Development of a low-rise industrial source dispersion model (METI-LIS model)

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Abstract: In Japan, with amendment of the Air Pollution Control Law in May 1996, various substances, including benzene and trichloroethylene, were newly designated as hazardous air pollutants, and environmental standards were established. In this situation, it is necessary to develop a dispersion model that is applicable to environmental impact assessment of industrial areas with a complex of factory buildings. To overcome this problem, modification of the ISC downdraught model was undertaken based on datasets from wind tunnel experiments by the Ministry of International Trade and Industries and Japan Environmental Management Association for Industry. This new model is called the METI-LIS model, and comparison shows that the performance of the model is better than that of the original ISC model.

Keywords: dispersion model, downdraught, ISC model, METI-LIS model, wind tunnel experiments.

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1 Introduction

In Japan, with amendment of the Air Pollution Control Law in May 1996, a number of substances were newly designated as hazardous air pollutants, and environmental standards were established as annual averaged concentrations. Consequently, environmental impact assessment (EIA) of chemical factories became necessary.

In atmospheric EIA of conventional air pollutants, such as SO_x or NO_x, the Pasquill–Gifford type plume models are widely used. However, these are not suitable for the prediction of dispersion of new hazardous air pollutants because the dispersion of these gases has different characteristics from the dispersion of conventional air pollutants. First, the emission source type is different. Conventional air pollutants are usually emitted from a taller stack than surrounding buildings with high momentum and high buoyancy, i.e. at a higher temperature than the atmosphere. On the other hand, the new air pollutants are usually emitted with low momentum and low buoyancy from relatively shorter stacks. In these cases dispersion is strongly influenced by the wake of buildings and usually down-draught occurs, causing relatively high ground-level concentrations. Thus it is necessary to use an effective and efficient atmospheric dispersion model that is applicable to EIA of industrial areas with a complex of factory buildings, taking downdraught into account.

The Industrial Source Complex (ISC) model [1] is a dispersion model that can take into account the downdraught effect due to building wake. However, this model is good at predicting the maximum concentration during some period, but not at predicting the horizontal distribution of 1-h averaged concentrations.

To overcome this problem, the development of an improved ISC downdraught model was undertaken by the Ministry of Economy, Trade and Industry (METI) and Japan Environmental Management Association for Industry (JEMAI). This newly developed dispersion model is called the METI-LIS (Ministry of Economy, Trade and Industry – Low-rise Industrial Source) dispersion model.

In this paper we report the outline and the methodology of development of the model and we also introduce some results of model comparison between the METI-LIS model and the original ISC downdraught model.

2 Methodology of development of the METI-LIS model

The model we wanted to develop should be able to predict not only the maximum concentration that will occur in a short period, but also the annual average concentration. Therefore, the model is expected to be able to effectively simulate the concentration distribution in the cross-wind (y) direction and not only the maximum concentration. This means the model must treat the dispersion parameters (such as σ_y, σ_z, H_e, etc., where σ_y and σ_z are lateral and vertical dispersion parameters and H_e is the effective stack height) correctly as a function of building arrangement, source position, wind direction, etc.

The method for the development of the METI-LIS model was to modify the ISC model based on the wind-tunnel experiment dataset, and to enable the model to treat more detailed source-type conditions than the original model.

The ISC model is based on a Gaussian-type plume model (Equation 1):

$$C = \frac{Q}{2\pi\sigma_y\sigma_z u} \cdot \exp\left[-\frac{y^2}{2\sigma_y{}^2}\right] \cdot \left\{\exp\left[-\frac{(H_e - z)^2}{2\sigma_z{}^2}\right] + \exp\left[-\frac{(H_e + z)^2}{2\sigma_z{}^2}\right]\right\} \tag{1}$$

where C is the concentration, Q is the source strength, and u is the wind speed. In the case where the source height is low and the gas dispersion can be influenced by the building wake, the downdraught must be taken into account. In the original ISC model the H-S (Huber-Snyder) [1] model is used to predict downdraught. The second column in Table 1 describes the original H-S model. As we can see, coefficients in the dispersion parameter, such as Cz_1, Cz_2, Cy_1 and Cy_2 have constant values in the H-S model. Therefore, values of σ_y and σ_z do not change even if the source type (source height or number of buildings, etc.) changes.

The third column in Table 1 lists an outline of the METI-LIS model. In the modification we tried to model Cz_i and Cy_i (i = 1, 2) as functions of the non-dimensional building height (Z_s/H_b), the aspect ratio of the building (W_b/H_b), building arrangement, wind direction, etc. Not only Cz_i and Cy_I, but also the following parameters were modelled in the same manner in the METI-LIS model.

- effective plume height H_e;
- wind speed used in the plume model;
- source position displacement Δy_s: introduction of a virtual source position, which is Δy_s away from an actual source position in the y direction.

Table 1 Outline of parameter modification.

Source height	ISC model	Parameter to be modified
$Z_s > 2.5H_b$ (without down-draught model)	$\sigma_y = 465.1 \times x \tan(TH)$ $TH = 0.1745[c - d \ln(x)]$ $\sigma_z = ax^b$ where a, b, c and d are constants determined by the atmospheric stability	Same as ISC model
$Z_s < 2.5H_b$ (use down-draught model)	H-S model (except for squat building) $\sigma_z' = Cz_1 \cdot h + Cz_2 \cdot (x - 3h)$ $3h \le x < 10h$ $\sigma_z' = \sigma_z(x + x_z)$ $10h < x$ $\sigma_y' = Cy_1 \cdot W_b + Cy_2 \cdot (x - 3H_b)$ $3H_b \le x < 10H_b$ $\sigma_y' = \sigma_y(x + x_y)$ $10H_b < x$ where, h = Min(H_b, W_b), Cz_1= 0.7, Cz_2= 0.067, Cy_1= 0.35, Cy_2 = 0.067 and x_z, x_y are virtual downwind distance from a source.	(1) Cz_i, Cy_I (i = 1, 2) (2) Effective plume height H_e (3) Wind speed used in the plume model. (4) Source position displacement Δy_s These parameters are modified as functions of building width, height, building arrangement and source position, etc.

The displaced source position is applied in the case where the wind direction is not perpendicular to the building wall and the source position is located near upwind of the wall of building. In this situation, the plume of the gas runs along the wall, and the effect is that the source position is displaced in a lateral direction (see Figure 1).

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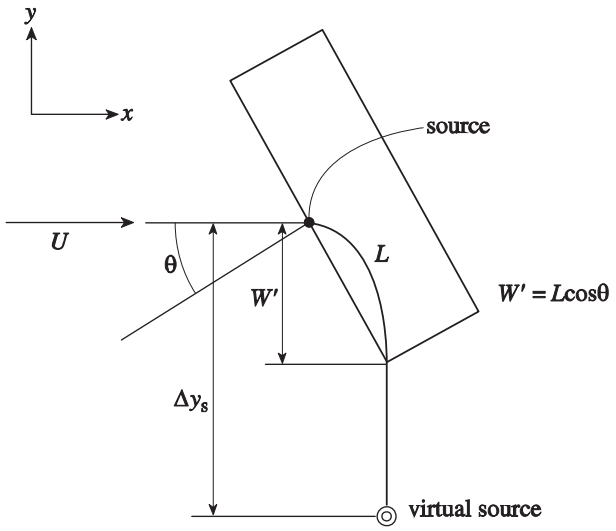


Figure 1 Introduction of source position displacement.

In this study, all these parameters were modelled based on the data obtained from wind-tunnel experiments. For the purpose of investigating the features of gas dispersion around buildings and for obtaining the data needed for the modification and validation of the model, two types of wind-tunnel experiment were carried out. One type is the 'simple-buildings experiment' and the other is the 'factory-buildings experiment'.

In the simple-buildings experiments, 1/200 scale models of simple cubic or squat buildings were used. The data were collected under a total of 44 different conditions, i.e. different source height, aspect ratio of the building, wind direction, arrangement of buildings, and so on. The summary of the experimental conditions is shown in Table 2 and Figure 2, while an example of the arrangement of buildings is shown in Figure 3a. The ground-level concentrations, vertical profiles of concentration and wind speed were measured, and the data obtained from these measurements, such as the lateral standard deviation (σ_y) and the vertical standard deviation (σ_z), were used to modify the downdraught model.

Table 2 Summary of simple-buildings experiments conditions.[a]

Condition of buildings	
Shape of buildings	W_b/H_b = 1, 3, 5 L_b/H_b = 1
Number of buildings (arrangement: row × line)	1, 2, 3, 9 (3 × 3), 25 (5 × 5), 35 (5 × 7)
Wind direction	θ = 0, 20, 40 [deg]
Source height	Z_s/H_b = 0.5, 1.0, 1.5
Meteorological conditions	
Wind speed	4 m/s
Atmospheric stability	Neutral condition

[a] W_b, H_b, L_b = width, height and length of buildings; Z_s = source height; θ = wind direction.

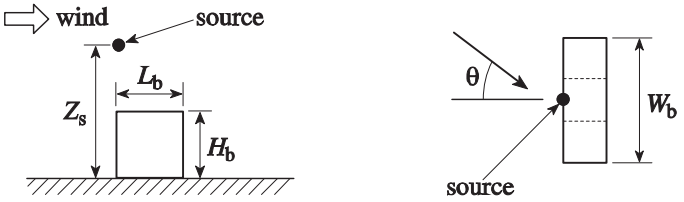
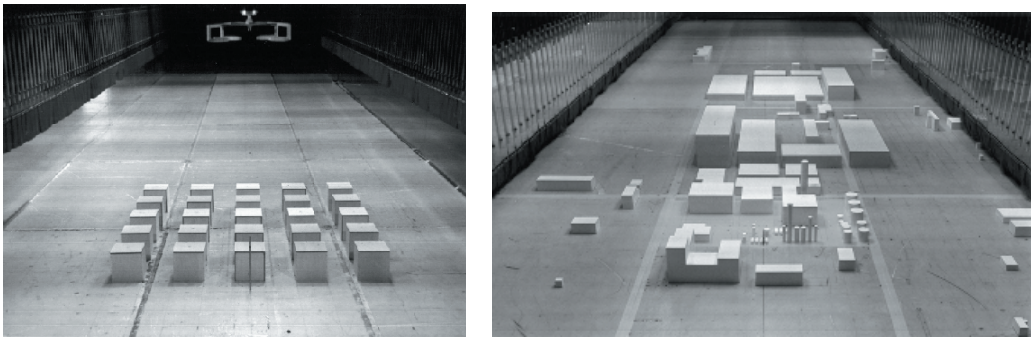


Figure 2 Parameters of building shape.



(a) (b)

Figure 3 Examples of scale models used in the experiments: (a) simple-buildings experiments (buildings arrangement 5 × 5); (b) factory-buildings experiments.

In the factory-buildings experiments, 1/200 scale models of three different chemical factories were used. The data were collected from a total of five tests, in which the ground-level concentrations were measured. An example of a factory-buildings model is shown in Figure 3b. The data were used for the validation of the model itself and of the methods for the model application to actual industrial areas, which are composed of many different types of building.

The Large Scale Dispersion Wind Tunnel was used for all the wind-tunnel experiments described above, and the experiments were conducted by the Nagasaki Research & Development Center of Mitsubishi Heavy Industries in 1998 and 1999. The wind tunnel has a working section 3 m wide, 2 m high and 25 m long. The neutral atmospheric condition was simulated in the working section by using surface roughness and spires located in the upstream section of the tunnel. In the modelling of the METI-LIS model, we used the data obtained only in the neutral atmospheric condition because in the downdraught region the turbulence produced mechanically is more dominant than the turbulence caused by thermal effects.

The performance of the METI-LIS model was examined with the datasets from wind-tunnel experiments and several field experiments.

The procedure for the development of the METI-LIS model is shown in Figure 4.

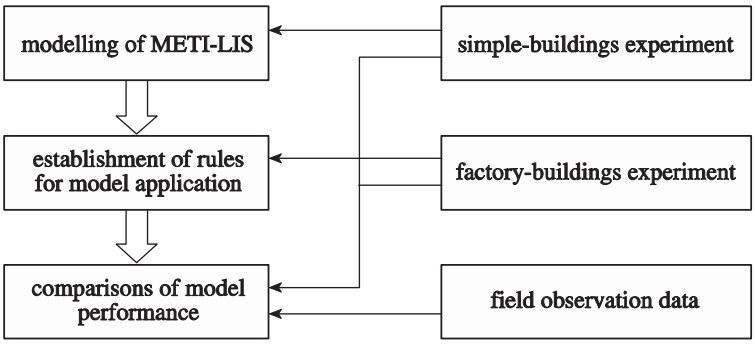


Figure 4 Procedure for development of the METI-LIS model.

3 Model description

All the modelled parameters are based on the data obtained from simple-buildings experiments. The experimental results showed that most of the parameters to be modified have obviously different characteristics when a small group of buildings or a large group is considered, where 'large group' means that a large area is occupied by buildings, for example the number of buildings is 5×5 or 5×7 in the simple-buildings experiments. On the other hand, 'small group' means that a small area is occupied by buildings, for example the number of buildings is 1 or 2. Therefore, as a first step of modelling, we classified the buildings in two major categories, i.e. 'large group type' and 'small group type', and then, for each group, model formulas of every parameter were determined.

