Numerical Simulations of Leaching and Volatilization Behaviors from Stabilized and Solidified Mercury Metal Waste in Landfill

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Abstract

This study investigates the environmental safety of landfill sites in which mercury metal waste is disposed. Serial batch tests were conducted to evaluate the long-term leaching and volatilization rates of mercury waste stabilized in its sulfide form, and solidified using either a sulfur polymer or two different types of low-alkaline cement. Using measured mercury leaching and volatilization rates, numerical simulations were conducted to investigate the long-term behavior of mercury after its disposal in landfill sites. For leaching behavior, the concentration profiles of dissolved mercury accumulating in a drainage pipe at the bottom of a landfill site were calculated, while volatilization behavior was assessed by analyzing the gaseous mercury emissions from the final cover surface. The effects of covering the disposed mercury waste with either soil sorption layers or cut-off layers were evaluated through analysis of the calculated concentration profiles.

Key words : final disposal, leaching, numerical simulation, volatilization, waste metal mercury

1. Introduction

In limiting the use of mercury by 2020, the Minamata Convention on Mercury has led to widespread decommissioning of mercury-based products, and a general increase in the disposal of hazardous mercury waste. Mercury is a toxic substance which is strictly regulated by environmental quality standards: in the final stages of disposal in landfill sites, waste mercury has to be stabilized in a sulfide form, and/or solidified with polymer or cement to minimize emission of the metal, through either leaching or volatilization. The containment performance in terms of both stabilization and solidification has been of great concern—in Japan, the leaching concentration from waste materials such as incineration bottom ash, incineration fly ash and crushed fluorescent lamps (mercury content < 10%) have been investigated. However, there have been limited findings on the leaching behavior of waste containing greater than 10% mercury, as is the case with mercury metal waste; additionally, the vaporization of mercury has also yet to be sufficiently investigated in this context. To evaluate the long-term containment performance of mercury

stabilization and solidification, serial batch tests were conducted using four stabilized mercury metal waste specimens which were solidified with different binding agents, and measuring the mercury leaching and volatilization rates over a period of 64 days. In addition, this study simulated the movement of emitted mercury through a landfill site, yielding cross-sectional concentration distributions: the concentration profiles of dissolved mercury in leachate accumulating in drainage pipes at the bottom of the disposal site were evaluated, along with gaseous mercury emitted from the surface of the final cover, and the effects of covering the bound waste with soil sorption and cut-off layers were clarified.

Based on the results, disposal strategies are discussed in terms of how well the different binding and disposal methods satisfy environmental quality standards.

2. Materials and Methods

2.1 Materials

According to previous studies, mercury metal waste stabilized as mercuric sulfide and solidified with sulfur polymer has a leaching concentration of < 0.005 mg/L,

fulfilling the Japanese waste acceptance criteria for controlled landfill sites. On the other hand, low-alkaline cement has been considered as an alternative binder with an aim at reducing mercury leaching. To prepare the testing materials, mercury metal waste with a purity of 99.9% was first stabilized as mercuric sulfide, and then solidified with one of four binders, (1) sulfur polymer, (2) low-alkaline cement A, (3) low-alkaline cement B, and (4) low-alkaline cement B with a water reducing agent.

Low-alkaline cement A was a manufactured product. Low-alkaline cement B was prepared by mixing ordinary Portland cement, silica fume and fly ash in a 3: 2: 5 ratio. Sulfur polymer had been considered effective for reducing mercury leaching, but it was relatively expensive. This study aimed to investigate whether low-alkaline cements A or B could exhibit the same performance as the sulfur polymer.

2.2 Serial Batch Leaching Tests

Serial batch leaching tests were conducted to measure the mercury leaching rate from stabilized, solidified mercury metal waste. The specimens used in these tests were diamond-cut into 2 cm cubes, then tied up with fluoroplastic string and suspended from the cap of a 200 mL bottle. Eighty-four mL of distilled water—3.5 times the surface area of the specimen—was injected into the bottle, and the suspended specimens were immersed in the water by replacing the cap. The bottles were placed in a temperature-controlled room at 25°C, thereby commencing the serial batch leaching tests. The leaching liquids were exchanged with fresh distilled water after 1, 2, 4, 8, 16, 32, and 64 days, and the concentrations of dissolved mercury in the exchanged leaching liquids were measured. The leaching rate was calculated by,

$$j_{w,i} = \frac{c_i}{t_i - t_{i-1}} \frac{L}{S} \tag{1}$$

where t_i = elapsed time (day), c_i = i-th mercury concentration (ng/m^3), $j_{w,i}$ = i-th mercury leaching rate (ng/m^2/day), L = liquid volume (m^3), S = sample surface area (m^2), and i = index of the fraction. The obtained leaching rates, $j_{w,i}$, are related to the average elapsed time, $\bar{t}_i$, by,

