid gas emissions

3.1.1. Predicted results

The estimated LHV and ash contents of CIWF under various CW co-incineration ratios (0%, 5%, 10%, 15%, and 20%) are summarized in Figs. 1A, B, and S1. LHV(Dulong)_wet and LHV(literature)_wet were 8691 ± 1563 kJ/kg (Fig. 1A) and 8233 ± 1558 kJ/kg (Fig. 1B), respectively, when the CW co-incineration ratio was 0%. Similar LHV results were obtained using two different methods, and the LHV of CIWF showed an upward trend with the increasing CW co-incineration ratio. Moreover, the ash contents of CIWF significantly increased from 11% ± 7–15% ± 6% as the CW co-incineration ratio was increased from 0% to 20% (Fig. S1), which was mainly caused by the inorganic components (glass and metals) in CW (Table S4).

The concentrations of HCl and SO2 in flue gas under different CW co-incineration ratios are shown in Fig. 1C and D. The HCl concentrations were 825 ± 78, 1536 ± 438, 2190 ± 604, 2795 ± 1119, and 3359 ± 1392 mg/Nm³ when the CW co-incineration ratios were 0%, 5%, 10%, 15%, and 20%, respectively (Fig. 1C), whereas the corresponding SO2 concentrations were 395 ± 28, 376 ± 23, 359 ± 21, 344 ± 21, and 329 ± 21 mg/Nm³, respectively (Fig. 1D). Thus, a significant increase (P < 0.001) in HCl concentration and decline (P < 0.001) in SO2 concentration were observed with the increase in CW amounts.

These results are in agreement with the reported actual concentrations of HCl and SO2 in MSWI flue gas from boiler (about 500–2000 and 200–1000 mg/Nm³, respectively) (Neuwahl et al., 2010; Zhang et al., 2019; Bai, 2009). It should be noted that the concentration of HCl reached or exceeded the maximum of this range in MSWI when the CW co-incineration ratio was > 10%. These findings confirmed that the ratio of CW treated in MSWI should be limited.

3.1.2. Uncertainty analysis

The results of uncertainty analysis of the impact factors, contents of plastic, paper, and textile in CW are presented in Table 1. The variation in plastic proportion showed significant effects on the estimated LHV(Dulong)_wet and LHV(literature)_wet of CIWF, as well as the source strengths of HCl in flue gas, with d-factors of 0.13, 0.13, and 0.14, respectively. Paper and textile exhibited similar influences on the estimated results, whereas tissue proportion exerted minimum influence.

The use of plastic products has sharply increased for achieving protection from COVID-19 infection, which has resulted in subsequent massive generation of plastic waste (Klemes et al., 2020; Zhou et al., 2021; Jung et al., 2021). Plastic waste with high organic carbon content, which is a statistically significant predictor of LHV (Komilis et al., 2012), can increase the average LHV of feedstock in co-incineration plants (Fig. 1A, Table 1), thus influencing the temperature of the furnace. Furthermore, the Cl content in PVC, one of the main compounds in plastic waste (e.g., syringes, vinyl gloves) (Klemes et al., 2020; Sharma et al., 2020) can enhance HCl emission (Fig. 1C) and formation of chlorinated aromatic compounds (Vejarano et al., 2013). Therefore, the exhaust gas emissions from stack must be monitored when CW is co-incinerated in MSWI, and if an increasing trend is detected, then the operation parameters of the APC system should be adjusted to ensure compliance with the emission limits, especially the semi-dry scrubber for absorbing acid gas and activated carbon injection for removal of heavy metals and chlorinated aromatic compounds.

