

Mercury behaviour in flue gas from sewage sludge incinerators and melting furnace

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ABSTRACT

Thermal conversion of sewage sludge can be a clean source of renewable energy if the emission of air pollutants from the source is controlled. In 2013, the Minamata Convention on Mercury was adopted, placing greater emphasis on the control of mercury emissions, including mercury emissions from sewage sludge incinerators. To characterise the behaviour of mercury in flue gas, particulate and gaseous mercury concentrations in two incinerators and a melting furnace were measured by manual sampling. In a third facility, continuous emission monitoring was used to characterise temporal trends in gaseous mercury concentrations. Wet scrubbers were determined to be effective air pollution control devices suitable for mercury removal. Stack mercury concentrations were found to be $<10 \mu\text{g}/\text{Nm}^3$, which meets the mercury emission standard for existing plants ($50 \mu\text{g}/\text{Nm}^3$).

Key words | incinerator, mass balance, mercury, sewage sludge, wet scrubber

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INTRODUCTION

Mercury has significant adverse neurological and other health effects, particularly in unborn children and infants. The Minamata Convention on Mercury (MCM), adopted in 2013, promotes global action to address the worldwide transport of mercury in the environment (UNEP 2013). Article 9 of the MCM mandates that the release of mercury or mercury compounds to land or water should be catalogued and minimised. Wastewater treatment plants (WWTPs) are an anthropogenic source of mercury release to the environment. In water, a portion of inorganic mercury is converted to toxic organic mercury, which bio-accumulates in the environment; this is one reason why mercury levels in fish often exceed environmental standards.

Studies on the fate of mercury in WWTPs (Balogh & Nollet 2008; Yoshida *et al.* 2015; Mao *et al.* 2016) and the impact of mercury release from WWTPs on the water environment (Gbondo-Tugbawa *et al.* 2010; Fricke *et al.* 2015) have shown that total mercury concentration in

influent is ranged from 0.08 to $2.8 \mu\text{g}/\text{L}$, with a ratio of methyl mercury to total mercury of less than 0.7% and the majority of the mercury in influent being transferred to sewage sludge in WWTPs. Therefore, sewage sludge management is crucial to mercury pollution control.

Sewage sludge is a type of biomass waste, and there are several options to treat and recycle sewage sludge. Because sewage sludge contains large amounts of the macronutrients nitrogen and phosphorus, land application of treated sewage sludge or biosolids is popular in many countries. However, because of the adverse effects of heavy metals and persistent organic pollutants in the environment, its use has been restricted in some areas (LeBlanc *et al.* 2009). In addition, due to the relatively small land area available in densely populated countries such as Japan, it is difficult to dispose of the volume of sewage sludge produced at WWTPs without some form of intermediate treatment (Imhoff *et al.* 1971; Wang *et al.* 2017). Incineration is one intermediate

treatment option for sewage sludge and is widely used. It has several advantages, including reducing the original volume of sludge, converting to electricity or heat energy and transforming the material into incineration ash, which facilitates handling of the waste. However, sewage sludge incinerators are a source of mercury emissions and waste incinerators are identified as a major emission source of mercury in Annex D of the MCM. Article 8 of the MCM calls for the control and minimisation of mercury release to the atmosphere.

In 2015, approximately 2.26 million tonnes of sewage sludge (dry basis) was produced in Japan (Ministry of Land Infrastructure Transportation & Tourism 2016). More than 70% of sewage sludge is thermally combusted/gasified/melted. In Japan, mercury has been designated as a possible toxic air pollutant under the Air Pollution Control Act, but no emission standards have been set. In response to the MCM, mercury emission standards for sewage sludge incinerators were set at $30 \mu\text{g}/\text{Nm}^3$ for newly constructed plants and $50 \mu\text{g}/\text{Nm}^3$ for existing plants set by the Ministry of the Environment. The standards became effective once the amended Air Pollution Control Act came into effect on April 1, 2018.

To quantify mercury in flue gas, the Japan Industrial Standard (JIS) defines sampling and measurement methods for gaseous mercury but not particulate mercury. The MCM mandates control of total mercury emissions, including both the gaseous and particulate forms. To obtain a complete understanding of mercury emissions in flue gas, it is essential to understand the contribution of particulate mercury. Japanese data on particulate mercury concentrations in sewage sludge incinerator flue gas have not been readily available until now.