$$j_{w,i} = K_w \bar{t}_i^{-a_w} \tag{2}$$

where K_w = initial leaching rate (ng/m^2/day) and a_w = the leaching parameter. $\bar{t}_i$ was calculated by,

$$\bar{t}_i = \left(\frac{\sqrt{t_i} + \sqrt{t_{i-1}}}{2} \right)^2 \tag{3}$$

The fitting parameters K_w and a_w can be evaluated by plotting the relationship between logarithmic $j_{w,i}$ and logarithmic $\bar{t}_i$. In the plotted graph, the value of K_w denotes the intercept, and the value of a_w shows the slope. As reported by Sakanakura *et al.* (2009), a_w can take

values between 0.0 and 1.0: a value of 0.0 indicates that dissolution processes dominate leaching; at 0.5, leaching becomes diffusion-controlled (depending on the concentration difference between liquid and solid); and at 1.0, leaching reaches chemical equilibrium.

2.3 Serial Batch Volatilization Tests

Serial batch volatilization tests were conducted to measure the volatilization rate of mercury from the stabilized and solidified mercury metal waste samples. As with the leaching tests, 2 cm sample cubes were used: these were placed in a 500 mL bottle, which was sealed with a cap equipped with two connectors for attaching silicon tubes. After the silicon tubes were closed with pinchcocks, the tests were initiated, maintaining a constant temperature of 25°C. The gases in the bottle were extruded using purified air after periods of 1, 2, 4, 8, 16, 32, and 64 days, and gaseous mercury in the extruded mixture was absorbed onto gold amalgam: the mercury content in the amalgam was then measured to ascertain the quantity of volatilized metal. The volatilization rate of mercury was calculated by,

$$j_{g,i} = \frac{1}{t_i - t_{i-1}} \frac{m_i}{S} \tag{4}$$

where m_i = i-th mercury content in the gold amalgam (ng), and $j_{g,i}$ = i-th mercury volatilization rate (ng/m^2/day). The obtained leaching rates, $j_{g,i}$, were also related to the average elapsed time, $\bar{t}_i$,

$$j_{g,i} = K_g \bar{t}_i^{-a_g} \tag{5}$$

where K_g = initial volatilization rate (ng/m^2/day) and a_g = the volatilization parameter. The fitting parameters of K_g and a_g were evaluated from changes in the mercury volatilization rate with the average elapsed time.

2.4 Numerical Simulations

A seepage analysis and advection-diffusion analysis were conducted to investigate the long-term behavior of mercury in landfill sites containing stabilized, solidified mercury metal waste. When the waste layer in a landfill site is assumed to be an isotropic porous medium, the velocity of water fluid flow is given by Darcy's law:

$$u_w = -\frac{k_{rw}K}{\eta_w}(\nabla p_w + \rho_w g \nabla z) \tag{6}$$

where u_w = volumetric water velocity (m/s), η_w = water viscosity (=1×10^{-3} Pa·s), k_{rw}=relative hydraulic conductivity, K = intrinsic permeability (m^2), p_w = water pressure (Pa), ρ_w = water density (=1×10^3 kg/m^3), and g = gravitational acceleration (=9.81 m/s^2). Thus, the equation governing seepage flow can be written as,

$$\frac{\partial(\rho_w \theta_w)}{\partial t} = \nabla \cdot \left[-\rho_w \frac{k_{rw}K}{\eta_w}(\nabla p_w + \rho_w g \nabla z) \right] \tag{7}$$

where θ_w = volumetric water content. The gas flow
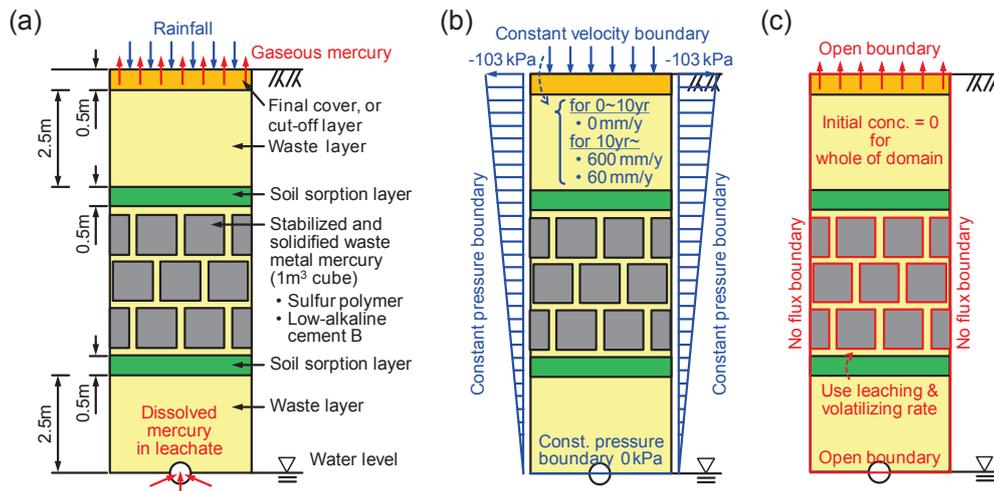
Fig. 1 Analysis domain and conditions: (a) two-dimensional cross section of landfill, (b) initial and boundary conditions for seepage analysis, (c) initial and boundary conditions for advection-diffusion analysis.