3.1.3. Limitation of prediction calculation

Prediction calculation using material flow model helps to investigate the influence of CW co-incineration on feedstock LHV and acid gas

Table 1

| Component | LHV(Dulong)_wet | LHV(literature)_wet | Ash | HCl | SO2 |
|-----------|----------------|---------------------|-----|-----|-----|
| plastic   | 0.13           | 0.13                | 0.00| 0.14| -0.08|
| paper     | 0.02           | 0.03                | 0.00| -0.01| -0.02|
| textile   | 0.03           | 0.03                | 0.01| -0.01| -0.03|
| tissue    | 0.00           | 0.00                | 0.00| 0.00| 0.00|

LHV(Dulong)_wet: estimated lower heating values (LHV) based on Dulong formula; LHV(literature)_wet: estimated LHV based on the LHV results of each component of MSW and CW from the literatures.
emission. However, it cannot predict the effect of co-incineration on the emission of NOx, heavy metals, dioxins, etc., because their conversions are more complicated and significantly influenced by incineration conditions besides the feedstock characteristics. Owing to the possible differences in the compositions and characteristics of CW and MSW before and during the COVID-19 pandemic, the characterization of CW and MSW during the pandemic could help to increase the accuracy of the modeling result. However, sampling and analysis of COVID-19-related materials provided by hospitals and public health centers are needed for further study.
3.2. Fate of elements in the incineration residues

3.2.1. Variances in major elements

Incineration of solid waste not only emits gases (SO\textsubscript{2}, HCl, etc.), but also generates solid residues such as APC residues and BA. To investigate the changes in major elements in the incineration residues in response to the introduction of CW in MSWI, elemental analysis of the incineration residues was conducted by using XRF. The top 6 elements were used as factors and subjected to principal component analysis (Fig. 2).

Two principal components explained 94.9% and 93.9% of the variability in the datasets of APC residues (Fig. 2A) and BA (Fig. 2B), respectively. The most common elements in APC residues were C, Ca, Cl, Na, K, and S, whereas those in BA were Ca, C, Si, Cl, Al, and Fe, which are in agreement with those reported in previous studies (Quina et al., 2008; Lindberg et al., 2015).

A good distinction in the element contents in the APC residues was observed between NCWG and CWG, which was governed by specific major contributors of the Dim2 scores, including the contents of C, Ca, Cl, Na, and K (Student’s t-test, P < 0.01). On the contrary, no statistically significant difference in the element contents in BA was detected.

The differences in the C and Ca contents in the APC residues can be ascribed to the variance in APC operational condition, instead of feedstock (Lindberg et al., 2015), whereas the differences in other elements in the APC residues can be attributed to the input of CW components.

Fig. 3 summarizes the contents of major elements in the APC residues (Fig. 3A) and BA (Fig. 3B) with varying CW co-incineration amounts. In the APC residues, with the increasing CW co-incineration amount from 0.0 to 87.0 t/day (<10 wt% of MSW incinerated, around 1000 t/day), the contents of Na, K, Si, P, and Ti increased; those of Ca and C decreased; and those of Cl, S, Mg, Al, and Fe showed negligible variations. Besides, the Na content (9.0% ± 1.6%) in CWG exceeded that reported in previous studies (0.6–8.4%) (Quina et al., 2008; Lindberg et al., 2015). When compared with the APC residues, the fluctuation in the major element contents in BA with varying CW co-incineration amounts was insignificant (Fig. 3B).

3.2.2. Distribution of elements

It is generally accepted that elements in waste are distributed in flue gas during MSWI by two potential processes: (i) elements are entrained by particles and (ii) elements evaporate in the furnace and exist in gas phase or condense and are adsorbed on the surface of particles (Belevi and Moench, 2000; Belevi and Langmeier, 2000; Zhang et al., 2008; Jung et al., 2004).

The results of correlation matrix and hierarchical clustering determined in the present study for the variation tendency of element richness in the APC residues (Fig. 4) are consistent with those previously reported (Jung et al., 2004). It has been indicated that Na, K, P, Ti, Si, and Mg in input waste are non-volatile and more likely to be entrained.
into flue gas with particles and then captured into the APC residues. The elements Br, Pb, Zn, and Cu are mainly transformed by evaporation, while Mn and Cr represent the least volatile elements (Ruth, 1998).