Few studies have examined the measures required to control mercury emissions from sewage sludge incinerators (Balogh & Nollet 2008; Pudasainee et al. 2013; Takaoka et al., 2012). To better understand mercury behaviour in flue gas, the present study measured particulate and gaseous mercury concentrations in two incinerators and a melting furnace using manual sampling. Particulate mercury refers to mercury in the flue gas captured with dust on a filter, which cannot be measured continuously. Because the mercury concentration in flue gas is highly dependent on the mercury in waste, gaseous mercury concentrations in another incinerator were measured using a portable continuous emission monitor (CEM) and manual sampling. These data allowed us to evaluate temporal trends in mercury concentration and confirm air pollution control device (APCD) stability.

MATERIALS AND METHODS

Facilities and flow diagrams of air pollution control devices (APCDs)

The experiments were performed at two continuous fluidised-bed-type incinerators (SSI-A and SSI-B) and a melting furnace (MF-A) in a wastewater treatment facility in Japan. The capacities of SSI-A, SSI-B and MF-A are 160 tonnes, 165 tonnes and 180 tonnes wet-basis/day, respectively. Figure 1 presents a flow diagram of the facilities. The flue gas treatment flow in SSI-A and SSI-B is composed of a bag filter (BF) and a wet scrubber (WS). In MF-A, the APCD system consists of a WS and a wet electrostatic precipitator (wet EP). Flue gas in SSI-A and MF-A was manually and sequentially sampled three times at various sampling points (Figure 1) using impingers, according to the Japanese Industrial Standard Method described in the next section (JIS, 1997) to understand mercury behaviour in flue gas and the difference between APCDs.

In SSI-B, mercury concentrations after the WS were monitored continuously using a portable CEM. Eight manual samples of flue gas were also collected to compare with values obtained by CEM. The CEM is described in detail in the next section.

Solid samples of dewatered sludge, dried sludge and dust captured by the BF, and liquid samples of returned water from the WS, were collected to quantify mercury concentrations and understand mass balance. The flow rate of

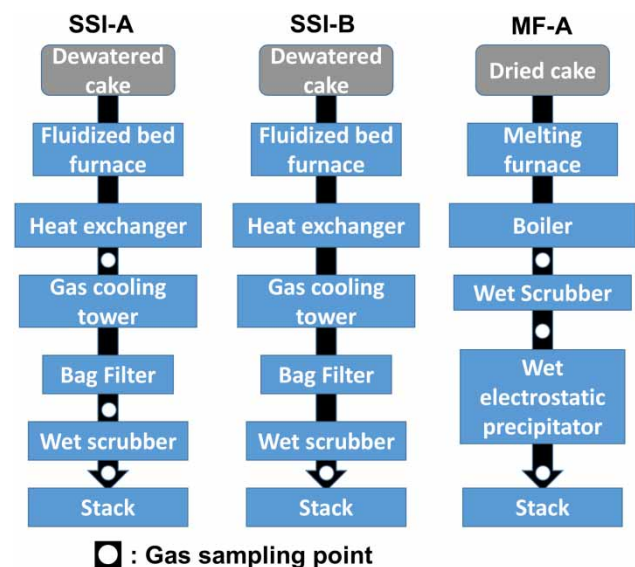


Figure 1 | Flow diagrams of the facilities and sampling points.

wastewater from the WS is required to calculate mass balance; however, no flow meter was available.

Manual gas sampling and mercury measurement

Gaseous mercury was measured according to JIS K0222. The flue gas was passed through a glass filter and bubbled through a sulphuric acid (H_2SO_4) solution with potassium permanganate (KMnO_4). The mercury absorbed in the solution was measured using cold vapour atomic absorption spectrometry (CVAAS) (RA-4300; NIC, Japan). The gas flow rate for sampling was 1 L/min, and the sampling time was 100 min. H_2SO_4 solution with KMnO_4 oxidizes organic or inorganic Hg into Hg^{2+} . For analysis, a stannous chloride was used as a reducing agent, producing Hg vapour. The Hg vapour was directed to an absorption cell, and quantified using atomic absorption at a wavelength of 253.7 nm. Based on the precision of the Hg analyser and the sampling volume of the flue gas, the limit of quantitation was $0.1 \mu\text{g}/\text{Nm}^3$. Each sample was measured three times.