3.1 Model of Cy_i and Cz_i

Dispersion parameters (σ_y, σ_z) were estimated by analysing the lateral distribution of ground-level concentrations and the vertical concentration distribution obtained in simple-buildings experiments, where each distribution was assumed to be Gaussian. Figure 5 shows examples of σ_y and σ_z obtained from simple-buildings experiments. The lines in the figure also show the results of the H-S model, where Cy_i and Cz_i are not fixed as in the original H-S model but are obtained by the least-squares method. As we can see, the values of σ_y and σ_z increase linearly as the downwind distance increases, and regarding σ_y, the initial dispersion width increases as the building width increases while the initial dispersion width of σ_z does not vary with the building width. These tendencies accord with expressions of the H-S model, and the lines based on these expressions fit the experimental values well. These facts validate the use of the same type of downdraught model as the H-S model.

For conciseness reasons, we present only an example of the formulas for Cz_1. Based on data for σ_z obtained from wind tunnel experiments, we found the most suitable value of Cz_1 by the least-squares method. Figure 6 shows the summary of Cz_1. As we can see, Cz_1 is not constant as in the original H-S model: it looks like an almost linear function of non-dimensional source height and a function of the aspect ratio of the building, and it is one of the rare parameters not influenced very much by the group type of the buildings. As shown in Table 1, Cz_1 is the parameter that represents the initial vertical dispersion width caused by downdraught. The tendency in Figure 6 means that initial vertical

dispersion width is larger when the non-dimensional source height is small and the building width is large. This result seems to be physically appropriate.

Source height	Number of buildings	Wind direction	W_b/H_b	
$Z_s/H_b = 0.5$	1	0°	1.0	○
		0°	3.0	△
		0°	5.0	□

Source height	Number of buildings	Wind direction	W_b/H_b	
$Z_s/H_b = 0.5$	1	0°	1.0	○
		0°	3.0	△
		0°	5.0	□

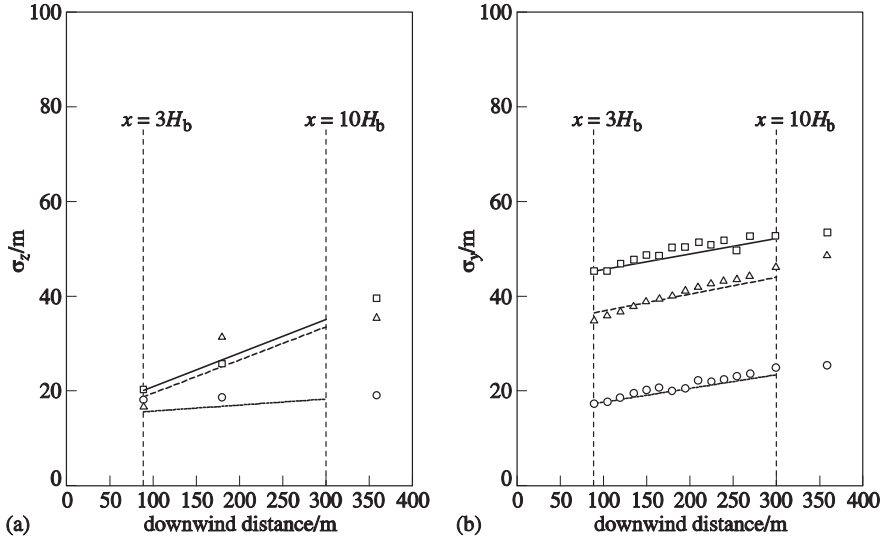


Figure 5 Dispersion parameters: (a) vertical dispersion; (b) lateral dispersion. Symbols show experimental values; lines show the downdraught model.

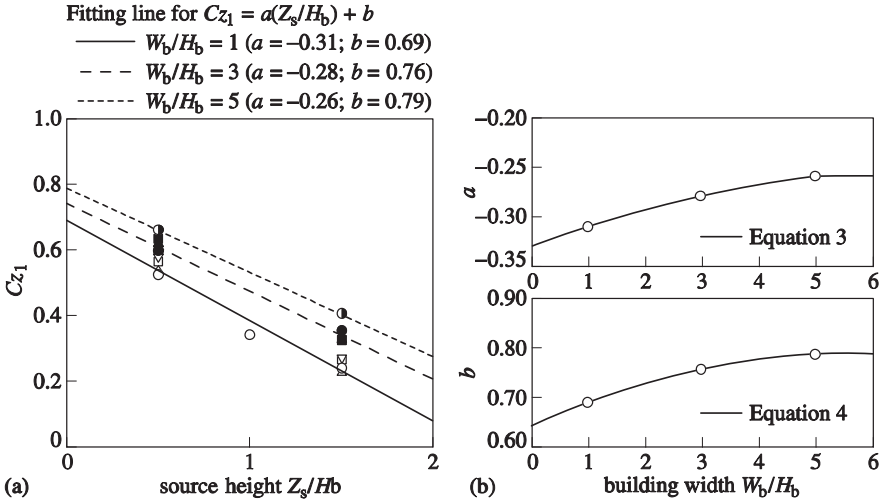


Figure 6 (a) Cz_1 as a function of source height; (b) model of coefficients a and b. For $W_b/H_b = 1$: number of buildings: ○ = 1; △ = 2; □ = 5 × 5; ▽ = 5 × 7. For $W_b/H_b = 3$, ● = 1, ▲ = 2, ■ = 5 × 5. $W_b/H_b = 5$, ◑ = 1).

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The lines in Figure 6 are the results of fitting of Cz_1 as a linear function of the non-dimensional source height. Based on these results, we modelled the parameter Cz_1 as Equations 2, 3 and 4, where a and b are functions of the aspect ratio.

$$Cz_1 = a(Z_s/H_b) + b \qquad [\text{for } \theta = 0\,] \tag{2}$$

$$a = -0.00125(W_b/H_b)^2 + 0.02(W_b/H_b) - 0.329 \qquad [\text{for } W_b/H_b \le 5]$$

$$a = -0.26 \qquad [\text{for } W_b/H_b > 5] \tag{3}$$

$$b = -0.045(W_b/H_b)^2 + 0.051(W_b/H_b) + 0.645 \qquad [\text{for } W_b/H_b \le 5]$$

$$b = -0.788 \qquad [\text{for } W_b/H_b > 5] \tag{4}$$

3.2 Model of effective stack height

When the downdraught occurs, the effective stack height H_e in Equation 1 seems to be smaller than the actual source height owing to the wake of the buildings. In the METI-LIS model, this decrease of H_e is also modelled.

Non-dimensional source heights used in the simple-buildings experiments were Z_s/H_b = 0.5, 1.0 and 1.5. By many test calculations, changing H_e for each experimental case, we found the most suitable plume height for each case. Based on these trials, the modelling of the effective stack height was done. Actually in the modelling, for the case of $Z_s/H_b < 0.5$, we used the result of $Z_s/H_b = 0.5$ and for the case of $0.5 < Z_s/H_b < 1.0$, we used the result of $Z_s/H_b = 1.0$, for the case of $Z_s/H_b > 1.0$, we used the result of $Z_s/H_b = 1.5$. H_e is now expressed as a function of Z_s and the aspect ratio of the building:

For $1.0 < Z_s/H_b < 2.5$

$$H_e = 0.44Z_s \quad [\text{for } W_b/H_b > 1] \tag{5}$$

$$H_e = 0.56Z_s \quad [\text{for } W_b/H_b \le 1, \text{ for small group type}] \tag{6}$$

$$H_e = 0.67Z_s \quad [\text{for } W_b/H_b \le 1, \text{ for large group type}]$$

3.3 Model of effective wind speed

The experimental results show that the wind speed in the presence of buildings is usually smaller than the wind speed above flat terrain due to drag force caused by buildings. Considering this effect, the wind speed to be used in the Equation 1 is reduced in the METI-LIS model. This reduced value is the effective wind speed. Based on the wind speed measured in the simple-buildings experiments, the reduction ratio α was modelled. The definition of α is as follows:

$$u = \alpha u_s \tag{7}$$

where u is the effective wind speed and u_s is the wind speed above flat terrain. An example of the modelling is shown below:

For $H_e/H_b > 1.0$ ($\theta = 0$)

$$\alpha = 0.76 \qquad [\text{for } W_b/H_b < 1] \tag{8}$$

$$\alpha = 0.8 - 0.039(W_b/H_b) \qquad [\text{for } 1 \le W_b/H_b \le 5]$$

$$\alpha = 0.61 \qquad [\text{for } W_b/H_b > 5]$$

3.4 Model of source position displacement

When the wind direction is not perpendicular to the building wall and the source position is located near upwind of the wall of building, the plume runs along the wall and the effect is as if the source position were displaced in a lateral direction. To take this effect into account, a displaced source position, i.e. a virtual source which is Δy_s away from an actual source in a lateral direction, is introduced.

Wind directions used in the simple-buildings experiments were $\theta = 0$, 20° and 40°. By many test calculations changing Δy_s for these each experimental case of $\theta = 20°$ and 40°, we found the most suitable Δy_s for each case. Based on these trials, Δy_s was modelled, as a function of building width, non-dimensional source height, etc.

In this model, Δy_s increases as the non-dimensional source height decreases and it has large value especially when $Z_s < H_b$. Considering this source displacement is due to the effect of a building wall, this modelling seems to be physically appropriate.

An example of the modelling is shown below:

For 'large group type'

$$\begin{aligned} \Delta y_s/W' &= 2.5 && [\text{for } Z_s/H_b \le 1.0] \\ \Delta y_s/W' &= -3.0(Z_s/H_b) + 5.5 && [\text{for } 1.0 < Z_s/H_b \le 1.83] \\ \Delta y_s/W' &= 0.0 && [\text{ for } Z_s/H_b > 1.83] \end{aligned} \tag{9}$$

where

$$W' = L \cos\theta \tag{10}$$

Figure 7 compares the results of the calculations with and without the source displacement. Each graph represents correlation of ground-level concentrations (standardized by wind speed and source strength) between an experimental result and the corresponding calculation. It can be seen that the performance of the model with the source displacement is better than the model without the source displacement.

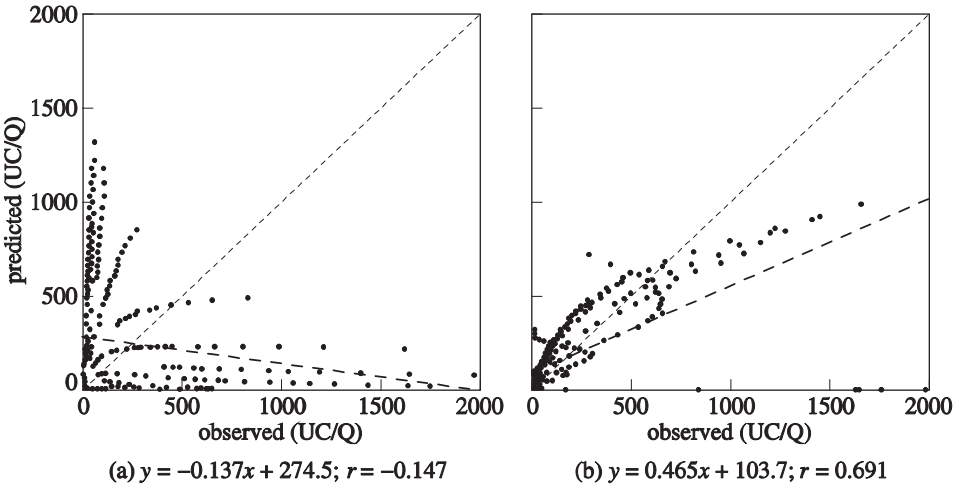

Figure 7 Effect of source position displacement (r = correlation factor). (a) Without displacement. (b) With displacement, where $Z_s/H_b = 0.5$, $W_b/H_b = 3$, $\theta = 20°$, number of buildings = 1.

4 Establishment of rules for model application

Information about the width and height of a building is required for the METI-LIS model's input data. Additionally, the type of building, i.e. 'small group type' or 'large group type', must be determined.

However, obtaining such information for an actual factory is not an easy task because there are usually many buildings, which have different sizes, shapes and distances from a source.

Therefore our next task was to establish rules for the application of the model to actual factory buildings. Our aims were:

- To describe a method indicating how to determine the type of the factory (i.e. small group or large group);
- To describe a method indicating how to choose one building, which is the most dominant in gas dispersion, among many different buildings in a factory. Data for the width and height of this building form part of the input to the METI-LIS model.

The former method was developed by comparing the correlations among simple-buildings experiments. As described before, characteristics of ground-level concentration in experimental cases of building arrangement 5 × 5 or 5 × 7 are different from those of 1, 2 or 3 × 3. Therefore, as a rough approximation, based on the building area, building height and distance from a source of experimental case of 3 × 3, we evolved a method for the determination of the type of building, as shown below

> If the area occupied by the group of buildings is larger (smaller) than a rectangle composed of lines which have $5H_b$ length in the wind direction and $6H_b$ in the crosswind direction, the group is defined as 'large (small) group', where H_b is height of the building that is the most dominant in gas dispersion. In this method, no attention is paid to the location of the source (see Figure 8).

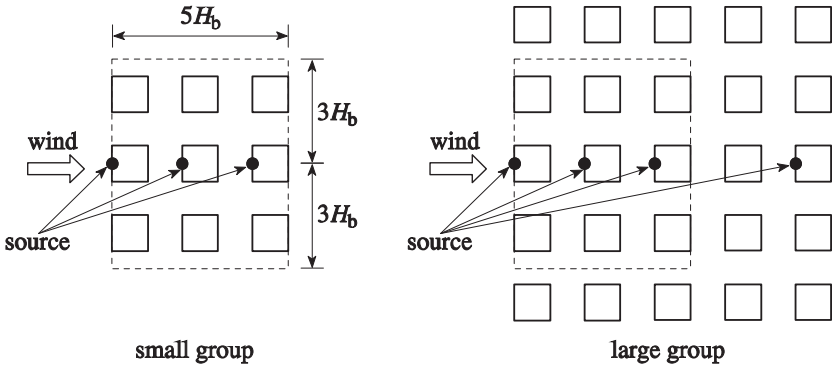

Figure 8 The method for determination of type of building.