velocity was assumed to be negligible, as the gas pressure in the main waste layer where incineration ash is disposed is almost the same as the atmospheric pressure, and therefore the gas pressure gradient is regarded to be zero.

The transports of dissolved and gaseous mercury are formulated respectively as follows:

$$\frac{\partial(c_w\theta_w)}{\partial t} + \nabla[-(D_w+\theta_wD_e)\nabla c_w + u_w c_w]$$
$$= \theta_g K_H (c_g - Hc_w) - \rho_d K_d \frac{\partial c_w}{\partial t} \tag{8}$$

$$\frac{\partial(c_g\theta_g)}{\partial t} + \nabla(-\theta_g D_e \nabla c_g)$$
$$= \theta_g K_H (Hc_w - c_g) - \rho_d K_d \frac{\partial c_g}{\partial t} \tag{9}$$

where c_w = dissolved mercury concentration (ng/m^3), c_g = gaseous mercury concentration (ng/m^3), D_w = the water phase dispersion coefficient (m^2/s) (Bear, 1972), D_e = the effective diffusion coefficient (m^2/s), K_H = the mass transfer rate (1/s), H = the dimensionless Henry constant, ρ_d = the dry bulk density (kg/m^3), and K_d = the distribution coefficient (m^3/kg). In equations (8) and (9), the first term on the right-hand side indicates the amount of material transferred between the water and gas phases due to dissolution and volatilization of mercury, and the second term on the right-hand side represents soil adsorption of both mercury and gaseous mercury. The set of equations (7) to (9) was numerically solved using COMSOL ver 5.1.

The cross-sectional analysis domain of the modelled landfill site is shown in Fig.1. The stabilized, solidified mercury waste metal was assumed to take the form of 1 m^3 cubes. The entire array of mercury waste cubes was covered with a soil sorption layer, designed to slow the transport of emitted mercury. A drainage pipe was placed at the bottom of the analysis domain to accumulate leachate, and a final cover or cut-off layer to reduce

Table 1 Analytical conditions.

Parameters	Unit	Waste layer	Soil sorption layer
Porosity	1	0.3	0.3
Intrinsic permeability	m^2	1×10^{-12}	1×10^{-12}
Dry bulk density	kg/m^3	1,400	1,800
VG parameter, a	1/m	2	2
VG parameter, n	1	1	1
Long. dispersivity	m	3	3
Trans. dispersivity	m	1	1
Distribution coefficient	mL/g	0	100
Henry constant	1	0.43	0.43

Note: Henry constant value from Andersson *et al.* (2008)

rainfall permeation was placed on top. Figures 1(b) and 1(c) show the initial and boundary conditions of the analysis: the boundary condition at the top of the domain indicates the rainfall intensity. For the first 10 years of the analysis, the rainfall intensity for the top boundary condition was considered to be 0 mm/y, as landfilling of the mercury waste would be carried out under a roof; at the end of the landfilling process, the roof would be removed, so in subsequent years, rainfall at the top boundary condition is considered to permeate the final cover with an intensity of either 600 mm/y or 60 mm/y. The different rainfall intensity values were used to evaluate the effects of using a cut-off layer covering the waste site, which would decrease the overall ingress of water into the landfill site. The measured leaching and volatilization rates of mercury were applied to the surface boundaries of the stabilized, solidified mercury metal waste. In this numerical simulation, effects of the soil sorption and cut-off layers on the concentrations of the dissolved mercury in leachate in the bottom drainage pile and gaseous mercury emitted from the final cover were investigated: the analytical conditions for these models are listed in Table 1.

3. Results and Discussion

3.1 Mercury Leaching and Volatilizing Rates

Figure 2 shows the results of the serial batch leaching tests. As shown in Fig. 2(a), the sulfur polymer was found to be the most effective binder for reducing mercury leaching: in comparison, the amount of mercury leached when solidified with low-alkaline cement was more than 10 times higher. The inclusion of a water reducing agent had little effect on decreasing the amount of leached mercury. Figure 2(b) shows the measured leaching rates and results obtained by fitting with equation (2). There were significant differences between the initial leaching rates among the samples bound with sulfur polymer and low-alkaline cement A and B, and the leaching rates were observed to decrease over time in all cases. The mercury metal waste solidified with the sulfur polymer had a leaching parameter of 0.53, demonstrating that leaching was dominated by the inner diffusion effect. In contrast, the mercury solidified with either low-alkaline cement (A or B) exhibited leaching parameters of 0.75–0.85.