Particulate mercury was sampled according to JIS Z8808. Particulate samples in flue gas were isokinetically collected using a thimble filter. The sampling time was 100 min. The filter was acid digested using nitric acid (HNO_3), H_2SO_4 and KMnO_4 solutions based on the bed material investigation method (Ministry of Environment Japan 2012). The solution was measured using CVAAS. Each sample was measured three times.

Liquid samples

Wastewater from the WS was sampled three times in SSI-A and MF-A. Thermal decomposition using heat-vaporisation mercury measuring equipment (MA-2000; Nippon Instruments Co., Ltd, Japan) was performed to determine mercury concentrations in the samples. Mercury was collected as a gold amalgam and detected via CVAAS (Fukuda *et al.* 2011). The detection limit was 0.002 ng using MA-2000. Each sample was measured three times.

Solid samples

Three samples of fly ash from the BF were obtained from SSI-A and SSI-B. Mercury concentration in the fly ash was measured by heat-vaporisation mercury measuring equipment (MA-2000). Dewatered sludge in SSI-A and SSI-B, and dried sludge and slag in MF-A, were sampled three times. The solid samples were acid digested based on the bed material investigation method and the solution was

measured by CVAAS. Each sample was measured three times.

Mercury continuous emission monitor

Mercury concentration in flue gas is dependent on mercury content in waste, and it is necessary to continuously monitor the concentration in flue gas. CEM (model WLE + EMP-2, Nippon Instruments) was used in SSI-B. In the pre-treatment system (WLE), the flue gas reacts with a tin (II) chloride (SnCl_2) solution in the first bottle; oxidised mercury is then converted to elemental mercury before flowing into a potassium hydroxide (KOH) solution in the second bottle to remove acid gases. Excess moisture in the gas is condensed with an electric cooler. These solutions are automatically replaced periodically (Tanida *et al.* 2013). The elemental mercury is detected in the EMP-2 with an atomic absorption spectrophotometer. The detection limit of this device is $0.1 \mu\text{g}/\text{Nm}^3$. The CEM was set to sample during the day for 4 days for comparison with the values obtained by manual sampling (JIS K0222).

Other gases

Oxygen, sulphur dioxide (SO_2) and nitrogen oxide concentrations were monitored continuously based on JIS K301, JIS K0103 and JIS K0104, respectively. In SSI-A and MF-A, HCl and dust concentrations were measured manually based on JIS K0107 and JIS Z8808, respectively.

RESULTS AND DISCUSSION

Mercury behaviour in flue gas

Table 1 presents the data from SSI-A. The mercury concentration before the cooling tower (CT) was $25\text{--}54 \mu\text{g}/\text{Nm}^3$, of which $17\text{--}52 \mu\text{g}/\text{Nm}^3$ was gaseous mercury and $<2\text{--}8 \mu\text{g}/\text{Nm}^3$ was particulate mercury. The mercury concentration before the WS was $37\text{--}47 \mu\text{g}/\text{Nm}^3$, of which $28\text{--}33 \mu\text{g}/\text{Nm}^3$ was gaseous mercury and $4\text{--}19 \mu\text{g}/\text{Nm}^3$ was particulate mercury. Between the CT and WS, the flue gas passed through the BF. Generally, BFs are effective at removing dust and, in this facility, the dust concentration was reduced to less than the detection limit (Shiota *et al.* 2015). However, the mercury concentration did not decrease. Mercury removal by BF depends on the composition of the fly ash. In a fluidised bed incinerator such as SSI-A, all ash including bottom ash discharged separately in a stoker