The latter method was examined with many test calculations simulating the factory-buildings experiments. First, we looked at all buildings near the source and tried test calculations for each building's width and height. Then we compared the ground-level concentrations with the results of a factory-buildings experiment. In this manner, we

found the most dominant building for every five cases of factory-buildings experiment. Considering shape and position of the most dominant building in each five cases, we developed a method for the determination of the most dominant building among the many buildings near a source. The method is as follows:

1 Examine the buildings near a source.

2 If the distance between the source and the nearest part of the building is within $5L^*_i$ in the wind direction and within L^*_i in the crosswind direction, the building is listed as an influential building in gas dispersion, where L^*_i is the smallest value of H_{bi} or W_{bi} and subscript i means each building around the source (see Figure 9)

3 For each influential building, calculate H_{Gi} (= H_{bi} + 1.5L^*_i : i.e. the GEP stack height [2]) and choose the building that has the largest H_{Gi} value as the most dominant building.

4 The width and height of the most dominant building (W_b and H_b) are used in the downdraught model.

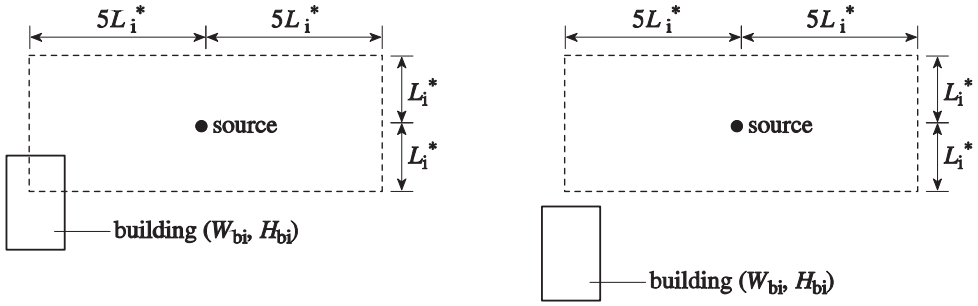

Figure 9 Determination of the most influential building: (left) with influential building; (right) without influential building.

Of course, this method was invented based on wind-tunnel data using only three different factory models. It should be tested in other many factories.

5 Simulation results and model comparisons

Performances of two models (the ISC downdraught model and the METI-LIS model) were compared. The datasets used in the comparison are as follows.

- Simple-buildings experiments: wind-tunnel experiment
- Factory-buildings experiments: wind-tunnel experiment
- Millstone nuclear power station [3]: field observation
- Tsurumi Point [4] in Japan: field observation

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Both field observations were carried out for the purpose of obtaining data for dispersion under the influence of the building near a source, and the source heights in both experiments were around or below roof level.

Several comparison results are shown in Figures 10 and 11. Figure 10 shows examples of the comparisons based on the datasets from wind-tunnel experiments. We can see that the performance of the METI-LIS model is better than that of the original model. The same tendency was observed in almost all other comparisons regarding wind tunnel experiments we have done.

On the other hand, Figure 11 shows the comparison based on field observations, where the tendency is not so obvious as in Figure 10 because of fluctuations of meteorological conditions and a relatively small number of sampling points. However, we can see also the modified model is a little bit better than the original model.

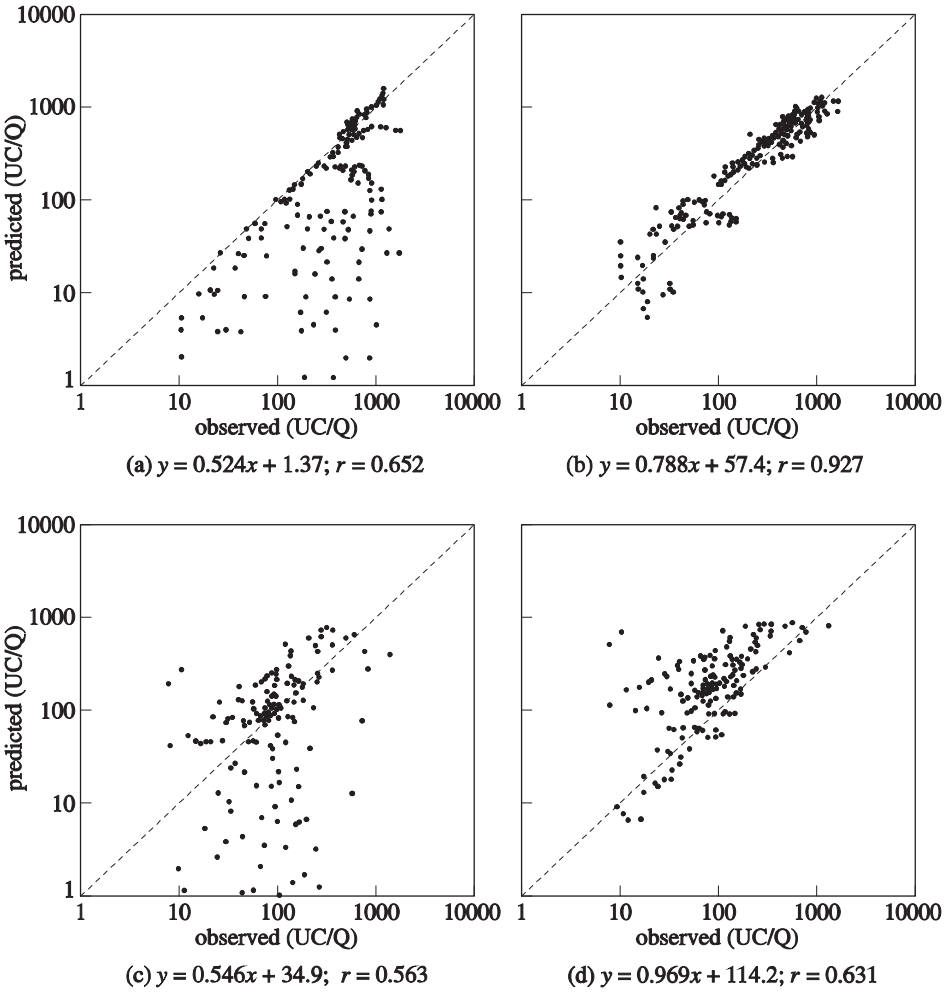

Figure 10 Comparison between METI-LIS model and original model (wind-tunnel data).(a) ISC vs. simple building experiment (5×7, $Z_s/H_b = 0.5$, $W_b/H_b = 1.0$, $\theta = 0°$). (b) METI-LIS vs. simple building experiment (5×7, $Z_s/H_b = 0.5$, $W_b/H_b = 1.0$, $\theta = 0°$). (c) ISC vs. factory building experiment. (d) METI-LIS vs. factory building experiment.

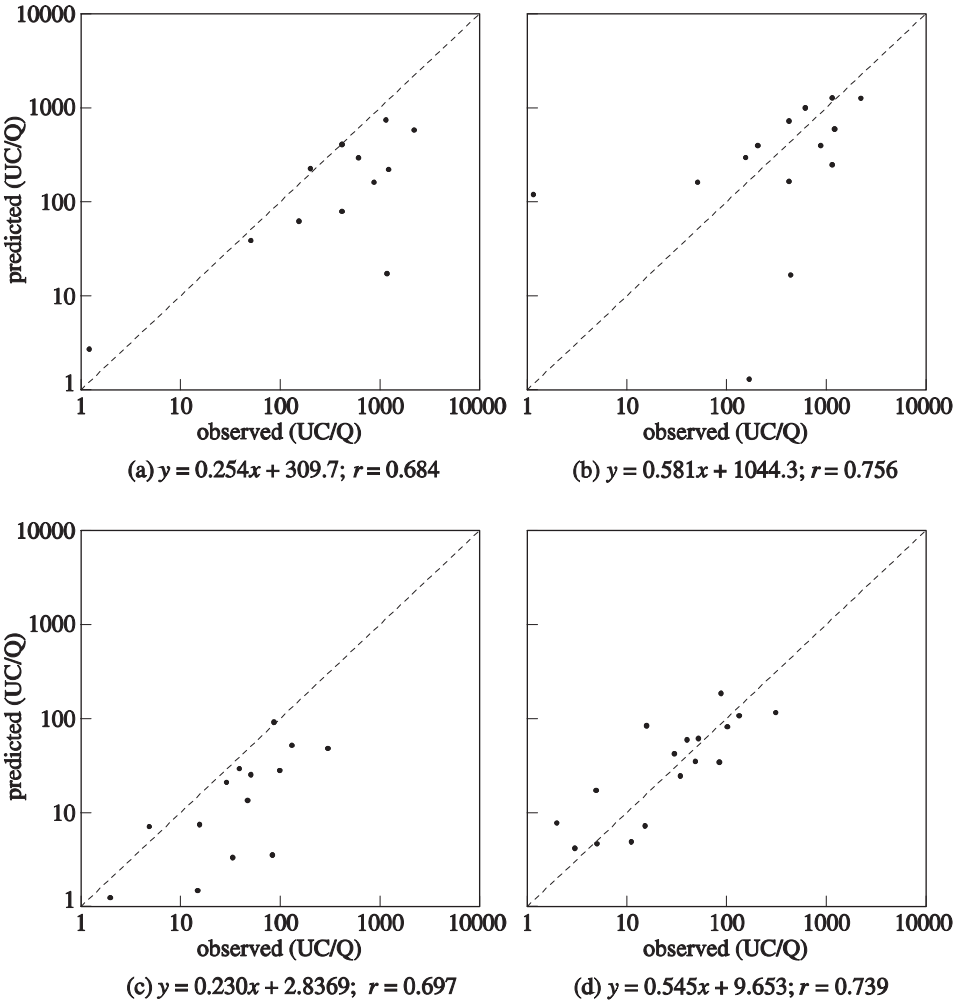

Figure 11 Comparison between METI-LIS model and original model (field observation): (a) ISC vs. Tsurumi Point (Run 8, 6/7/1971); (b) METI-LIS vs. Tsurumi Point (Run 8, 6/7/1971); (c) ISC vs. Millstone (Run 18, 25/10/1974); (d) METI-LIS vs. Millstone (Run 18, 25/10/1974).

6 Conclusion

For the purpose of developing a dispersion model applicable to the environmental impact assessment of industrial areas, the ISC downdraught model was improved based on the data of 'simple-buildings experiments' and the rules to apply this model to actual factories were examined using 'factory-buildings experiments'. The model was tested using wind-tunnel experiments and several field experiments, and proved to perform better than the original model. Investigations of the effect of atmospheric stability and comparisons with other downdraught models [5] are continuing.

This modified dispersion model is called the METI-LIS model. At present, the METI-LIS model has been packaged as user-friendly software, which runs on a personal computer and has been released on our internet homepage (www.kantou.meti.go.jp/sesaku/recycle/meti-lis_top.htm or www.jemai.or.jp/ems/meti-lis.htm)

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Modeling of atmospheric dispersion of mercury from coal–fired power plants in Japan

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ABSTRACT

The Air Quality Management Division of Ministry of the Environment in Japan selected the maximum level of annual mean air quality standard for mercury as 0.04 μg/m^3. The yearly average atmospheric emissions of mercury from two nearly located point sources, background concentrations of mercury in the atmosphere and one–year meteorological data was used to predict the ambient concentrations of mercury at ground level by atmospheric dispersion modeling. To estimate the mercury concentration in the air of the local area, two different models have been used. The first one is AIST–ADMER model that estimates regional atmospheric distribution of mercury concentration. The second one is METI–LIS model that estimates the atmospheric distribution of mercury concentration in the vicinity of industrial facilities. The annual mean concentration of mercury in the atmosphere was calculated for the central Honshu Island of Japan using the AIST–ADMER model, which served as a background data for the METI–LIS model to calculate atmospheric mercury concentration in the vicinity of industrial facilities. Maximum annual mean atmospheric concentrations of mercury in the vicinity of the two hypothetical coal–fired power plants were calculated as 0.0118 μg/m^3 that was lower than the Japanese annual mean air quality standard for mercury.

Keywords:
Dispersion modeling
Atmospheric mercury concentration
METI–LIS
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1. Introduction

In Japan, mercury was categorized as a hazardous air pollutant (HAP) in 1996 and is on the list of "Substances Requiring Priority Action" published by the Central Environmental Council of Japan (Kida, 2005). The Central Environmental Council prepared the second report "Future direction of measures against hazardous air pollutants" in October 1996, which also proposed that the voluntary action to reduce emissions, as well as an investigation of hazards, atmospheric concentration and pollution sources should be promoted. Although the industrial emissions of mercury in Japan have decreased in recent years (Ito et al., 2006), primarily due to the voluntary reduction of mercury emissions from industrial sources, the concentration distribution of these pollutants in the local atmospheric environment has remained largely unknown (Shirane, 2007).