Based on the above-mentioned data, although addition of CW in MSWI had an influence on the major constituents of the APC residues (Fig. 3A), the negligible alteration with regard to the distribution behavior of elements when the CW amounts treated in MSWI were <10 wt% was ascertained.
The contents of heavy metals in the APC residues analyzed by ICP-OES are shown in Fig. 5. The Ba, Cr, Mn, and Ni contents in CWG were significantly higher when CW was treated in MSWI (231–697, 17–152, 118–285, and 7–35 mg/kg, respectively), when compared with those in NCWG (243–311, 15–41, 123–149, and 5–12 mg/kg, respectively) (Fig. 5 and S2). No significant differences in the contents of As, Cd, Cu, and Zn were observed between NCWG and CWG (1–65, 71–293, 216–694, and 2216–6909 mg/kg, respectively) (Figs. 5 and S2). The Pb content in CWG was significantly lower (358–1500 mg/kg), when compared with that in NCWG (1075–1356 mg/kg) (Figs. 5 and S2), and exhibited 5-fold deviation (300 mg/kg) than that in NCWG (59 mg/kg), which could be attributed to the difference in the properties presented in Fig. 4. Ba, Cr, Mn, and Ni were transferred to flue gas during co-incineration (Liu et al., 2020). NaCl and KCl are prone to form alkali metal chlorides and subsequently transfer into flue gas (Ma et al., 2020). The ash contents and LHV of CIWF as well as the concentration of HCl in flue gas were significantly increased when the CW co-incineration ratio was increased from 0 wt% to 20 wt%.

Analyses of the element contents in the APC residues and BA from MSWI with and without CW co-incineration (CWG and NCWG, respectively) revealed the following: (i) the major element contents in the APC residues significantly changed, whereas those in BA showed negligible variation in CWG, when compared with those in NCWG; (ii) the contents of non-volatile heavy metals, such as Ba, Cr, Mn, and Ni, significantly increased in CWG, whereas they were still within the reported ranges in NCWG; and (iii) the increased contents of alkali metals and HCl in flue gas might exacerbate corrosion of boiler surface. These findings provide a better understanding of the impacts of CW on MSWI and can guide future applications of co-incineration during and post pandemic.

3.3. Adverse impact of CW co-incineration on MSWI

3.3.1. Volatilization of heavy metals

The contents of heavy metals in the APC residues analyzed by ICP-OES are shown in Fig. 5. The Ba, Cr, Mn, and Ni contents in CWG were significantly higher when CW was treated in MSWI (231–697, 17–152, 118–285, and 7–35 mg/kg, respectively), when compared with those in NCWG (243–311, 15–41, 123–149, and 5–12 mg/kg, respectively) (Fig. 5 and S2). No significant differences in the contents of As, Cd, Cu, and Zn were observed between NCWG and CWG (1–65, 71–293, 216–694, and 2216–6909 mg/kg, respectively) (Figs. 5 and S2). The Pb content in CWG was significantly lower (358–1500 mg/kg), when compared with that in NCWG (1075–1356 mg/kg) (Figs. 5 and S2), and exhibited 5-fold deviation (300 mg/kg) than that in NCWG (59 mg/kg), which could be attributed to the difference in the properties presented in Fig. 4. Ba, Cr, Mn, and Ni were transferred to flue gas during co-incineration (Liu et al., 2020). NaCl and KCl are prone to form alkali metal chlorides and subsequently transfer into flue gas (Ma et al., 2020). The ash contents and LHV of CIWF as well as the concentration of HCl in flue gas were significantly increased when the CW co-incineration ratio was increased from 0 wt% to 20 wt%.

Analyses of the element contents in the APC residues and BA from MSWI with and without CW co-incineration (CWG and NCWG, respectively) revealed the following: (i) the major element contents in the APC residues significantly changed, whereas those in BA showed negligible variation in CWG, when compared with those in NCWG, and the transfer behavior of the elements in the incinerator remained similar and consisted with that reported in the literature; (ii) the contents of non-volatile heavy metals, such as Ba, Cr, Mn, and Ni, significantly increased in CWG, whereas they were still within the reported ranges in NCWG; and (iii) the increased contents of alkali metals and HCl in flue gas might exacerbate corrosion of boiler surface. These findings provide a better understanding of the impacts of CW on MSWI and can guide future applications of co-incineration during and post pandemic.