Table 1 | Mercury concentrations in SSI-A samples

Location	Unit	Before cooling tower			Before WS			Stack		
		No.1	No.2	No.3	No.1	No.2	No.3	No.1	No.2	No.3
Flue gas volume	Nm ³ /hr	17,600			17,800			24,600		
Flue gas temperature	degree C	356			194			143		
Concentration		No.1	No.2	No.3	No.1	No.2	No.3	No.1	No.2	No.3
O ₂	%	5.8	8.1	8.6	6.8	9.2	9.6	14	14.2	15
Dust	g/Nm ³	25	18	130	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
SO ₂	ppm	920	880	840	780	710	740	4	<1	<1
HCl	ppm	35	37	96	41	32	53	<1	<1	<1
NOx	ppm	4.4	5.9	2.8	3.8	5.5	2.1	2.9	2.7	2
Hg (Gas)	μg/Nm ³	49	52	17	33	30	28	4	5	5
Hg (Particulate)	μg/Nm ³	<2	<2	8	4	9	19	<2	<2	<2

furnace is captured by the BF. Therefore, it is considered that the mercury concentration did not decrease because less adsorbent materials are contained in fly ash. If the fly ash contains materials adsorbent for mercury, such as unburned carbon and activated carbon (Balogh & Nollet 2008), a significant amount of mercury can be removed. Other gas components such as HCl and SO₂ also affect the removal of mercury from flue gas. If the HCl concentration is high, like that found in a municipal solid waste incinerator, mercury is easily removed in the fly ash (Takaoka et al. 1995). In addition, high SO₂ and low HCl concentrations tend to keep the mercury species as elemental mercury (Takaoka 2001).

The concentration of particulate mercury increased after passing through the BF, which was an unexpected result. Because dust was completely removed at the BF stage, the particulate mercury was likely produced after the BF. Although condensation and aggregation are possible

mechanisms, there was no clear cause. Therefore, the BF was not considered to be effective for mercury removal in SSI-A.

The stack mercury concentration was 4–7 μg/Nm³, of which 4–5 μg/Nm³ was gaseous mercury and <2 μg/Nm³ was particulate mercury. The WS between the BF and stack was very effective at removing mercury in the flue gas, even when the flue gas volume increased due to heated atmospheric gas mixing into the flue gas. Acid gases such as SO₂ and HCl were also removed efficiently in the WS.

Table 2 presents the data from MF-A. The mercury concentration before the WS was 32–56 μg/Nm³, of which 30–54 μg/Nm³ was gaseous mercury and <2 μg/Nm³ was particulate mercury. The mercury concentration before the wet EP was 9–14 μg/Nm³, of which 7–12 μg/Nm³ was gaseous mercury and <2 μg/Nm³ was particulate mercury. Between the boiler and the wet EP, the flue gas passed

Table 2 | Mercury concentrations in MF-A samples

Location	Unit	Before WS			Before wet EP			Stack		
		No.1	No.2	No.3	No.1	No.2	No.3	No.1	No.2	No.3
Flue gas volume	Nm ³ /hr	13,400			12,700			8,220		
Flue gas temperature	degree C	236			28			27		
Concentration		No.1	No.2	No.3	No.1	No.2	No.3	No.1	No.2	No.3
O ₂	%	9.3	10.2	8.5	10.3	10.9	10.0	8.7	10.2	9.1
Dust	g/Nm ³	0.48	0.47	0.46	0.40	0.40	0.11	0.024	0.026	0.016
SO ₂	ppm	<1	4	13	19	10	9	<1	1	<1
HCl	ppm	1	<1	<1	12	5	3	<1	<1	<1
NOx	ppm	46	51	100	74	62	82	116	82	86
Hg (Gas)	μg/Nm ³	30	54	53	12	8	7	7	7	6
Hg (Particulate)	μg/Nm ³	<2	<2	<2	<2	<2	<2	<2	<2	<2

through the WS. Compared to the fluidised bed incinerator, dust concentrations before the APCDs were lower. This is most likely because almost all of the ash content was melted and fixed in slag in the melting furnace, allowing only vaporised substances and tiny amounts of ash to comprise the dust in the flue gas. Although removal of dust by the WS was low, mercury removal by the WS was effective.

The stack mercury concentration was 6–9 $\mu\text{g}/\text{Nm}^3$, of which 6–7 $\mu\text{g}/\text{Nm}^3$ was gaseous mercury and <2 $\mu\text{g}/\text{Nm}^3$ was particulate mercury. The wet EP between the WS and the stack was slightly effective at removing mercury in the flue gas.

The average mercury concentration in stack gas in an SSI was previously reported as 21.1–36 $\mu\text{g}/\text{Nm}^3$ (Takaoka *et al.*, 2012). The stack mercury concentrations in this study were much lower.