Mercury is a natural trace component in the environment. Notwithstanding, the bioaccumulation of methylmercury (MeHg) via the food chain, especially through fish, concentrates mercury and poses serious toxicity hazards to the biosphere (Harada, 1995). For that reason, natural and anthropogenic emissions of mercury in the environment (Nriagu and Pacyna, 1988), its transportation and fate (Schroeder and Munthe, 1998; Boening, 2000), and its adverse effects on human health and the ecosystem (Ditri, 1991) have all attracted great attention as facets of a major environmental problem. Stack emissions from coal–combustion power industry includes both vapor and particle–bound phases. Reactive gaseous mercury [RMG or Hg(II)] (Schroeder and Munthe, 1998) can be inorganic (e.g., mercuric chloride, $HgCl_2$) or organic [e.g., methylmercury (MeHg)]. It can also be present as particulate mercury (e.g, mercuric oxide, HgO, or mercury sulfide, HgS). In the global atmosphere, gaseous elemental mercury [GEM or Hg(0)] is the dominant form. Hg(II) typically constitutes a small percentage of total mercury and is predominantly in the gas phase. MeHg concentration in the atmosphere is relatively low, about 10% – 30% lower than total Hg(II) concentrations, according to analysis of precipitation samples (Seigneur et al., 1998). However, Hg(II) becomes methylated in water bodies, where it can bioaccumulate in the food chain. Hg(0) is sparingly soluble in cloud particles and is not removed significantly by wet deposition, and its dry deposition velocity is also believed to be low. As a result, Hg(0) has a long atmospheric lifetime. On the other hand, Hg(II) is quite soluble with cloud particles, so is removed rapidly by wet and dry deposition processes, and has much shorter atmospheric lifetimes (Hedgecock and Pirrone, 2004). Particulate mercury [PM or Hg(p)] is mostly present in the fine fraction of particulate matter ($PM_{2.5}$), although some Hg(p) may be present in coarse PM (Landis and Keeler, 2002).

The concentration of mercury should be estimated both on a regional scale as well as on a local scale, because not only the concentration of mercury in the general environment is important (i.e. the area which includes most of the total population), but also those in the vicinity of industrial sources (i.e. areas of high concentration) should be considered carefully, as particular industrial sources are expected to be associated with relatively high–risk areas. In this study, two different models have been

selected, which were used to assess the extent of exposure: the AIST–ADMER (National Institute of Advanced Science and Technology–Atmospheric Dispersion Model for Exposure and Risk Assessment) estimates regional concentration distribution of hazardous chemical substances (Higashino et al., 2003; Higashino et al., 2004), and the METI–LIS (Ministry of Economy, Trade and Industry–Low–Rise Industrial Source Dispersion Model) estimates the concentration distribution in the vicinity of particular industrial facilities (Kouchi et al., 2004).

Gaseous mercury, including both Hg(0) and Hg(II), were considered as total mercury emissions in the atmosphere, which served as input emission data for these two air pollutant dispersion models. More than 99.5% of mercury in the stack emissions was in the gaseous form (Lindqvist and Rodhe, 1985) and the proportion in particulate form was extremely low in Japan (Yokoyama et al., 2000). Since mercury treatment systems of the coal combustion facilities are very advanced in Japan, Hg(II) emission from the stack is also very low (Takahashi et al., 2008).

The objective of this study was to estimate the concentration of mercury in Japan, whereas the above mentioned two models were used for the assessment of the atmospheric concentration of mercury.

2. Method

2.1. AIST–ADMER model

The AIST–ADMER (Higashino et al., 2003; Higashino et al., 2004) version 1.5e is a series of models and systems designed for estimating the regional atmospheric level of chemicals, developed by the National Institute of Advanced Industrial Science and Technology. The functions of the AIST–ADMER model provide the following calculations and simulations:

- Generation and confirmation of meteorological data
- Generation and confirmation of chemical substance emission data
- Calculation of atmospheric concentrations and deposition of chemicals
- Graphical images of calculation results
- Calculation of resulting histogram
- Population exposure assessment

The purpose of this model is to estimate a long–term, average distribution of chemical concentration in a relatively wide region, such as the Kanto and Kansai areas of Japan. Atmospheric concentration distribution of chemical substances of a 5 km × 5 km square spatial grid for an average of one month to one year can be calculated by this model. Generally, use of models requires preparation of various data, such as meteorological data, creating target substance emission data, and setting calculation param–eters, in order to estimate the atmospheric concentration of chemicals and assess their exposure.

In this study, meteorological input data, calculated monthly for a year, i.e., from January to December 2006, have been used for the AIST–ADMER model calculation. Meteorological input data were produced from AMeDAS (Automated Meteorological Data Acquisition System) (Akasaka and Nimiya, 1986) data, whereas the solar radiation and cloud cover were obtained from individual weather stations.

Simulations calculated from the AIST–ADMER model need information on target substances, such as the amount and geographical location (i.e., latitude, longitude) of emission etc. The AIST–ADMER contains a function for creating the gridded emission data required for the calculation. The methods used for creating gridded emission data can be classified mainly into two types, i.e., point sources, which specify a location using latitude and longitude, and enter the emissions generated from the location, and area sources, which specify emissions for each region or city, and allocate the emissions to calculation grids according to indices such as population, area, industrial statistics, and traffic volume.

The AIST–ADMER model calculation range consists of a number of calculation grids. Total 11 calculation ranges are pre–registered in ADMER in order to cover overall Japanese region (Table 1). Before starting the simulation, it is recommended to select a calculation range that includes target ranges.

***Table 1**. ADMER Calculation range*

Name	Range	Number of grids	Regions
Hokkaido	E 139° 15' 00" – 145° 56' 15" N 41° 17' 30" – 45° 35' 00"	107 × 103	Hokkaido
Tohoku	E 139° 07' 30" – 142° 11' 15" N 36° 45' 00" – 41° 37' 30"	49 × 117	Aomori, Iwate, Miyagi, Akita, Yamagata, Fukushima
Hokuriku	E 136° 07' 30" – 139° 56' 15" N 36° 02' 30" – 38° 35' 00"	61 × 61	Niigata, Toyama, Ishikawa
Kanto	E 138° 18' 45" – 140° 56' 15" N 34° 50' 00" – 37° 12' 30"	42 × 57	Ibaraki, Tochigi, Gunma, Saitama, Chiba, Tokyo, Kanagawa
Chubu	E 135° 22' 30" – 139° 11' 15" N 35° 07' 30" – 37° 05' 00"	61 × 47	Fukui, Yamanashi, Nagano, Gifu
Tokai	E 135° 48' 45" – 139° 15' 00" N 33° 40' 00" – 35° 40' 00"	55 × 48	Shizuoka, Aichi, Mie
Kinki	E 134° 11' 15" – 136° 30' 00" N 33° 22' 30" – 35° 50' 00"	37 × 59	Shiga, Kyoto, Osaka, Hyogo, Nara, Wakayama
Chugoku	E 130° 41' 15" – 134° 33' 45" N 33° 40' 00" – 35° 40' 00"	62 × 48	Tottori, Shimane, Okayama, Hiroshima, Yamaguchi
Shikoku	E 131° 56' 15" – 134° 56' 15" N 32° 37' 30" – 34° 37' 30"	48 × 48	Tokushima, Kagawa, Ehime, Kochi
Kyushu	E 128° 15' 00" – 132° 11' 15" N 30° 55' 00" – 34° 17' 30"	63 × 81	Fukuoka, Saga, Nagasaki, Kumamoto, Oita, Miyazaki,
Okinawa	E 122° 48' 45" – 131° 26' 15" N 24° 00' 00" – 27° 57' 30"	138 × 95	Okinawa

2.2. METI–LIS model

The METI–LIS (Kouchi et al., 2004) is a user–friendly computer model developed originally by Ministry of Economy, Trade and Industry (METI). The METI–LIS model version 2 is now available in English to download from the online site (Kouchi et al., 2004). This model puts special importance to express downdraft effect, which often affects the atmospheric dispersion from lower emission sources, while it gives solutions of simple Gaussian plume and puff formula (Bosanquet and Pearson, 1936; Sutton, 1947; Turner, 1994; Beychok, 2005) for elevated sources. In addition to a short–term estimation with fixed meteorological conditions, a long–term average estimation can be obtained with the model, when hourly meteorological datasets are prepared by the users.

Equation (1) (Sutton, 1932; Sutton, 1947) is used in the METI–LIS model for the transport of pollutants from a point source, such as a smokestack or exhaust outlet. This section deals with the plume–rise height of exhaust gas, methods of determining dispersion parameters, methods of modeling down–wash effects caused by buildings neighboring the emission source, and the applicable conditions of the dispersion model.

For each source and every hour, the origin of the coordinate system calculation is placed on the ground surface at the base of the stack. The *x*–axis is positive in the downwind direction, the *y*–axis is crosswind (normal) to the *x*–axis, and the *z*–axis extends vertically. The user–defined calculation points are converted to each source's coordinate system for the calculation of concentration at each time period. The conversion method in the *x*–axis and *y*–axis direction is described below. The concentration calculated for each source at each calculation point is summed to obtain the total concentration produced by the combined source emissions for that time period (Bosanquet and Pearson, 1936; Sutton, 1947; Turner, 1994; Beychok, 2005).

$$C(x,y,z) = \frac{QV}{2\pi u_s \sigma_y \sigma_z} \times exp\left[-0.5\left(\frac{y}{\sigma_y}\right)^2\right] \quad (1)$$

where C is the concentration of pollutants (g/m³), at any receptor located, x is the downwind distance from the emission source (m), y is the crosswind distance from the emission plume centerline (m), z is the distance above the ground level (m), Q is the pollutant emission rate (g/s), V is the vertical term [Equation (5)], u_s is the horizontal wind velocity along the plume centerline (m/s), σ_z is the dispersion parameter in vertical direction (m), and σ_y is the dispersion parameter in horizontal direction (m).

Table 2. *Atmospheric stability categories*

Wind speed at ground level U (m/s)	Daytime Solar radiation Q (0.01 kW/m²)				Nighttime (Solar radiation Q < 0)
	60 < Q	30 ~ 59	15 ~ 29	1 ~ 14	
U < 2.0	A	A–B	B	D_d	F
2.0–2.9	A–B	B	C	D_d	E
3.0–3.9	B	B–C	C	D_d	D_n
4.0–5.9	C	C– D_d	D_d	D_d	D_n
6.0 < U	C	D_d	D_d	D_d	D_n

Equations (2), (3), and (4) fit the Pasquill–Gifford curves (Venkatram, 1996) approximately, which are used in the METI–LIS model to calculate the dispersion parameters (σ_y and σ_z). The same equations are also used in the ISC (Industrial Source Complex) model (Bowers and Anderson, 1981; Bowers et al., 1982). ISC is a popular steady–state Gaussian plume model and can be used to assess pollutant concentrations from a variety of sources associated with industrial complexes. The approximation equations are the functions of downwind distance from the source and they calculate the lateral dispersion width, σ_y and the vertical one σ_z of Equation (1), respectively. These dispersion widths are contingent on atmospheric stability (Pasquill, 1961), which is determined by meteorological conditions. Table 2 shows the classification method for the atmospheric stability.

The atmospheric stability category can be selected from Table 2 using the data of wind speed and solar radiation in the area of emission sources (Luna and Church, 1972). While 11 different categories (i.e., A–G) can be accepted as the atmospheric stability, those in the approximation in Table 3 are divided into only six categories (i.e., A–F). Table 3 shows the atmospheric stability category mapping between the observed and the approximation situation with the standard values of the power exponent (p). In Table 3, the atmospheric stabilities, A and A–B are unified to A stability, B and B–C are unified to B stability, C and C–D_d are unified to C stability, D_d and D_n are unified to D stability, F and G are unified to F stability. Among the stability categories, (A, A–B, B, B–C, C, C–D_d, D_d) are the daytime and (D_n, F and G) are the nighttime stability categories. The value of the wind profile exponent (p) is used in Equation (6), which can be obtained from Table 3.

The dispersion parameters σ_y and σ_z are used in Equation (1), which can be obtained from Pasquilll–Gifford curves [Equations (2), (3), and (4)]. The values for dispersion coefficients (a, b, c, and d) (Turner, 1967; Turner, 1994) are available in the online technical manual of the METI–LIS model.

$$\sigma_y = 465.11628(X)\tan(TH) \quad (2)$$

where x is the downwind distance (m).

$$TH = 0.017453293[c - d(\ln(x))] \quad (3)$$

$$\sigma_z = ax^b \quad (4)$$

where a, b, c, and d are the dispersion coefficients.

The vertical term, V in Equation (5) (Sutton, 1932; Sutton, 1947) represents the atmospheric distribution of the Gaussian plume in the vertical direction. This term includes the elevation of calculation point and the effects of the height caused by the emitted plume rise (the effective plume–rise height) (Bosanquet and Pearson, 1936; Sutton, 1947; Turner, 1994; Beychok, 2005). Most of the time, the gases that are emitted from the stacks of a power plant are heated and are warmer than the outdoor air. Emitted gases are less dense than the outside air and therefore they are buoyant. A combination of the gas momentum and buoyancy causes the gases to rise. This is referred to as plume rise and allows air pollutants emitted in this stack gas stream to be lifted higher in the atmosphere.