CRediT authorship contribution statement

Dong-Ying Lan: Conceptualization, Investigation, Resources, Software, Data mining, Writing – original draft; Hua Zhang: Investigation, Resources, Methodology, Visualization, Writing – original draft, Funding acquisition, Project administration; Ting-Wei Wu: Methodology, Software, Data mining; Fan Li: Supervision, Data curation, Formal analysis; Li-Ming Shao: Supervision, Data curation; Pin-Jing He: Writing – review & editing, Funding acquisition, Project administration, Validation.

Declaration of Competing Interest

The authors declare that they have no known competing financial
interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.jhazmat.2021.127144.

References

Bai, L.C., 2009. Incineration Treatment Engineering Technology of Municipal Solid Waste. China Building Industry Press., Beijing.
Belevi, H., Langmeier, M., 2000. Factors determining the element behavior in municipal solid waste incinerators. 1. Lab. Exp., Environ. Sci. Technol. 34, 2507–2512.
Belevi, H., Moench, H., 2000. Factors determining the element behavior in municipal solid waste incinerators. 1. Field Stud. Environ. Sci. Technol. 34, 2501–2506.
He, X.H., Lou, C., Qiao, Y., Lim, M., 2020. In-situ measurement of temperature and alkali metal concentration in municipal solid waste incinerators using flame emission spectroscopy. Waste Manag. 102, 486–491.
Huang, Y.Y., Li, H.X., Jiang, Z.W., Yang, X.J., Chen, Q., 2018. Migration and transformation of sulfur in the municipal sewage sludge during disposal in cement kiln. Waste Manag. 77, 537–544.
Israelsson, N., Hellström, K., Svensson, J.-E., Johansson, L.-G., 2015. KC3-induced corrosion of the FeCrAl alloy Kanthal® at 600 °C and the effect of H2O. Oxid. Met. 83, 1–27.
Jedruchniok, K., Kaczyński, K., Oleszczuk, P., 2021. COVID-19 discarded disposable gloves as a source and a vector of pollutants in the environment. J. Hazard. Mater. 417, 125938.
Jung, C.H., Matsuto, T., Tanaka, N., Okada, T., 2004. Metal distribution in incineration residues of municipal solid waste (MSW) in Japan. Waste Manag. 24, 381–391.
Jung, S., Lee, S., Dou, X., Kwon, E.E., 2021. Valorization of disposable COVID-19 face masks. Waste Manag. 126, 323–330.
Klemet, J.J., Fan, Y.V., Tan, R.R., Jiang, P., 2020. Minimising the present and future plastic waste, energy and environmental footprints related to COVID-19. Renew. Sustain. Energy Rev. 127, 109883.
Komilis, D., Evangelou, A., Giannakits, G., Lyperis, C., 2012. Revisiting the elemental composition and the calorific value of the organic fraction of municipal solid wastes. Waste Manag. 32, 372–381.
Koumestirom, I., Godelis, A., Tsabaris, C., Stathopoulos, V., Papandreou, A., Gamalatos, P., Economou, G., Papadosopoulos, D., 2011. Characterisation and management of ash produced in the hospital waste incinerator of Athens, Greece. J. Hazard. Mater. 187, 412–432.
Lin, X.B., Wang, F., Chi, Y., Huang, Q.X., Yan, J.H., 2015. A simple method for predicting the lower heating value of municipal solid waste in China based on wet physical composition. Waste Manag. 36, 24–32.
Lindberg, D., Molin, C., Hupa, M., 2015. Thermal treatment of solid residues from WtE plants: a review. Waste Manag. 57, 82–94.