Relationship between manual sampling and CEM

The temporal change in mercury concentration at the stack in SSI-B is shown in Figure 2. Mercury concentration was

high immediately after the start of CEM due to the instability of the analyzer. The reason for the instability is not completely clear. But, it is speculated that flue gas temperature and adsorption of Hg inside the pretreatment system affected the instability. A subtle amount of mercury was adsorbed on the remaining chemical reagent and glass wares in the pretreatment system. When the flue gas passed at first, the temperature inside the pretreatment system increased, and then the mercury adsorbed in the pretreatment system was released, which caused the high concentration. Once the analyzer stabilised, the mercury concentration ranged from 4–10 $\mu\text{g}/\text{Nm}^3$ and remained consistent over several days. Generally, gaseous mercury concentration in flue gas showed great fluctuations in municipal solid waste incineration because the fluctuations seem to be derived mainly from great heterogeneity of combusted wastes (Takaoka 2018). The instantaneous concentration measured by CEM often exceeds 100 $\mu\text{g}/\text{Nm}^3$. Compared to the change in mercury concentration in the municipal solid waste incinerator, the temporal change in

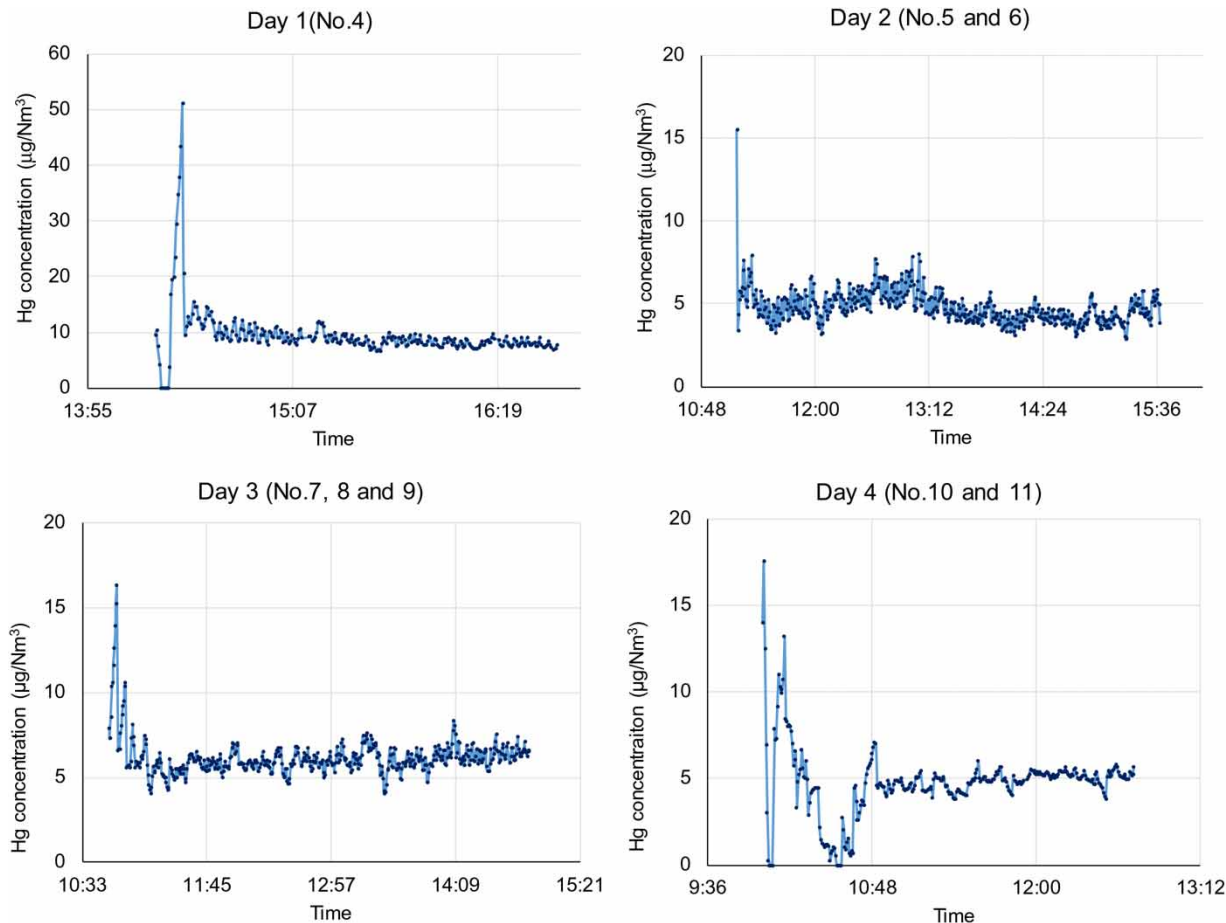


Figure 2 | Temporal changes in SSI-B stack mercury concentrations.

Table 3 | Mercury concentrations in SSI-B samples

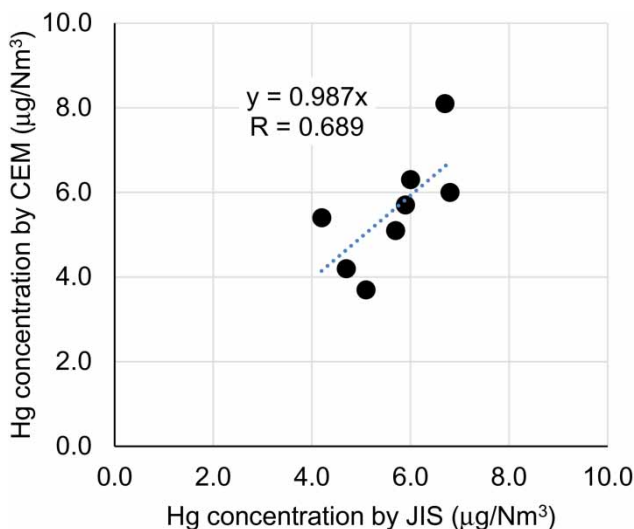
Location	Unit	After WS (stack)								
		No.4	No.5	No.6	No.7	No.8	No.9	No.10	No.11	
Flue gas volume	Nm ³ /hr	24,100	31,500		31,000				31,600	
Flue gas temperature	degree C	173	170		169				176	
Concentration										
O ₂	%	15.9	15.7		15.9				15.6	
Dust	g/Nm ³	<1	<1		<1				<1	
NOx	ppm	<1	<1		<1				1	
Hg (Gas by JIS)	μg/Nm ³	6.7	4.2	4.7	5.9	6.8	6.0	5.1	5.7	