Table 3. *Relationship between the observed atmospheric stability and the approximation index. This approximation simplified 11 different atmospheric stability categories into only six categories (A to F). The standard values of the power exponent p are used in the METI–LIS model to adjust wind–speed, which is similar to the ISC model (Bowers and Anderson, 1981; Bowers and Anderson, 1982) applies these values to rural areas*

Atmospheric Stability	A	A–B	B	B–C	C	C– D_d	D_d	D_n	E	F	G
Approximation	A		B		C		D		E	F	
Rural exponent (p)	0.07		0.07		0.10		0.15		0.35	0.55	

$$V = exp\left[-0.5\left(\frac{Z_r - h_e}{\sigma_z}\right)^2\right] + exp\left[-0.5\left(\frac{Z_r + h_e}{\sigma_z}\right)^2\right] \quad (5)$$

where z_r is the elevation of calculation points of any receptor, which is located z meters above ground level (m), h_e is the effective plume–rise height (m), which is the sum of the physical stack height and the plume rise.

The wind profile power law (Peterson and Hennessey, 1978; Elliott et al., 1986; Robeson and Shein, 1997; Beychok, 2005) [Equation (6)] is a relationship between the wind speeds at one height, and those in another, which converts the observed wind speed to an equivalent wind speed at the actual height of the emission source. The wind speed used in the dispersion equation is the equivalent wind speed at the stack, or release height. If the height of measurement point of the wind speed is lower than the stack height, the power law equation [Equation (6)] will be applied in MET–LIS model. The power law equation is in the form of:

$$u_2 = u_1\left(\frac{z_2}{z_1}\right)^p \quad (6)$$

where u_2 is the wind speed at the stack outlet height (m/s), u_1 is the wind speed at the measurement height (m/s), z_2 is the stack outlet height (m), and z_1 is the wind–speed measurement height (m).

The wind profile exponent, p in Equation (6) is set according to the atmospheric stability. The values shown in Table 3 can be used as average values.

When the point source is a stack, it acts as a drag to the wind, producing a down–wash known as stack–tip down–wash. When the exit velocity of the exhaust gas from the source is less than 1.5 times the velocity of the horizontal wind along the plume centerline, a correction is applied to the stack height corresponding to stack–tip down–wash by using Equation (7). This method adjusts the physical height of the stack as follows:

$$h'_s = h_s + 2d_s\left(\frac{v_s}{u'_s} - 1.5\right) \quad (7)$$

where h'_s is the modified physical stack height (m), h_s is the physical stack height (m), d_s is the stack diameter (m), v_s is the exit velocity of the exhaust gas (m/s), and u'_s is the velocity of the horizontal wind along the plume centerline (m/s).

The Equation (8) is applicable when $v_s \geq 1.5\ u'_s$.

$$h'_s = h_s \quad (8)$$

This modification is not applied when down–wash effects due to a building are calculated.

The METI–LIS model also emphasized on famous "Briggs equations" to calculate buoyancy–induced plume rise due to hot buoyant plumes of bent–over. In general, plume rise equations for bent–over, hot buoyant plumes are based on observations and data involving plumes from typical combustion sources such as the flue–gas stacks from steam–generating boilers burning fossil fuels in large power plants. Therefore, most of the coal combustion power plants in Japan, the stack exit velocities are about 30 m/s and the exit temperatures are about 90°C (Ito et al., 2006). If the gas emitted by the source is comparatively warmer than the ambient temperature, the CONCAWE equation (Briggs, 1965; Briggs, 1968) is applied as follows:

$$h_e = h_s + \Delta h \quad (9)$$

$$\Delta h = 0.175 Q_H^{1/2} u^{-3/4} \quad (10)$$

where h_e is the effective stack height (m), h_s is the physical stack height (m), Δh is the buoyancy–induced plume rise (m), Q_H is the emitted heat quantity (cal/s), and u is the wind speed at top of stack (m).

$$Q_H = \rho C_p Q(T_s - T_A) \quad (11)$$

where ρ is the gas density at 0°C (1.293×10^3 g/m^3), C_P is the isobaric specific heat (0.24 cal/K/g), Q is the exhaust–gas flowrate (Nm^3/s), T_S is the exhaust–gas temperature (°C), and T_A is the ambient temperature (°C, default is 15°C).

This model also includes building downwash, terrain effects, and line source emissions. The METI–LIS model adopted a downwash scheme based on that of the US Environmental Protection Agency's (EPA) Industrial Source Complex (ISC) model, but the parameters in the dispersion widths describing the down–wash effect were improved by incorporating the results of wind tunnel experiments. Another characteristic point of the METI–LIS model different from the ISC model is that the evaluation time which affects the dispersion width especially in the y (crosswind) direction can be adjusted for a simulation of short time dispersion.

3. Atmospheric Mercury Emissions in Japan

3.1. Mercury emission sources

Of the primary anthropogenic sources of mercury emissions to the environment, the principal sources are those where mercury is emitted mainly as an unintentional byproduct. With the exception of mercury mining itself, the atmospheric mercury emissions arise from the mercury that is present as an impurity in the fuel or used raw materials. The main emissions of mercury as byproducts are from the sectors that involve combustion of coal or oil, production of pig iron and steel, production of non–ferrous metals, and cement production. Stationary combustion of coal, and the combustion of other fossil fuels associated with energy or heat production in major power plants, small industrial or residential heating appliances, and various industrial processes are the largest single source category of anthropogenic mercury emissions to the global atmosphere. Although coal does not contain high concentration of mercury, the amount of mercury emissions to the atmosphere from coal–fired industrial facilities indicates that coal burning is one of the largest anthropogenic sources of unintentional mercury emissions to the atmosphere. Mining and industrial processing of ores, particularly in primary production of iron and steel, and non–ferrous metal production (specially copper, lead and zinc smelting) release mercury to the atmosphere due to fuel combustion, the presence of mercury in ores as impurities, and through accelerating the exposure of rock to natural weathering process. Metal production including mining, the production of mercury itself (a relatively minor source) and the production of gold, where mercury is present in ores and used in some industrial processes, are the minor sources of mercury emissions to the atmospheric environment. Meanwhile, one of the major sources of by–product releases of mercury is associated with cement production, where mercury is released primarily as a result of the combustion of fuels (mainly coal but also a range of wastes) to heat the cement kilns (AMAP and UNEP, 2008).

According to recent research reports (Kida et al., 2007; Moritomi, 2008) and emission data provided by the Japan Ministry of Economy, Trade and Industry (METI, 2001–2004), the total amount of mercury released to the atmosphere from Japan was estimated as 24 – 28 Mg/year, taking into account the releases from specified facilities not reported by PRTR (Pollutant Release

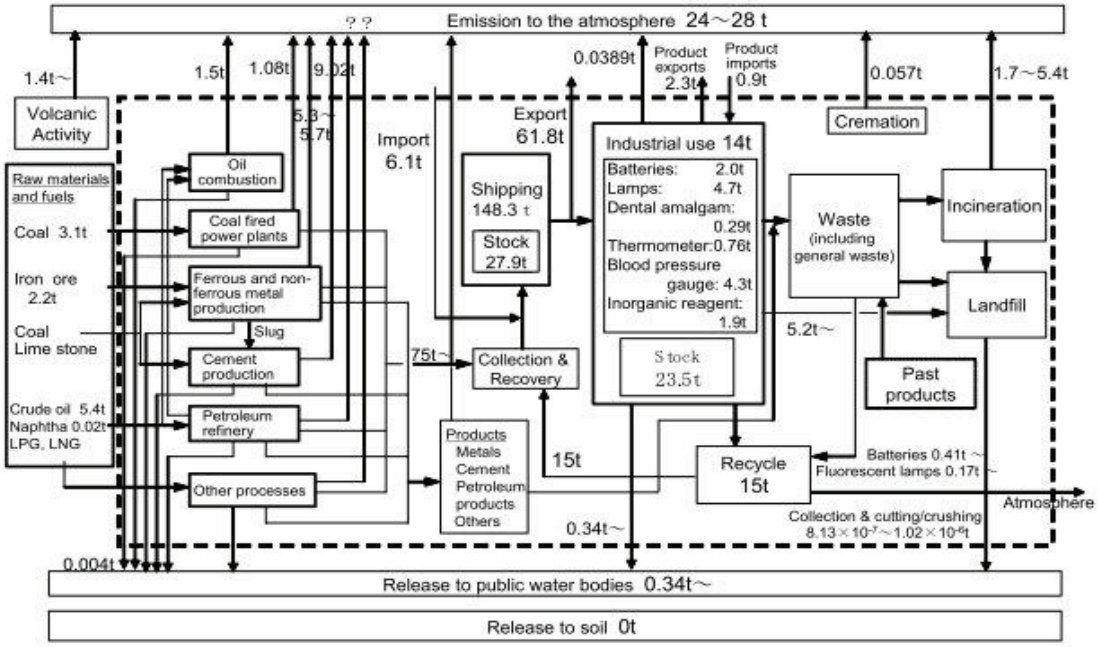

Figure 1. *Material flow diagram of mercury in Japan (Kida et al., 2007; Moritomi, 2008).*

and Transfer Register) (Lerche et al., 2004; Wexler and Harjula, 2005). In the combustion category, coal–fired power plants, industrial oil combustion boilers, incinerators of medical waste, sewage sludge and other wastes are considered to be significant mercury emission sources. Among the heavy industrial production units, primary ferrous and non–ferrous metal production as well as cement production are thought to be major contributors of atmospheric mercury emissions in Japan. Atmospheric mercury emissions in Japan are calculated to be 0.190–0.225 g/year/person (METI, 2001–2004; Kida et al., 2007; Moritomi, 2008). Figure 1 shows the material flow of mercury, which depicts the net mercury load to the atmosphere in Japan from primary anthropogenic sources.

3.2. Estimation of mercury emissions

In this study, the amount of atmospheric mercury emissions from different point sources and area sources in Japan were estimated according to the report on the mercury emissions inventory of Japan (Kida et al., 2007; Moritomi, 2008). Total coal consumption data for 2005 was considered as a calculation basis to estimate the mercury emissions to the atmosphere in Japan, whereas the emission of atmospheric mercury in 2006 is almost similar to that of 2005. The power plant of electrical capacity 1 000 MW consumes 360 Mg/hour coal and the mean concentration of mercury in that coal was 0.045 ppm (Ito et al., 2006). Since about 30% of the total mercury of feed coal goes to the atmosphere from the stack of the coal combustion power industries (Moritomi, 2008) in Japan, the mean emission rate of mercury to the atmosphere was 4.4 µg/KW h. The power plant of electric capacity 1 000 MW emits mercury to the atmosphere is (360 Mg coal/hour) × (0.045 g Hg/Mg coal) × (365×24 hour/year) × 0.3 = 42 600 g/year (42.6 kg/year). The coal combustion rate has been used as a basis to calculate the amount of mercury emissions to the atmosphere from coal combustion power industries in Japan. The emission factors were derived from estimates of the annual emission rate and the total production capacity for each plant in 2006. Total productions are 69.5×10^6 Mg and total emissions of mercury to atmosphere are 5.7 Mg in the industrial sector of iron works, total productions are 79×10^6 Mg and total atmospheric mercury emissions are about 9.8 Mg in the industrial sector of cement plants, total productions are 9 057 Mg and total atmospheric mercury emissions are 0.3 Mg in the industrial sector of chemical plants in 2006 in Japan (JCOAL, 2005; Kida et al., 2007). A simple unitary calculation method was applied to calculate atmospheric mercury emissions from each point source of iron works, cement plants, chemical complexes. The data of yearly production capacity and yearly mercury emissions for each industrial sector were considered as the basis of calculation (METI, 2001–2004; JCOAL, 2005; Kida et al., 2007; Moritomi, 2008). For example, mercury emissions from a specific cement industry = {(total mercury emissions from cement industries in Japan) × (production capacity of that industry)} ÷ total cement production capacity in Japan. The geographical locations of mercury emission sources from coal–fired industrial facilities can be easily pointed out on the map from the website of Japan Coal Energy Center (JCOAL, 2005). In this study, yearly municipal and medical waste has also been considered as a big source of atmospheric mercury emissions in Japan. Total atmospheric mercury emissions from municipal and medical waste are about 1.7–5.4 Mg (Kida et al., 2007) in Japan, that have been distributed to the local atmosphere of each region on the basis of the prefectural population density (METI, 2001–2004; Kida et al., 2007; Moritomi, 2008).

Burning of fossil fuels (primarily coal) is the largest single anthropogenic source of atmospheric mercury emissions, although the emissions from combustion of medical and municipal waste, and industrial waste have a significant release of mercury to the atmosphere in Japan. It is very difficult to find out the actual locations and amounts of mercury emissions in Japan from industrial point sources, since the lack of reliable information on industrial emission assumptions and technologies to calculate mercury emissions as well as confidentiality. In this study, the coal–fired industrial facilities such as power plants, iron works, cement plants, chemical complexes, and oil or gas combustion heavy industries are considered as large emission sources of atmospheric mercury in Japan. Mercury emissions from municipal and medical waste from different areas in Japan have also taken into consideration for the AIST–ADMER model as important area sources of mercury emissions. To calculate the regional atmospheric concentration distribution of mercury, about 28 Mg/year (Kida et al., 2007; Moritomi, 2008) of mercury emissions to the atmosphere have been distributed hypothetically throughout Japan.

4. Study Area

In this study, the industrial source complexes are considered as mercury emission sources, which are located in the central region of the Honshu island of Japan. Total nine regions (Aichi, Mie, Gifu, Fukui, Ishikawa, Niigata, Nagano, Gunma and Toyama) have been selected for the AIST–ADMER model simulation to calculate the regional distribution of mercury concentration. There are different types of heavy and medium–scale industrial facilities located in this area that are in operation.