Figures 2(c) and 2(d) show the results of the serial batch volatilization tests. As was the case for leaching, binding with the sulfur polymer yielded the lowest levels of mercury volatilization, and low-alkaline cement A was just as effective in preventing gaseous mercury emission. The cumulative amount of volatilized mercury from mercury waste solidified with low-alkaline cement B was about 10 times higher than that with either the sulfur polymer or low-alkaline cement A. As shown in Fig. 2(d), the volatilizing rates also decreased over time, as was the case for leaching. Their volatilizing parameters were 0.75–0.85.

In Table 2, the results of the serial batch leaching and volatilization tests are summarized. Mercury leachability and volatility were defined by dividing the amounts of leached and volatilized mercury by the initial mercury waste content. The mercury metal waste solidified with the sulfur polymer exhibited extremely low volatility (1.19×10^{-8}%) as well as extremely low leachability (7.45×10^{-8}%). In contrast, the low-alkaline cements were not as effective in reducing mercury leachability, 6.68×10^{-6}% and 3.64×10^{-5}% for low-alkaline cements A and B, respectively. However, mercury solidified with low-alkaline cement A restrained mercury volatilization as well as the sulfur polymer, whereas the rate of volatilization from cement B-bound mercury was roughly 10 times higher.

The parameters shown in Table 2 were evaluated at 25°C. The temperature in landfills may be elevated up to 60°C by biological degradation of organic substances in the landfills. The elevated temperature will enhance the leaching and volatilization of mercury. In particular, the temperature effect on the volatilization rate is considered significant, so further studies will be essential.

3.2 Mercury Behavior in Landfill Sites

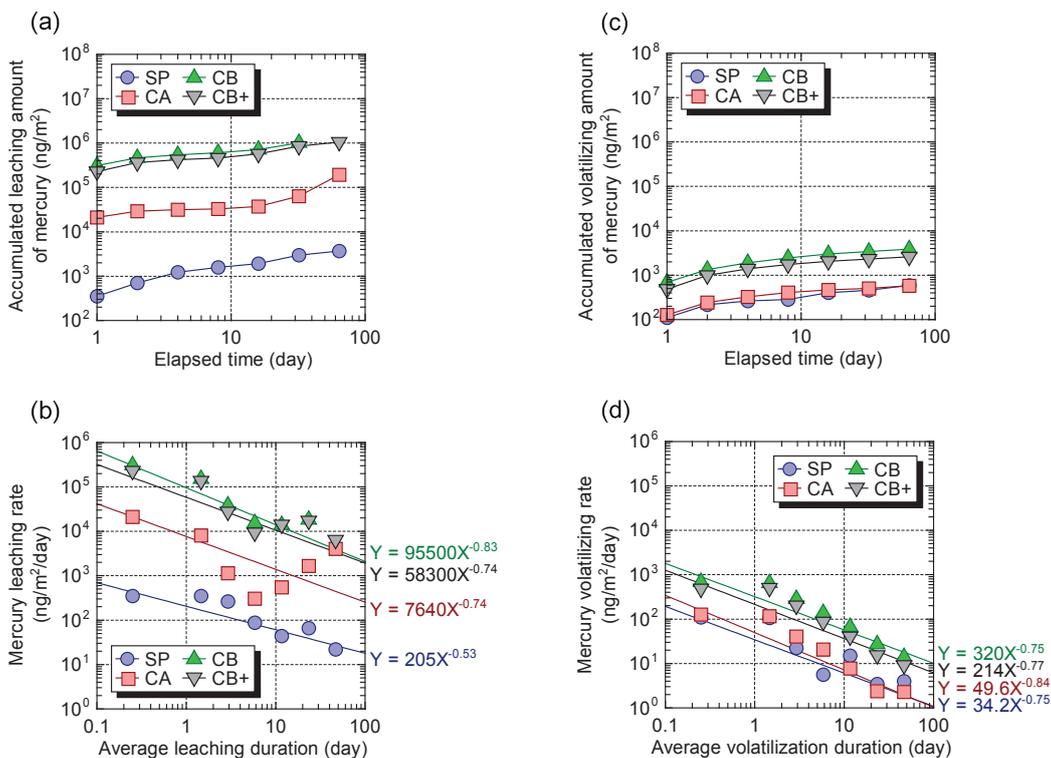Figures 3(a) to 3(d) show contour maps of the concentration distributions of dissolved and gaseous

Fig. 2 Results of serial batch leaching tests and serial batch volatilization tests: (a) cumulative amount of leached mercury, (b) leaching rate, (c) cumulative amount of volatilized mercury, (d) volatilizing rate.