Hg (Gas by CEM)	μg/Nm ³	8.1	5.4	4.2	5.7	6.0	6.3	3.7	5.1	

mercury concentration was small and relatively stable, indicating little variance in the mercury content of sewage sludge. This suggests that the WS was effective and stable in removing mercury.

In SSI-B, manual sampling was conducted in addition to CEM. The mercury concentrations measured by both methods are shown in Table 3. Both methods resulted in almost identical concentration readings in a narrow range from 4–9 μg/Nm³. The relationship between manual sampling and CEM values is shown in Figure 3. The slope is 0.987, with a correlation coefficient of 0.689. The correlation coefficient is relatively low due to the narrow concentration range or interference of coexisting gas.

Mercury concentrations in liquid and solid samples

Mercury concentrations in the liquid and solid samples are shown in Table 4. In the dewatered sludge cake of SSI-A

**Figure 3** | Relationship between manual sampling and CEM values.

and SSI-B, mercury levels were 0.100–0.134 and 0.322–0.499 mg/kg (wet), respectively. The moisture content of dewatered sludge from both incinerators ranged from 78.4 to 79.0%; based on this, the mercury content was calculated as 0.47–2.4 mg/kg-dry sludge (DS). In the MF-A, the mercury concentration was 0.409–0.569 mg/kg. The moisture content of dried sludge was about 30%; based on this, the mercury content was calculated as 0.68–0.81 mg/kg-dry sludge (DS). These results are in line with findings of previous studies. For example, we have previously reported mercury concentrations of 1.24 and 1.29 mg/kg-DS (Takaoka et al. 2012). Ozaki et al. (2006) found that the mercury content of sewage sludge from 22 facilities averaged 1.0 mg/kg-DS, while Yasuda et al. (1983) reported mercury concentrations of 1.13–2.59 mg/kg-DS. Higher mercury concentrations have been reported in other countries (Pudasainee et al. 2013; Mao et al. 2016).