On the other hand, a small domain (the blue rectangle in Figure 3) in Aichi Prefecture of Japan was selected as a site to calculate the ambient air concentration of mercury in the vicinity of two major industrial sources using the METI–LIS model. Among the two hypothetical power plants inside of the small domain, the plant–1 (the solid black circle in Figures 4, 5, 6) is located in Nagoya area (35° 1' 39.38" N, 136° 51' 54.98" E) and plant–2 (the solid red circle in Figure 4, 5, 6) is located in the Hekinan area (34° 50' 6.80" N, 136° 57' 44.75" E). The two sources are about 20 km apart from each other, and they are located in the coastal area of Japan.

5. Results

5.1. Regional concentration level

The input emission data for the AIST–ADMER model was compiled from the survey results of the Pollutant Release and Transfer Register (PRTR) of 2005. Moreover, mercury emission inventory work, prepared by Kida et al. (2007) and Japan Coal Energy Center (JCOAL, 2005) was applied as an input data for the AIST–ADMER model. Table 4 shows the input parameters for the AIST–ADMER model.

The mercury concentrations in the small domain (blue rectangle) of the Figure 3 served as a background concentration for the METI–LIS model to determine the mercury concentration in the vicinity of two nearly located hypothetical power plants. Since the maximum ambient concentration of mercury inside of the blue rectangle of the Figure 3 was 2.934 ng/m^3 (0.002934 μg/m^3) and minimum concentration was 0.66 ng/m^3 (0.00066 μg/m^3), the background concentration was determined to be (0.66 + 2.934)/2 = 1.797 ng/m^3 = 1.8 ng/m^3 (0.0018 μg/m^3). The values of the background mercury concentrations in different areas are obtained from the Figure 3, which can be compared with the monitoring survey data of hazardous air pollutants in 2006 prepared by the Japan's Ministry of the Environment (MOE, 1997; MOE, 1998–2009). The yearly average mercury concentration data of 12 air quality monitoring stations in Aichi and Mie regions, provided by Japan's Ministry of the Environment (MOE) along with the AIST–ADMER simulation result, are illustrated in Table 5. The magnitudes of mercury concentration calculated by the AIST–ADMER model are slightly overestimated relative to the observed results of 7 monitoring stations, while they are underestimated at 5 monitoring stations, suggesting that the industrial emissions or emission from biomass burning at underestimated monitoring sites were not significantly considered in this study and this needs to be improved. The distribution of the air quality monitoring stations for mercury and the characteristics of each site can refer to the presentation of Suzuki (2008) in Vietnam. Figure 2 shows the geographical locations of 12 monitoring stations. The two hypothetical coal–fired power plants that have been considered for the METI–LIS simulation are located inside of the small domain (blue rectangle) of Figure 2.

Table 5 shows a comparative evaluation of annual mean mercury concentration in 12 monitoring stations with the simulation results of the AIST–ADMER model (Figure 3). Since, most of the monitoring stations are located far away from the industrial point sources in Japan the simulation result of the METI–LIS model cannot be compared with the monitoring data in Table 5.

Table 4. *Input parameters for the AIST–ADMER model*

Start of calculation	January 2006
End of calculation	December 2006
Washout ratio	1
Half life (days)	365
Emission pattern	Yearly average emission

Figure 2. *Map of observation sites in Aichi and Mie regions and two industrial point sources in Aichi region. The red solid circle on the map shows the location of 12 monitoring stations. The blue solid circle (34° 50' 6.80" N, 136° 57' 44.75" E) inside the rectangle shows a point source location of mercury emission and the black solid circle (35° 1' 39.38" N, 136° 51' 54.98" E) shows another point source of mercury emission.*

Table 5. *Monitoring data of mercury concentration in 2006 provided by Ministry of Environment in Japan (MOE, 1997–2004; MOE, 1997)*

Name of the region	Number	Name of the monitoring station	Geographical location	One–year mean concentration (ng/m^3)		Observed concentration range (ng/m^3)
				The observed mean value	The AIST–ADMER simulation mean	
Aichi	1	Nagoya City (Chikusaku)	E 136° 56' 52" N 35° 9' 57"	0.73	1.63	0.20–1.7
	2	Nagoya City (Nakagawaku)	E 136° 51' 17" N 35° 8' 30"	2.1	2.32	1.8–2.4
	3	Toyohashi City (Oosaki)	E 137° 20' 36" N 34° 42' 58"	2.5	1.92	1.6–4.6
	4	Toyohashi City (Futagawa)	E 137° 26' 20" N 34° 43' 32"	1.8	0.21	0.98–2.5
	5	Okazaki City	E 137° 11' 15" N 34° 55' 51"	2.5	2.08	0.51–3.9
	6	Toyota City (Central)	E 137° 4' 44" N 35° 1' 59"	2.0	2.24	1.4–2.8
	7	Toyota City (North)	E 137° 3' 29" N 35° 1' 0"	1.7	2.06	0.81–2.6
	8	Komaki City	E 136° 55' 6" N 35° 17' 35"	1.8	1.43	1.0–2.3
Mie	9	Yokkaichi (North)	E 136° 38' 29" N 35° 0' 30"	2.4	2.65	1.9–3.2
	10	Yokkaichi City (Center)	E 136° 37' 28" N 34° 57' 59"	1.9	2.48	1.2–2.9
	11	Matsusaka City	E 136° 32' 25" N 34° 33' 43"	2.2	1.76	1.6–2.7
	12	Kuwana City	E 136° 41' 6" N 35° 3' 43"	2.1	2.34	1.5–2.5

Figure 3 shows the annual mean distribution of atmospheric mercury concentrations in the central Honshu island of Japan calculated by the AIST–ADMER model. The results established that the atmospheric mercury concentration was relatively high in major urban areas such as Nagoya and Yokkaichi, as emissions from industrial facilities, medical and municipal wastes tend to be concentrated in these densely populated areas. The annual mean concentration of atmospheric mercury was calculated to be less than 2.934 ng/m^3 (0.002934 μg/m^3) in major industrial areas, greater than 0.0263 ng/m^3 (0.000263 μg/m^3) in nonindustrial areas, which was calculated by the AIST–ADMER model in this study. The AIST–ADMER is a regional dispersion model, which can calculate wide–area chemical transportation considering several point sources, line sources and area sources in Japan. The simulation result shows that the mercury concentration calculated by the AIST–ADMER mode was diluted and always less than that of the result of the METI–LIS model, because of its regional scale chemical transportation scheme to calculate the atmospheric concentration of chemical substances. In some cases, concentrations were calculated to be 5 ng/m^3 (0.005 μg/m^3) – 10 ng/m^3 (0.01 μg/m^3) in the vicinity of major industrial point sources simulated by the METI–LIS model. Most of the cases, the concentration of mercury calculated by the METI–LIS model is slightly higher than observations, because METI–LIS generally calculate the pollutant concentration in the vicinity of industrial point sources for a small domain.

5.2. Concentration level near industrial sources

The ambient concentration of mercury in the vicinity of two major industrial sources was predicted by the METI–LIS model. Mercury releases to the atmosphere from these two hypothetical coal–fired power plants were calculated on the basis of mercury emission factor (Kida et al., 2007; Moritomi, 2008; METI, 2011). The selected site for the METI–LIS simulation had a calculation domain of 25 km × 25 km with a grid spacing of 200 m, which included the two largest point sources corresponding to the 3 km × 3 km calculation grids of the AIST–ADMER model. There are 5 units in power plant–1 and 6 units in power plant–2, whereas the capacity of each unit is 1 000 MW. Power plant–1 with 5 000 MW electric capacity is emitting 213 kg mercury/year into the local atmosphere. Similarly, the capacity of power plant–2 is 6 000 MW and it is emitting 256 kg/year mercury to the atmosphere. The amount of mercury emissions to the atmosphere from plant–1 and plant–2 have been considered as input data for the METI–LIS model. These two large coal combustion facilities in this area are the significant sources of mercury emissions to the atmosphere in Japan, emitting about 1.7% of mercury into the air every year (Kida et al., 2007). To evaluate the effect of mercury emissions from the two hypothetical coal–fired power plants in the local atmosphere, stack gas dispersion was calculated by the METI–LIS model. Table 6 shows the operational conditions of these two power plants. The specifications of each power plant provided in Table 6 are also very important input data for the METI–LIS model. Since there was no emission of Hg(p) from the coal combustion power plants in Japan, the effect of gravitational sedimentation, and the amounts of dry and wet depositions were not considered in this study. Necessary assumptions for mercury emissions and the specifications of each point source were determined from the report of Japan Coal Energy Center for 2005 (JCOAL, 2005) and the research work by Ito et al. (2006). It was assumed that the emission factors were constant for 365 days, 24 hours a day. AMeDAS (JMA, 2006) data were used as meteorological input data for the METI–LIS model. Source contributions from other sources (e.g., mobile sources or point sources located outside of the calculation domain) were not included in the input data for the METI–LIS model. Source contributions from other sources were calculated with the AIST–ADMER model and were superposed onto the simulation results of the METI–LIS model as the background concentration data of mercury in the atmosphere.

Figures 4 – 6 show the distribution of mercury concentrations in the vicinity of the two hypothetical power plants in winter, summer, and one–year average (2006), which were calculated using the METI–LIS model. The mark of the solid black circle (latitude 35° 1' 39.38" N, and longitude 136° 51' 54.98" E) with

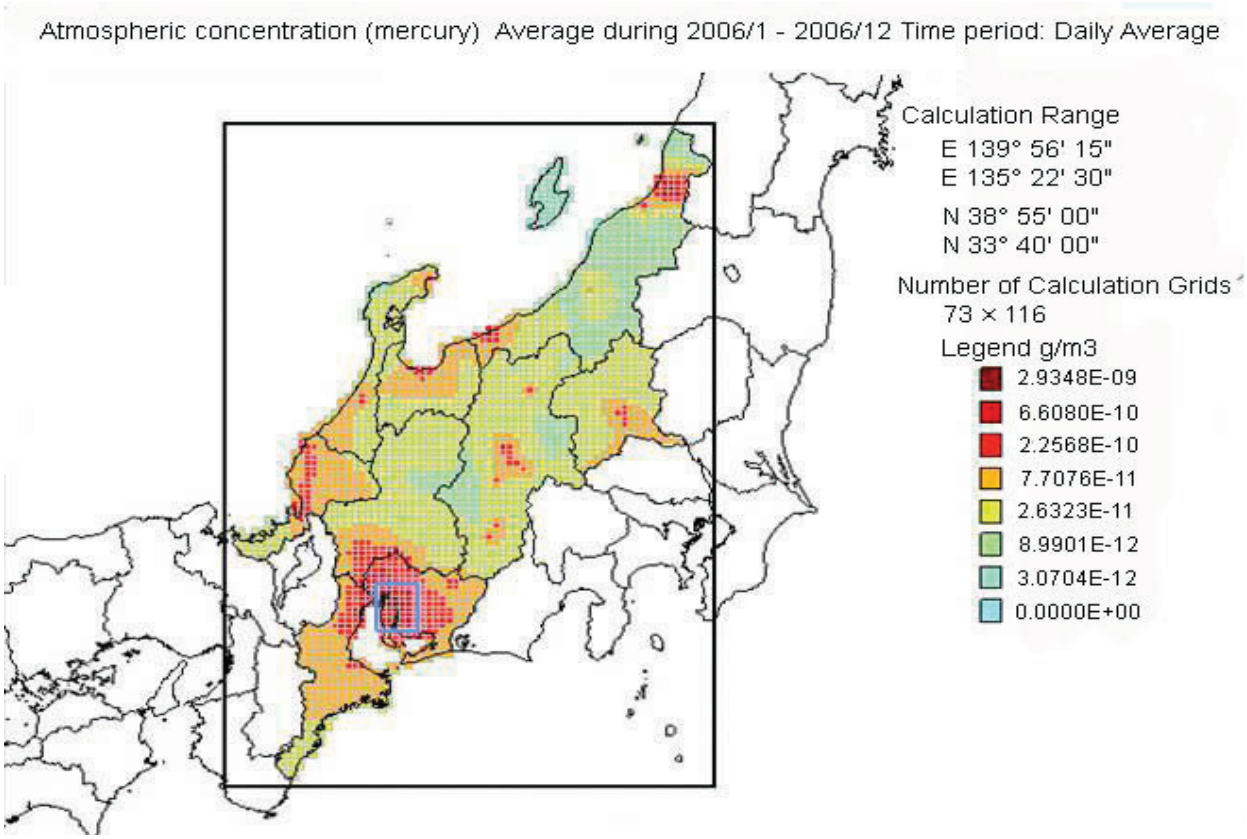

Figure 3. *The annual mean concentration distribution of atmospheric mercury calculated with the AIST–ADMER model in 2006. Nine areas (Aichi, Mie, Gifu, Fukui, Ishikawa, Niigata, Nagano, Gunma, and Toyama) are designated on the map. The blue rectangle was used for the METI–LIS model as a local domain, which served as background concentration data in the calculation areas for the METI–LIS model.*

Table 6. *Specifications of hypothetical coal–fired power stations (Ito et al., 2006). The plant–1 has five units and the Plant–2 has six units. The production capacity of each unit is 1 000 MW*

Operation condition	**Plant–1 (35° 1' 39.38" N, 136° 51' 54.98" E)**	**Plant–2 (34° 50' 6.80" N, 136° 57' 44.75" E)**
Output	1 000 MW × 5	1 000 MW × 6
Coal consumption	360 Mg/h × 5	360 Mg/h × 6
Height of stack	200 m	200 m
Stack gas temperature	90 °C	90 °C
Discharge velocity	30 m/s	30 m/s
Volume flow rate (wet)	3 400 000 Nm^3/h × 5	3 400 000 Nm^3/h × 6
Availability factor (annual)	100% (365 days/year)	100% (365 days/year)

0.213 Mg/year mercury emissions and the mark of the solid red circle (latitude 34° 50' 6.80" N, and longitude 136° 57' 44.75" E) with 0.256 Mg/year emission represents the industrial source location on the Figures 4–6. In winter, the mean distribution of mercury concentrations were calculated to range between 0.0068 and 0.0118 μg/m^3 near industrial sources. On the other hand, mean distribution of mercury concentrations ranged between 0.0028–0.0068 μg/m^3 in the same locations in summer, that were much lower than those of winter due to the effect of the boundary–layer meteorological conditions in coastal areas of Japan. In coastal regions, sea and land breezes can be important factors affecting the wind speed and direction. During the summer, the temperature difference between the sea surface and the land surface is much greater than that during the winter (Steve, 1995; JetStream, 2008). During the summer, the effect of the sea and land breeze causes a strong wind flow in the coastal ground level, driving the pollutants far away from their sources. As a result, the concentration of mercury is relatively low in the summer near the industrial point sources. Besides, the thermal circulations of wind in winter are very low, which causes a higher concentration of mercury in the vicinity of the industrial sources.