Table 2 Summary of serial batch leaching and volatilization tests.

| Name	SP	CA	CB	CB+
Binder for solidification	Sulfur polymer	Low-alkaline cement A	Low-alkaline cement B	Low-alkaline cement B & water reducing agent
Bulk density (kg/m^3)	3,440	2,550	2,440	2,410
Hg content (mg/kg)	430,000	340,000	340,000	350,000
Hg leachability (%)	7.45×10^{-8}	6.68×10^{-6}	3.64×10^{-5}	3.73×10^{-5}
Hg volatility (%)	1.19×10^{-8}	2.01×10^{-8}	1.40×10^{-7}	9.23×10^{-8}
Hg initial leaching rate, K_w	205	7,640	95,500	58,300
Hg leaching parameter, a_w	0.53	0.74	0.83	0.74
Hg initial volatilization rate, K_g	34.2	49.6	320	214
Hg volatilization parameter, a_g	0.75	0.84	0.75	0.77
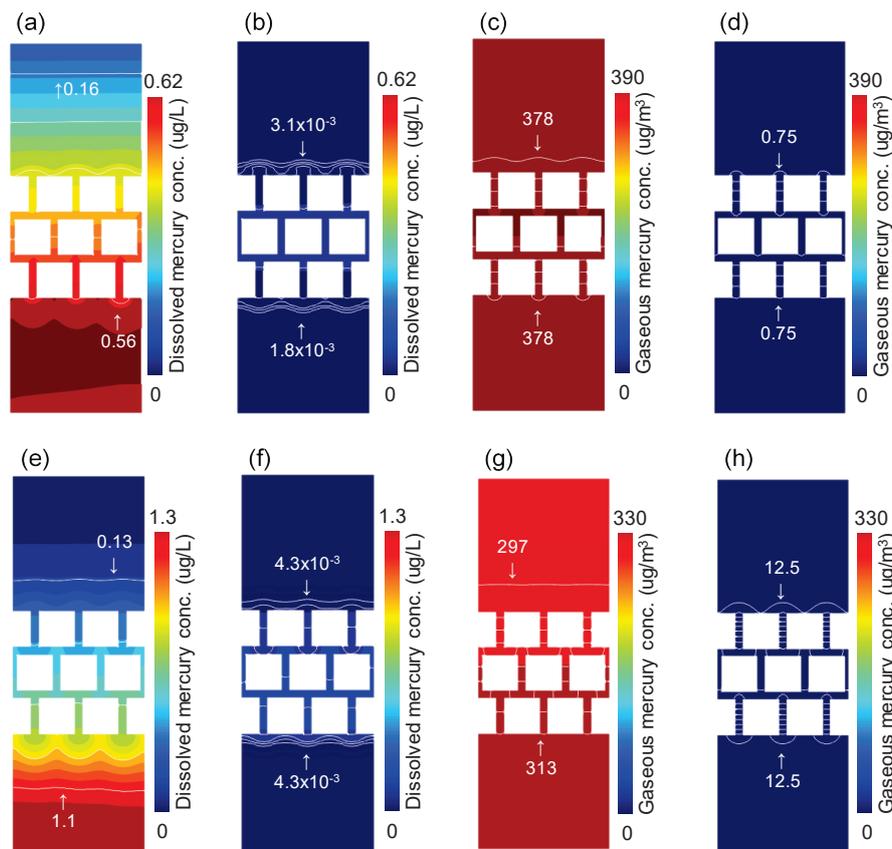
Fig. 3 Concentration distributions of mercury leaching and volatilization from sulfur polymer-solidified mercury waste in the 12th year after deposition (rainfall intensity = 0 mm/y for 0–10 years and 600 mm/y or 60 mm/y for > 10 years): (a) dissolved mercury concentration without a sorption layer with 600 mm/y, (b) dissolved mercury concentration with a sorption layer with 600 mm/y, (c) gaseous mercury concentration without a sorption layer with 600 mm/y, (d) gaseous mercury concentration with a sorption layer with 600 mm/y, (e) dissolved mercury concentration without a sorption layer with 60 mm/y, (f) dissolved mercury concentration with a sorption layer with 60 mm/y, (g) gaseous mercury concentration without a sorption layer with 60 mm/y, (h) gaseous mercury concentration with a sorption layer with 60 mm/y.

mercury in a landfill site in the 12th year after deposition, using high rainfall intensity values (600 mm/y) after an initial 10-year period with no rainfall. In these models, the disposed mercury waste was solidified with sulfur polymer, and the mercury leaching and volatilization rates from the solidified surface were taken to be $j_w^{SP} = 205t^{-0.53}$ and $j_g^{SP} = 34.2t^{-0.75}$, respectively (see Table 2). Significant effects of the soil sorption layer on the decrease in mercury concentration levels in the landfill were seen, as shown in Figs. 3(b) and 3(d).