The mercury content of MF-A melting slag was quite low (0.01–0.028 mg/kg). The mercury content of SSI-A and SSI-B fly ash ranged from 0.216 to 0.442 mg/kg and did not differ significantly between facilities. This is because fly ash is relatively uniform compared to dewatered sludge. If the solid material of sewage sludge is to be used as fertiliser in Japan, the mercury content must not exceed the regulation value of 2 mg/kg. Based on our findings, the dewatered sludge would not exceed that value.

Mercury concentrations in WS wastewater ranged from 1.74 to 22.1 μg/L. The range was wide compared to the mercury concentrations in dewatered sludge. In Japan, the standard for mercury in effluent from wastewater treatment facilities is 5 μg/L. Although the mercury concentrations in WS wastewater were higher than the effluent standard, the WS wastewater was mixed with other rejected water and returned to a main wastewater treatment process. Therefore, the WS wastewater cannot be directly released to the environment. However, attention should be paid to the

Table 4 | Mercury concentrations in liquid and solid samples

	Unit	SSI-A			MF-A			SSI-B		
		No.1	No.2	No.3	No.1	No.2	No.3	No.5&6	No.7-9	No.10&11
Dewatered sludge	mg/kg	0.100	0.134	0.121				0.322	0.479	0.499
Dried sludge	mg/kg				0.459	0.409	0.569			
WS wastewater	µg/L	22.1	1.74	1.76	17.0	12.1	13.8			
Fly ash	mg/kg	0.442						0.304	0.216	0.398
Slag	mg/kg				0.028	0.010	0.024			

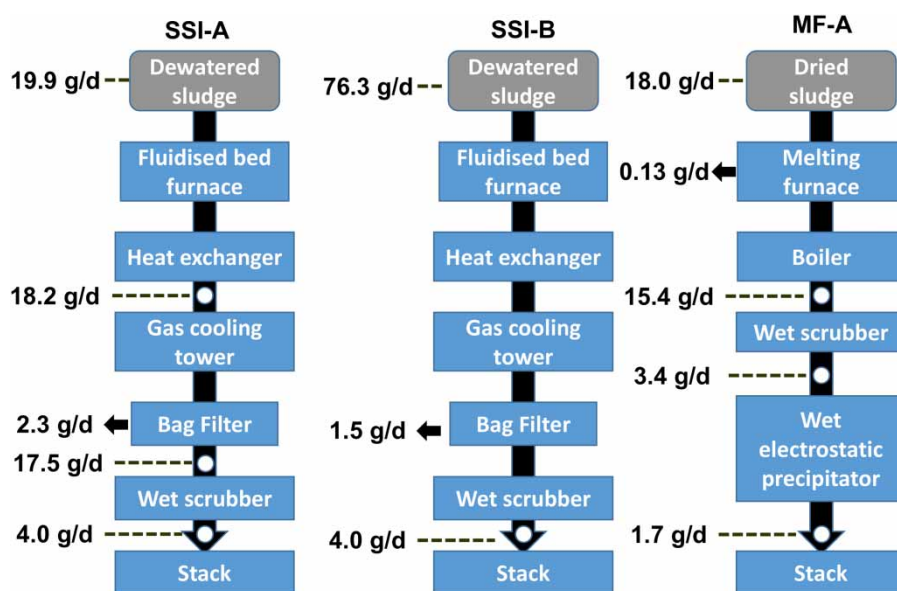
increase of mercury concentration by circulating the mercury in the WWTP.