The annual concentration distribution of mercury calculated by the METI–LIS model establishes that some people living in certain areas near industrial point sources were exposed to a little higher concentration of mercury compared to general population but the levels of mercury meets the air quality standard of Japan's Ministry of Environment. Figure 7 shows the annual wind rose plot, which gives a succinct view of how wind speed and direction are typically distributed at the location between the two point sources in 2006. The annual mean concentration was estimated not to exceed 0.04 μg/m^3 near an industrial source (Kida, 2005), whereas a similar concentration level was found in different seasons.

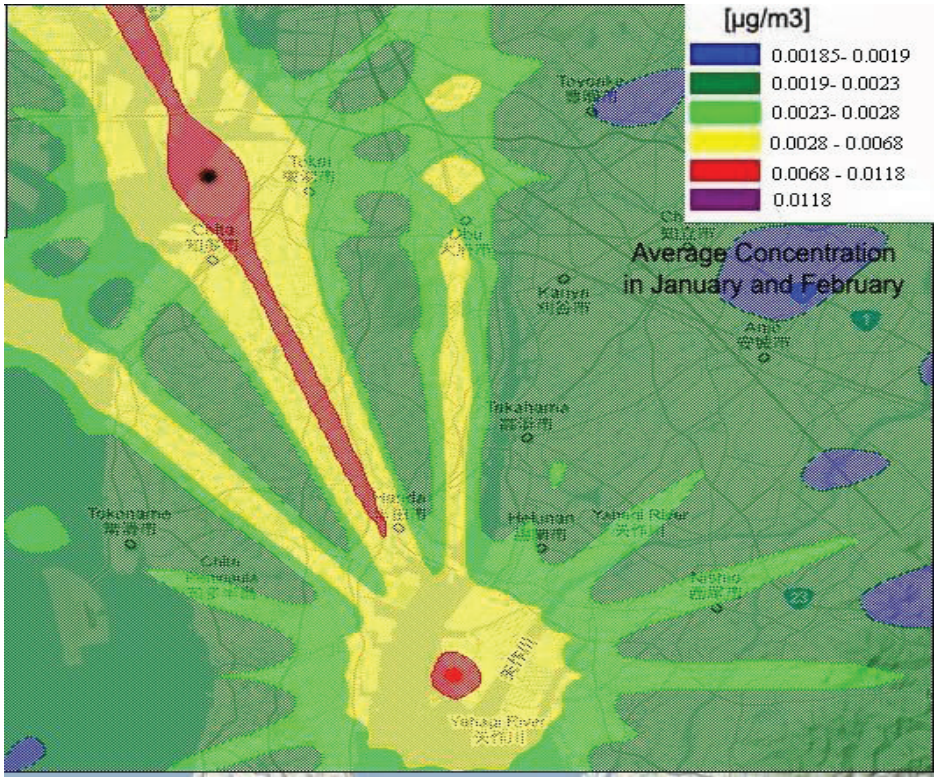

Figure 4. *The average concentration distribution of mercury vicinity of two large point sources, calculated with the METI–LIS model in January and February, 2006. The mark of the solid black circle and red circle represents the industrial source locations in the figure.*

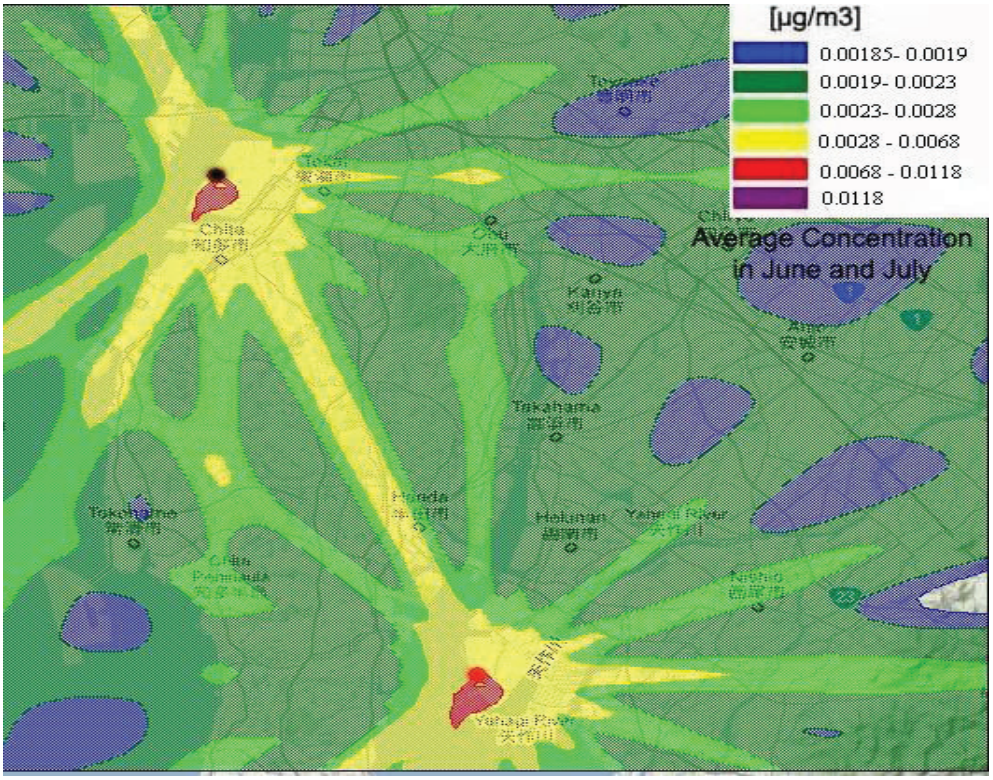

Figure 5. *The average concentration distribution of mercury vicinity of two large point sources, calculated with the METI–LIS model in June and July, 2006. The mark of the solid black circle and red circle represents the industrial source locations in the figure.*

6. Conclusions

In Japan, mercury was categorized as a hazardous air pollutant (HAP) in 1996 due to its high carcinogenic potential. The national government initiated a number of programs to establish emission evaluations and assessments of ambient concentrations. Reduction efforts of mercury emissions has been started on a community basis under public (local governments) and private partnership of industries in Japan which ongoing since 2005 under the support of a voluntary reduction program for emissions. In 2003, Japan initiated the PRTR system, such that the emission data regarding mercury from various sources could be made available in near

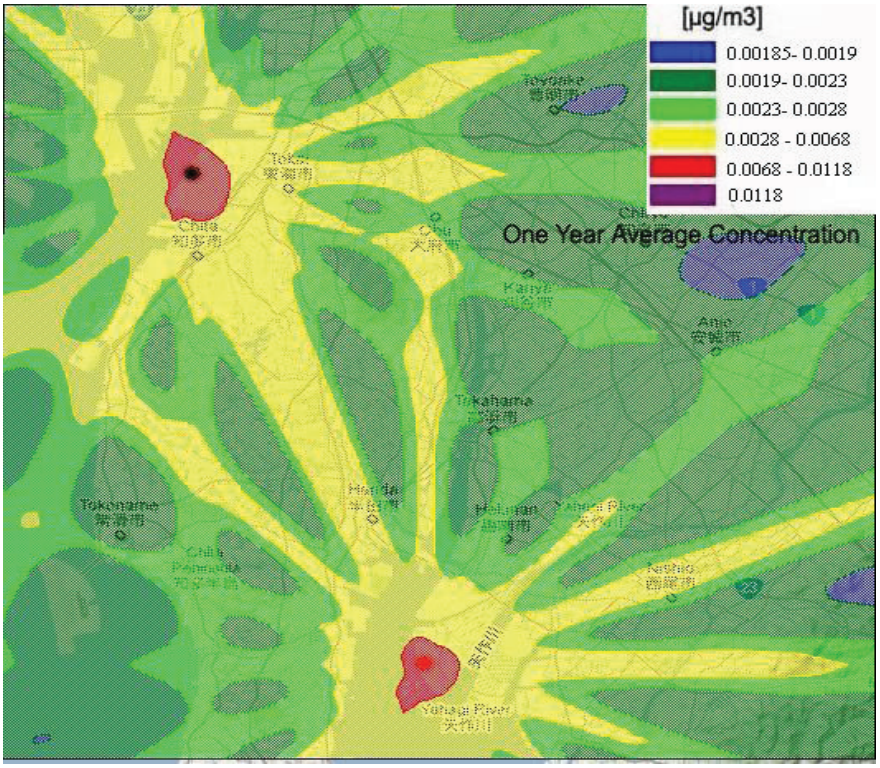

Figure 6. *One–year average concentration distribution of mercury vicinity of two large point sources, calculated with the METI–LIS model in 2006. The mark of the solid black circle and red circle represents the industrial source locations in the figure.*

future. However, the precise amounts remain somewhat uncertain due to ambiguities in the estimation methodologies employed to evaluate mobile sources. The main sources of mercury emissions to atmosphere in Japan are coal–fired cement plants, accounting for over 30% of the total emissions in the year 2006. On the other hand, industrial emissions from primary ferrous metal production and coal–fired power plants had a significant contribution of atmospheric mercury emissions in Japan in 2006. The assessment of mercury concentrations in the local atmosphere in Japan was performed using two different atmospheric dispersion models, i.e., the AIST–ADMER and the METI–LIS. The results of the present study indicated that the annual mean ambient concentrations of mercury in residential areas generally amounted to be less than 0.22 ng/m^3 (0.00022 µg/m^3), but there are no sites that exceed 0.04 µg/m^3 near industrial point sources. Though it is unrealistic to expect the Gaussian models to predict the real situation of mercury concentration in the local atmosphere, the major purposes of the present assessment was to conduct a methodology of comprehensive analysis of exposure and atmospheric distribution of mercury concentration, and thereby to develop a detailed picture of current air quality assessment of the different industrial areas of Japan.

In the present study, small–scale and medium–scale dispersion models for the different regions in the coastal area of Japan were used. The results show a reasonable agreement with the monitoring data with respect to predicting local atmospheric concentrations of mercury. Although there are many models have been dedicated to the modeling of mercury transport in the atmosphere of global and regional scales in the last decades, not many studies have been conducted to investigate the transport pathway of mercury from point sources. Readily available tools and data combined with these two dispersion models provide an accurate representation of the air quality at a lower cost than the existing air quality monitoring systems in Japan. The dispersion models that applied to the regions of Japan in this study, remove the assumptions for uniform air quality within the vicinity of a monitoring station. The preliminary results of the present study are encouraging as air dispersion models providing emission data for assessing air quality in the different regions in Japan.

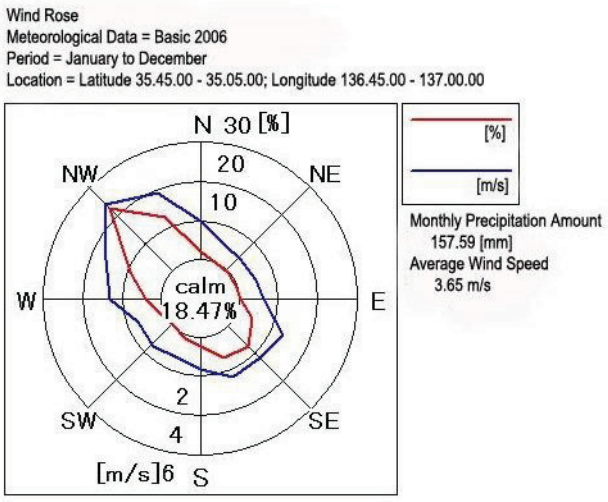

Figure 7. *The annual wind rose of the point–source area in 2006. Blue and red lines indicate the annual mean wind speed (m/s) and the frequency (%) of each direction, respectively.*

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Scientific Research

An Environmental Risk Evaluation Method Employing Atmospheric Dispersion Models and GIS

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ABSTRACT

This study aims to develop a method for evaluating the environmental risk of harmful chemical substances released from specific sources, using two atmospheric dispersion models and GIS (Geographic Information Systems). In the first stage of evaluation, ADMER was used to conduct a wide-area evaluation which covered the entire area of the evaluation target region. In the second stage, METI-LIS was used to conduct a detailed limited-area evaluation which targeted the vicinity of sources. In this study, incinerators were selected as sources and dioxins were selected as harmful chemical substances. The area selected for evaluation was the Tokyo Metropolis in Japan, and the evaluation method proposed in this study was used to evaluate environmental risk. Through the use of atmospheric dispersion models and GIS, the behavior of dioxins emitted into the atmosphere from incinerators was estimated. By superimposing atmospheric levels and population data, the amounts of dioxins that humans exposed to were found. Additionally, by superimposing deposition amounts and land use data, the amounts of dioxins accumulated in each land environment were found. Conducting these steps enabled the impact of dioxins on humans and the environment to be grasped quantitatively and visually, and the risk that dioxins emitted from incinerators pose to the environment to be evaluated.