In contrast, Figs. 3(e) to 3(h) show the results of modelling with lower intensity rainfall (60 mm/y) after the initial 10-year period: this lower rainfall intensity was used to simulate a cut-off layer covering the top of the landfill site, instead of a general final cover.

3.3 Mercury Concentration Profiles in Leachate and Landfill Surfaces

Figure 4(a) shows the concentration profiles of mercury dissolved in leachate accumulating in the drainage pipe at the bottom of the landfill where SP-solidified mercury metal waste was disposed. During the first 10 years, the concentration of mercury in the leachate was zero as, in the simulation, the landfill was covered by a roof, thereby preventing rainfall from reaching the surface. When the roof was removed (in the 10th year after deposition), rainfall began permeate the landfill, and mercury leached from the solidified waste was flushed towards the drainage pipe; hence, the dissolved mercury concentrations were calculated to increase after 10 years. The presence of a soil sorption layer was calculated to be able to decrease the peak concentration and delay the onset of leaching, whereas the use of a cut-off layer led to a substantial increase in peak concentration. It is thought that the flow velocity of rainfall passing through the cut-off layer would slow, thereby increasing the water retention time, and as a result, a greater proportion of mercury leaching from the solid waste could be dissolved in water accumulating in the layer pores. While the cut-off layer increased the peak concentration, the overall mercury flux would decrease due to the lower flow velocity, which is a major factor in determining mercury transport rates.

Figure 4(b) shows the concentration profiles of gaseous mercury emitted from the landfill surface: the gaseous mercury concentrations were calculated to increase over time, even during the first 10 years (in which no rainfall was considered to penetrate the landfill).

This is observed as gaseous mercury, which can be transported through pores by diffusion effects, even in the absence of water penetration. As was the case for mercury leaching, the soil sorption layer was also effective in decreasing the peak concentration of gaseous mercury and delaying its onset. It is noted that mercury sorption in this study is expressed by the equilibrium Henry model considering both adsorption and desorption, resulting in simulation of retardation transport not related with adsorption capacity.

The model showed that the use of a cut-off layer may decrease the peak concentration of gaseous mercury, despite its role in increasing mercury concentrations in leachate. Limiting the infiltration of rainfall using a cut-off layer can prevent the accumulation of water in the landfill site, thus maintaining pore widths in the layers—the gaseous mercury can then diffuse into these wide pores, which could lead to an overall decrease in the gaseous mercury concentration.

Figure 4(c) shows the fluxes of mercury dissolved in leachate accumulating in the drainage pipe at the bottom of the landfill. During the first 10 years, the flux of dissolved mercury in the leachate was zero because the landfill was covered. After 10 years, the dissolved mercury flux abruptly increased. Focusing on the effect of a cut-off layer, the presence of a cut-off layer can decrease the dissolved mercury flux. This effect on the flux differed from that on the concentration. The presence of the cut-off layer led the peak concentration to increase as shown in Fig. 4(a). Even if the concentration was increased due to the installation of the cut-off layer, the flow velocity transporting the dissolved mercury
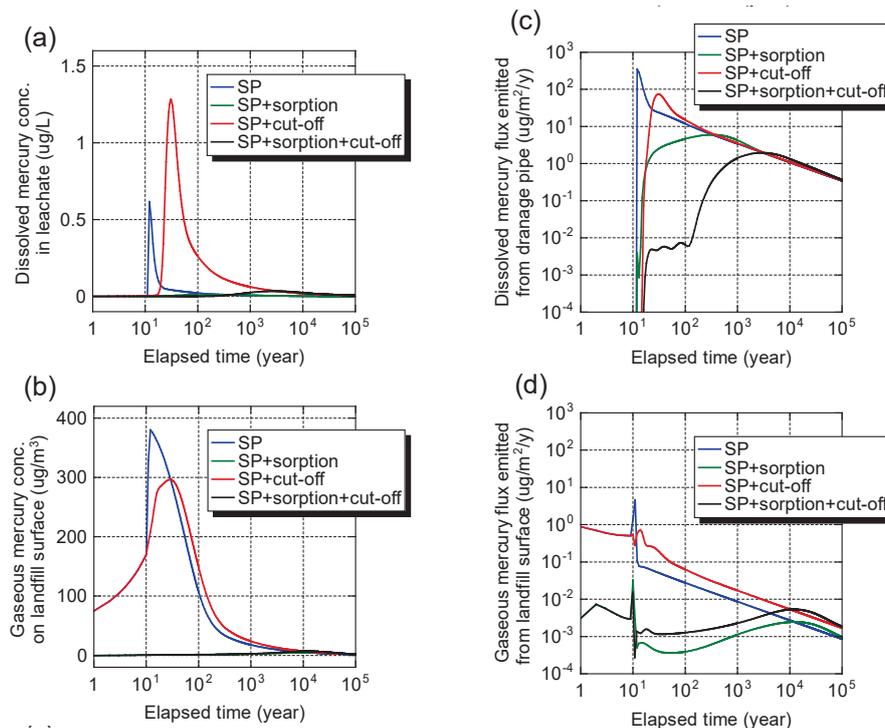