Mercury mass flow in the sewage sludge incinerators and melting furnace

Mercury mass flows in SSI-A, SSI-B and MF-A are shown in Figure 4 and Table S-1 (available with the online version of this paper). Based on the dewatered sludge volume and mercury content, SSI-A mercury input was calculated as 19.9 g/day. At the sampling location before the CT, the mercury content and volume of flue gas were 43 µg/Nm³ and 17,600 Nm³/hr, respectively (Table 1), indicating a mercury flow of 18.2 g/day. The two mercury input values are in good agreement. Results from the BF indicated removal of 2.3 g of mercury per day. Although the mercury flow before the WS was 17.5 g/day, the flow decreased to

4.0 g/day at the stack. The total volume of WS wastewater was not measured in our study; however, given the high concentration of mercury in WS wastewater, it was assumed that a significant amount of total mercury was transferred to WS wastewater. Based on the mercury flow rates, the removal efficiencies were 4% in the BF and 77% in the WS. Werther & Saenger (2000) reported that typical mercury concentrations in incineration ash range from 0.1 to 0.4 mg/kg, constituting 1–6% of the total mercury input. Similar results were observed in the present study.

For MF-A, mercury input to the furnace was calculated as 18.0 g/day based on dried sludge volume and mercury content (Table S-1). In the slag, which was discharged from the bottom of the melting furnace, mercury flow was very low (0.13 g/day). At the sampling location before the WS, the mercury content and volume of flue gas were 48 µg/Nm³ and 13,400 Nm³/hr, respectively (Table 2),

**Figure 4** | Mercury mass flow in each facility.

indicating a mercury flow of 15.4 g/day. The two mercury input values are in good agreement.

Mercury flow decreased significantly after passing through both the WS and the wet EP. The volume of WS wastewater in MF-A was not measured in our study; however, the high concentration in WS wastewater indicated that more mercury was removed by the WS and Wet EP processes. Based on the mercury flow rates, the removal efficiencies were 78% in the WS and 50% in the Wet EP.

The total mercury removal efficiencies for the APCDs in SSI-A, SSI-B and MF-A were 80%, 95% and 91%, respectively. The percentages were high compared to our previous study (Takaoka et al., 2012). However, mercury removal efficiencies of 98% and 98.6% have been reported in the United States and Korea (Balogh & Nollet 2008; Pudasainee et al. 2013). In the United States study, a carbon injection was used before the BF, and 83.4% of the mercury was present in the BF ash. In the Korean study, a spray dry reactor (SDR) was used between the BF and the WS. Mercury concentrations were 78.4%, 16.2% and 3.5% in the WS wastewater, SDR wastewater and fly ash, respectively. Other studies reported high distributions of mercury to exhaust gas, drainage water and ash (Dewling et al. 1980; Balogh & Liang 1995; Werther & Saenger 2000; Lopes et al. 2003). Therefore, mercury behaviour and removal by APCDs is dependent on other conditions in the facilities.

CONCLUSIONS

This study examined mercury emissions and behaviour in two incinerators and one melting furnace to evaluate the effectiveness of various APCDs.

- (1) In SSI-A and MF-A, the average mercury concentration at the inlet of APCDs was 25–56 $\mu\text{g}/\text{Nm}^3$. At times, the concentration exceeded the 50 $\mu\text{g}/\text{Nm}^3$ emission standard for existing facilities in Japan. In SSI-A, mercury concentrations at the inlet and outlet of the BF were similar, which suggests that mercury was not removed by the BF. The total concentration at the stack was 4–9 $\mu\text{g}/\text{Nm}^3$, indicating that the WS was effective for mercury removal. CEM revealed that the temporal change in mercury concentration was small and relatively stable. The results from manual sampling and CEM were similar. Therefore, the applicability of CEM to flue gas was presumed.
- (2) In the flue gas, gaseous mercury concentrations were higher than particulate mercury concentrations. At the

stack, particulate mercury levels were less than the detection limit. However, particulate mercury was detected at the APCDs. In the future, the mechanism underlying these effects should be elucidated.

- (3) According to mercury flow, 80% and 50% of the total mercury was removed in the WS and wet EP, respectively. Total mercury removal efficiencies were 80% in SSI-A and 91% in MF-A. It is necessary to improve APCDs in sludge incineration facilities, particularly if it is difficult to limit influent mercury levels.
- (4) Mercury concentrations in the dewatered sludge in SSI-A and SSI-B were 0.100–0.134 and 0.322–0.499 mg/kg (wet), respectively. The mercury content in melting slag was low (0.01–0.028 mg/kg). In SSI-A and SSI-B, the mercury concentration in fly ash ranged from 0.216 to 0.442 mg/kg. The concentration did not exceed the regulation value (2 mg/kg) for use as a fertiliser.

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