Liu, H.M., Wang, Y.C., Zhao, S.L., Hu, H.Y., Cao, C.Y., Li, A.J., Yu, Y., Yao, H., 2020. Review on the current status of the co-combustion technology of organic solid waste (OSW) and coal in China. Energy Fuels 34, 15448–15487.
Ma, S.J., Zhou, C.B., Liu, Y.J., Zhao, Z.L., 2021. The impact of the COVID-19 pandemic on waste-to-energy and waste-to-material industry in China. Renew. Sustain. Energy Rev. 162, 105052.
Talebizadeh, M., Moridnejad, A., 2011. Uncertainty analysis for the forecast of lake level fluctuations using ensembles of ANN and ANFIS models. Expert Syst. Appl. 38, 4126–4135.
United States Environmental Protection Agency, 1996. Acid Digestion of Sludges, Solids and Soils, USEPA 3050B. Office of Solid and Hazardous Wastes, USEPA, Cincinnati, Ohio.
van Doremalen, N., Bushmaker, T., Morris, D.H., Holbrook, M.G., Gamble, A., Williamson, B.N., Tamin, A., Harcourt, J.L., Thornburg, N.J., Gerber, S.L., Lloyd-Smith, J.O., de Wit, E., Munster, V.J., 2020. Aerosol and surface stability of SARS-CoV-2 as compared with SARS-CoV-1. N. Engl. J. Med. 382, 1564–1567.
Vejarano, E.P., Holder, A.L., Marr, L.C., 2013. Emissions of polycyclic aromatic hydrocarbons, polychlorinated dibenzo-p-dioxins, and dibenzofurans from the incineration of nanomaterials. Environ. Sci. Technol. 47, 4866–4874.
Wang, J., Shen, J., Ye, D., Yan, X., Zhang, Y.J., Yang, W.J., Li, X.W., Wang, J.Q., Zhang, L.B., Pan, L.J., 2020. Disinfection technology of hospital waste and wastewater: suggestions for disinfection strategy during coronavirus Disease 2019 (COVID-19) pandemic in China. Environ. Pollut. 262, 114665.
Wang, J., Chen, Z.Q., Lang, X.J., Wang, S.L., Yang, L., Wu, X.L., Zhou, X.Q., Chen, Z.L., 2021. Quantitative evaluation of infectious health care waste from numbers of confirmed, suspected and out-patients during COVID-19 pandemic: a case study of Wuhan. Waste Manag. 126, 323–330.
World Health Organization, 2020. WHO Coronavirus Disease (COVID-19) Dashboard. The World Health Organization, (URL: https://covid19.who.int/).
Yu, S., Sonne, C., Ok, Y.S., 2020. COVID-19’s unsustainable waste management. Science 368, 1438.
Yu, H., Sun, X., Solvang, W.D., Zhao, X., 2020. Reverse logistics network design for effective management of medical waste in epidemic outbreaks: insights from the coronavirus disease 2019 (COVID-19) outbreak in Wuhan (China). Int. J. Environ. Res. Public Health 17, 1776.
Zhang, H., He, P.J., Shao, L.M., He, P.J., 2019. Estimating source strengths of HCl and SO2 emissions in the flue gas from waste incineration. J. Environ. Sci. 35, 370–377.
Zhou, J., Li, B., Wei, X.L., Zhang, Y.F., Li, T., 2020. Slagging characteristics caused by alkali and alkaline earth metals during municipal solid waste and sewage sludge co-incineration. Energy 202, 117773.
Zhou, C.B., Yang, G., Ma, S.J., Liu, Y.J., Zhao, Z.L., 2021. The impact of the COVID-19 pandemic on waste-to-energy and waste-to-material industry in China. Renew. Sustain. Energy Rev. 139, 110693.
Zhou, P., Shi, Z.L., 2021. SARS-CoV-2 spillover events. Science 371, 120–122.
Zhu, F., Takaoka, M., Shiota, K., Oshita, K., Kitajima, Y., 2008. Chloride chemical form in municipal solid waste incineration: a review. Chem. Eng. J. 292, 398–414.