Fig. 4 Effects of soil sorption layer and cut-off layer on mercury concentration profiles: (a) dissolved mercury concentration in leachate in the bottom drainage pipe, (b) gaseous mercury concentration at the landfill surface, (c) dissolved mercury flux, (d) gaseous mercury flux.

Table 3 Summary of numerical simulation results.

Binder	Cut-off layer	Sorption layer	Leachate in drainage pipe		Gas emitted from landfill surface	
			Time that peak appears (year)	Peak conc. (ug/L)	Time that peak appears (year)	Peak conc. (ug/L)
Low-alkaline cement B (CB series)	No	No	12	94.4	12	30,900
	No	Yes	330	1.64	8,710	127
	Yes	No	30	195	32	30,300
	Yes	Yes	2,900	5.16	13,600	587
Sulfur polymer (SP series)	No	No	12	0.62	12	380
	No	Yes	330	0.01	12,400	4.93
	Yes	No	31	1.28	29	297
	Yes	Yes	2,900	0.03	12,200	7.32

decreased, so the flux also decreased. The installation of the cut-off layer was considered effective in reducing dissolved mercury emissions. On the other hand, the presence of the sorption layer affected the decrease in the flux of dissolved mercury over the simulation duration. The effect on the decrease was larger for the sorption layer than for the cut-off layer under the conditions of this analysis. The installation of the sorption layer was also effective in reducing dissolved mercury emissions.

Figure 4(d) indicates the fluxes of gaseous mercury emitted from the landfill surface. The presence of the sorption layer decreased the gaseous mercury flux. The flux could be kept to under 1×10^{-2} ug/m^2/y over the simulation duration. In contrast, the presence of the cut-off layer led the flux of gaseous mercury to increase slightly under these analytical conditions. When there was a cut-off layer covering the landfill surface, the amount of rainfall infiltration was limited. The degree of water saturation in the landfill decreased. Therefore, the effective diffusion coefficient of the gaseous mercury increased and the flux became large. Although installation of the sorption layer was effective in reducing gaseous mercury emissions, installation of the cut-off layer had no influence on reduction of the emissions.

Table 3 summarizes the results of the numerical simulations. The mercury metal waste solidified with low-alkaline cement B had a peak concentration of around 100 times higher than the sulfur polymer-solidified waste. In Japan, the water quality standard for effluents from controlled landfill sites sets a limit of 0.005 mg/L for mercury: when stabilizing waste mercury in its sulfide form and solidifying it with sulfur polymer, the mercury concentration in the leachate will be able to satisfy this quality standard; however, solidification with low-alkaline cement B was not found to satisfy the standard according to the calculations of this study. Moreover, the study indicates the necessity of installing a soil sorption layer to decrease peak mercury concentration.

3.4 Mercury Emissions from Landfill Sites

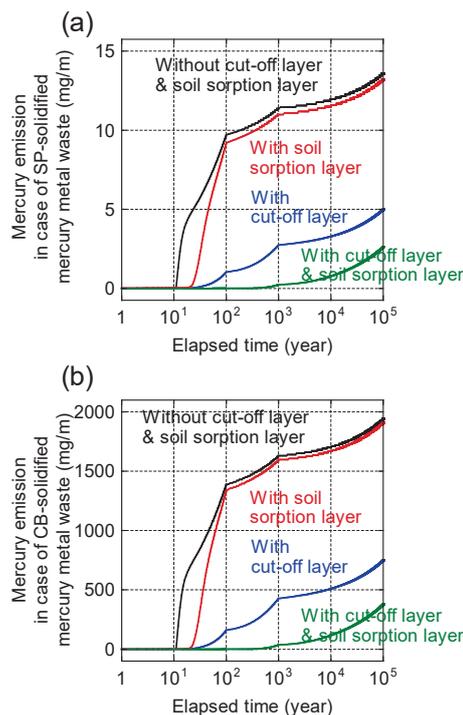Finally, regarding the total amount of mercury emissions from landfill sites, Figs. 5(a) and 5(b) show the

Fig.5 Total amount of mercury emitted from a landfill with SP-solidified or CB-solidified mercury metal waste disposal: (a) case of SP-solidified mercury metal waste, (b) case of CB-solidified mercury metal waste

total amount of mercury emitted from the landfill. In these figures, mercury metal waste was solidified using sulfur polymer or low-alkaline cement B, respectively. A drastic difference between sulfur polymer and low-alkaline cement B appeared in the total amount of mercury emissions. The total amount of emissions in the case of sulfur polymer was approximately 100 times smaller than in the case of low-alkaline cement B.

4. Conclusions

This study investigated the rates of leaching and volatilization of from mercury metal waste stabilized in its sulfide form and solidified with a sulfur polymer or low-alkaline cements. Numerical simulations were conducted to clarify the long-term behavior of mercury in landfill sites in which the stabilized, solidified mercury

waste was disposed. The following were found:

— According to the results of the serial batch leaching and volatilization tests, mercury waste solidified with the sulfur polymer presented excellent containment performance and the lowest leachability of the solidification materials used, 7.45×10^{-8}%. In contrast, when mercury waste was solidified with low-alkaline cements, the leachability was ~100 times greater.

— The sulfur polymer-solidified mercury waste also displayed the lowest volatilization, 1.19×10^{-8}%. Low-alkaline cement A performed similarly to the sulfur polymer in containing gaseous mercury emissions, while volatilization from low-alkaline cement B-solidified mercury waste was ~10 times greater.

— According to the numerical simulation results, when stabilizing mercury metal waste in its sulfide form and solidifying it with sulfur polymer, the resulting mercury concentration in the leachate will be less than 0.005 mg/L, thereby satisfying the Japanese water quality standard for effluent from controlled landfill sites.

— When using low-alkaline cements as a solidification agent, the dissolved mercury concentrations in the leachate were calculated to exceed 0.005 mg/L. The use of a soil sorption layer covering the solidified mercury waste was found effectively to decrease the peak concentration; in comparison, the application of a cut-off layer covering the solidified waste may substantially increase the dissolved mercury concentration, but simultaneously decrease its flux (emission), thereby also lowering its overall transport rate.

— The total amount of mercury emissions depended significantly on the presence of a soil sorption layer and cut-off layer as well as types of binders to solidify the mercury metal waste. The most effective countermeasure for reducing mercury emissions was considered to be sandwiching sulfur polymer-solidified mercury waste between sorption layers and covering the landfill surface with a cut-off wall. Numerical analysis will help us design the required geometry and material quality of the soil sorption layer, cut-off wall, and stabilized, solidified mercury metal waste.

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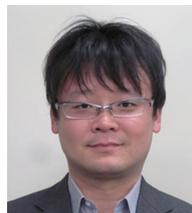
Hiroyuki ISHIMORI

Hiroyuki Ishimori, Ph.D., is a senior researcher at the National Institute for Environmental Studies, Japan. He specializes in mathematics and numerical simulations in geoenvironmental engineering. His skills are used for developing safe utilization and final disposal techniques for contaminated soils, wastes, byproducts and so on. He is currently studying the long-term deterioration of barrier performance of adsorption reagents, impermeable materials and reinforced concrete due to chemical attack. Numerical simulations considering long-term deterioration will provide effective countermeasures to protect the environment from various forms of chemical pollution.

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Ryo Hasegawa is a professional engineer at the Environmental Control Center Co., Ltd. He specializes in research and consulting on environmental measurement and waste management. His analytical technique in chemistry has significantly contributed to elucidation of the leaching and volatilization behavior of mercury-containing waste.

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Kazuto Endo heads the Fukushima branch of the National Institute for Environmental Studies, Japan. His research fields are environmental geotechnics and waste management. At present, his team is researching the management of radioactively contaminated waste and removed soil, aiming at early revitalization of Fukushima. He has researched the construction and management of seashore landfill sites through computer simulation and field surveys. He is also interested in the geotechnical use of recycled materials. He has published a guideline on geotechnical use for quality control of recycled waste gypsum powder.

Hirofumi SAKANAKURA

Dr. Hirofumi Sakanakura has worked for the Natural Institute for Environmental Studies since 2006. His main research area is development of leaching tests for wastes and recycled materials to evaluate environmental safety. He received his doctorate from Hokkaido University in 2000 and worked initially for Akita National College of Technology.

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Tomonori Ishigaki is a senior researcher at the National Institute for Environmental Studies. He serves as an editorial board member for the emission factor database and lead author of the Guidelines for National Greenhouse Gas Inventories for the Intergovernmental Panel on Climate Change. He is also serving as a member of the Technical Committee (Solid Recovered Fuels) of International Organization for Standardization. His published works cover subjects such as the mechanism of waste conversion in landfills, technology development for emission reduction from landfills, assessment of the fate of hazardous substances in the environment, and appropriate management of solid waste in urban areas of Asia to prevent urban floods.