Keywords: Environmental Risk; Atmospheric Dispersion Models; GIS; Dioxins; Incinerator

1. Introduction

Due to advances in the world's chemical technology, chemical substances which are beneficial to humans have been researched, developed, and manufactured. However, there are also many chemical substances that are harmful, and chemical substances which are produced artificially in the process of manufacturing things and chemical substances generated unintentionally when things incinerated are having a harmful impact on humans and the ecosystem. Presently, harmful chemical substances released from specific sources—in particular, chemical substances which are harmful to humans, such as carcinogenic substances—exist in the environment in places such as the atmosphere, soil and rivers. Harmful chemical substances such as these which exist in the environment are taken into the human body through routes such as breathing, eating and drinking, and skin contact, and there is a risk that they may affect health. Further, harmful chemical substances accumulate in the ecosystems of plants, animals, fish and so on, through environmental mediums such as the atmosphere and soil [1]. In the environmental risk field, importance is being placed on effectively and economically preventing or reducing the burden on the environment caused by such harmful chemical substances [2,3].

Based on the above background, this study aims to develop a method for evaluating the environmental risk of harmful chemical substances released from specific sources. Further, dioxins, which are representative examples of harmful chemical substances that became an issue of public concern in Japan at the end of the twentieth century, are taken up as an environmental risk for discussion; and focusing on incinerators, which are a major source of dioxins in the atmosphere of the Tokyo Me-

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tropolis, Japan [4,5], the environmental risk is evaluated using this evaluation method. Japan's measures against dioxins are promoted based on the basic guidelines for promotion of measures against dioxins and the law concerning special measures against dioxins, which were formulated in 1999 [6].

2. Related Works

In research related to dioxins [7-12] used atmospheric dispersion models to conduct simulations of the behavior of dioxins with incinerators as a source, similarly to this study. [1,13-15] conducted simulations of the behavior of dioxins using atmospheric dispersion models. [16-20] modeled the behavior of dioxins in the atmosphere. Further, [21,22] used dispersion models to conduct simulations of the behavior of dioxins in bodies of water such as sea waters and canals, and [23] modeled the behavior of dioxins in bodies of water.

[7,8] in the above-mentioned related research conducted a comparison of estimated dioxin levels, which were the results of simulation of behavior of dioxins conducted using ADMER, an atmospheric dispersion model, and actual measured levels of dioxins in the environment. They demonstrated the satisfactory reproducibility of the former, and thereby confirmed the usefulness of this atmospheric dispersion model. Therefore, this study also employs ADMER, when evaluating the environmental risk over a wide area in the first stage of evaluation. Further, [13,18] demonstrated the value of using an atmospheric dispersion model and GIS (Geographic Information Systems) in combination in order to estimate the behavior of dioxins in the atmosphere.

Compared to the preceding studies mentioned above, this study demonstrates its uniqueness in that it develops a method that can quantitatively evaluate impact on humans and the environment based on the behavior of dioxins in the environment. Further, the method can analyze dioxin level distribution using two spatial scales—wide area, and narrow area with high dioxin levels—and evaluate the environmental risk in detail. Further, taking into account the results of preceding studies, in the evaluation method, GIS is used in addition to an atmospheric dispersion model, and through this, dioxin behavior can be spatially analyzed. Therefore, the impact on people and the environment of dioxins released from sources can be quantitatively evaluated. Specifically, the environmental risk is evaluated by using atmospheric dispersion models to calculate atmospheric levels and deposition amounts of dioxins emitted from specific incinerators; using GIS to conduct overlay with population and land environment spatial distribution; and estimating amounts people are exposed to and amounts accumulated in the environment.

3. Evaluation Method

3.1. Evaluation Outline and Method

The evaluation outline and method of this study are as shown in **Figure 1**. Below, each stage is described in detail.

1) Incinerators were selected as the source to be evaluated, dioxins were selected as the harmful chemical substance which was the index of evaluation, and the environmental risk was evaluated. In Japan, according to the Law Concerning Special Measures against Dioxins, from the year 2000 onward it has been compulsory for places of business to carry out independent measurements of dioxins once a year or more; therefore, this data, which each prefecture releases, was processed into data necessary in calculations for the atmospheric dispersion model, and the source data was created.

2) Based on the created source data, data necessary to the dispersion calculations of the atmospheric dispersion models was entered, and the atmospheric levels and deposition amounts of dioxins released from the sources were calculated. In the evaluation method of this study, two atmospheric dispersion models, ADMER and METI-LIS, which will be described in detail in the next section, were used. In the evaluation of the entire area of the region for evaluation, in which ADMER was used, the entire area of the Tokyo Metropolis was evaluated. In the evaluation of the vicinity of the sources, in which METI-LIS was used, regions where the contamination risk from dioxins was high based on evaluation results of the previous stage were selected, and the areas surrounding general waste incinerators were the main targets of evaluation.

3) Results of analysis obtained using the atmospheric dispersion models were displayed on digital maps using GIS, and spatial analysis was conducted. Superimposi-

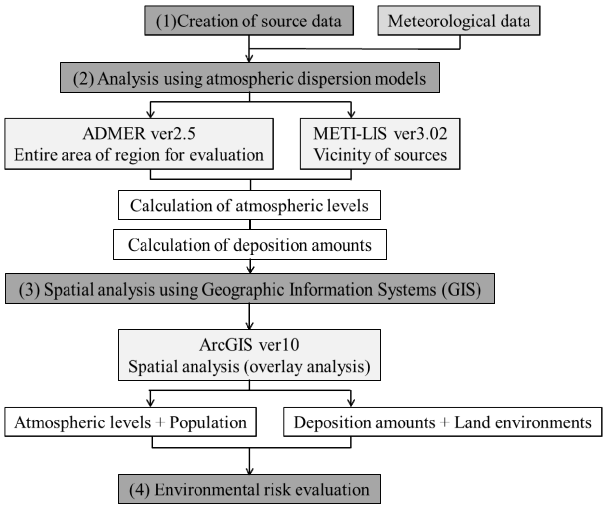


Figure 1. Evaluation method procedure.

tions of data of atmospheric levels and deposition amounts, estimated using the atmospheric dispersion models, and population and land use data were conducted. This enabled human dioxin exposure levels and land environment accumulated dioxin amounts to be estimated.

4) Based on the above-mentioned superimposed results, statistical processing was conducted, and the total population figure per unit of dioxin atmospheric level and the amount of dioxins accumulated in each land environment were aggregated. Comparing the atmospheric levels and the total population figures enabled the exposure level to humans from the atmospheric levels to be estimated and the total population in high dioxin level areas to be quantitatively calculated. Further, based on deposition amounts and land use, it was possible to calculate the amount of dioxins accumulated in each land environment, and to identify land environments with high levels of contamination. In addition, it was possible to fully understand whether contamination levels for rice fields, farming land, watercourses and the like, which are considered to be routes of ingestion to the human body, met environmental standards or not. Through conducting these steps, environmental risk was evaluated from the perspective of impact on people and the environment.

3.2. Outline of Atmospheric Dispersion Models and GIS

In this study, evaluation of the environmental risk was divided into two stages; therefore, two types of atmospheric dispersion model were used. In the first stage, which was a wide-area evaluation which targeted the entire area of the region for evaluation, the "National Institute of Advanced Industrial Science and Technology Atmospheric Dispersion Model for Exposure and Risk Assessment" [24-26] was used. This is an atmospheric dispersion model suited to estimating the atmospheric levels of a chemical substance based on the emissions of sources and meteorological conditions, and predicting long-term level distribution over a wide area. In the second stage, which was a detailed evaluation focusing on the vicinity of some sources, the "Ministry of Economy, Trade and Industry-Low rise Industrial Source Dispersion Model" [27-29] was used. This model predicts the level of chemical substances in the vicinity of sources, and can take into account the downwash which occurs when there is influence from turbulence of air current due to buildings in the vicinity of sources. That is, when this model is used, when data concerning the height of buildings in the area surrounding sources is input, the influence of buildings on dispersion can be taken into account, and detailed level distribution analysis can be performed for limited areas. Further, as the GIS, ESRI Inc.'s ArcGIS ver.10 was used. In assessing the environmental risk, ArcGIS ver.10 was used to conduct overlay analysis involving the analysis results obtained from the two types of atmospheric dispersion model, and the population and land use data, and to conduct statistical processing.

4. Data Collection and Processing

In this study, the data shown in **Table 1** was used. Source data and meteorological data were used for the atmospheric dispersion models, and population data, digital

Table 1. Usage data.

Type	Name	Source
Source data	Results of independent measurements based on the law concerning special measures against dioxins (data for the 23 wards of Tokyo and the Tama region for 2000 to 2008)	Bureau of Environment, Tokyo Metropolitan Government
Meteorological data	AMeDAS data for ADMER for 2001 to 2009	National Institute of Advanced Industrial Science and Technology, Japan
	AMeDAS annual reports, AMeDAS statistics (1996 to 2004)	Japan Meteorological Business Support Center
Population data	National census 500 m mesh total population figures (2000, 2005, 2010)	Statistics Bureau, Ministry of Internal Affairs and Communications
	National census subregion total population figures (2000)	
Digital map data	Administrative district (area) data (2012)	Policy Bureau, Ministry of Land, Infrastructure, Transport and Tourism
	Land use subdivision mesh data (1997, 2006)	
	Land use tertiary mesh data (1997)	
	Elevation/gradient fourth-level mesh data (2011)	
	Map basis information (Scale: 1/2500)	
Data of actual measurements	Tokyo Metropolitan dioxin emission estimation results and dioxin survey results (2000 to 2008)	Bureau of Environment, Tokyo Metropolitan Government

map data and data of actual measurements were each processed into GIS data and used in the spatial analysis. Further, both the two types of atmospheric dispersion model used in this study require input of the amount of emissions per hour (mg/h); therefore, results of independent measurements in the Tokyo Metropolis based on the Law Concerning Special Measures against Dioxins were referred to, and a method of calculating the emissions from an incinerator based on the dioxins in gas emissions (ngTEQ/m^{3}N) and incineration capacity (kg/h) is shown below. As the amount of gas emissions per unit of amount incinerated, which is the amount of gas emissions generated per ton of waste, 5000 (m^{3}N/ton), set by the Committee to Investigate Measures for the Reduction of Dioxins Related to Waste Treatment (1997) [30] and the Ministry of the Environment (2001) [31], was used.

Level in gas emissions C_D

$$C_D = \frac{C_1 + C_2 + \cdots + C_n}{n} \times 10^{-9} \quad (1)$$

Amount incinerated I_a

$$I_a = \frac{I_c \times \alpha \times \beta}{1000} \quad (2)$$

Amount emitted E_i

$$E_i = C_D \times I_a \times E_g \quad (3)$$

C_D: Level in gas emissions (gTEQ/m^{3}N)
N: Number of times measured
C_n: Level in gas emissions the nth time (ngTEQ/m^{3}N)
I_a: Amount incinerated (g/year)
I_c: Incineration capacity per hour (kg/h)
α: Hours operated per day (hours/day)
β: Days operated per year (days/year)
E_i: Amount emitted (gTEQ/year)
E_g: Amount of gas emissions per unit of amount incinerated (m^{3}N/ton)

Among the incinerators which were evaluated, many report dioxin levels as being unmeasured. General waste incinerators which were evaluated by ADMER are large-scale incinerators managed by local governments; therefore, none of these were among those that had not carried out measurements. However, all the incinerators required by the Law Concerning Special Measures against Dioxins to carry out independent measurements of dioxins once a year or more were evaluation targets of METI-LIS; therefore, besides general waste incinerators, medium-sized and small incinerators were also evaluation targets of METI-LIS. In particular, many small incinerators do not carry out measurements of dioxins, so it was necessary to estimate the emissions from these incinerators. Accordingly, incinerators which measure dioxins each year were classified in detail according to incineration capacity (the scale of the facilities of the incinerator), and as shown in **Table 2**, the average annual amount of emissions was estimated for each one. Then, after taking into consideration the number of days of operation per year and the number of hours of operation per day, based on **Table 2**, the amount of emissions per hour for incinerators that had not measured dioxins was calculated.

5. Evaluation of Entire Area of Region for Evaluation

5.1. Evaluation Targets

The evaluation targets in this section were all 41 large-scale general waste incinerators which were total-continuous-type incinerators and were set up by local governments. The first reason for evaluating general waste incinerators was that they are large-scale incinerators which continue operating 24 hours a day throughout the year, and they have a long burning time and a high incineration capacity; therefore there is a high probability that they will have a large impact on the environment. The second reason was the height of their stacks. ADMER is basically suited to analyzing level distribution over a wide area. In wide-area level distribution analysis, sources for which there is a possibility that chemical substances will be dispersed further by the behavior of the atmosphere must be selected; therefore, incinerators with high stacks were focused on. Small incinerators with low stacks are influenced by high buildings in their surroundings, and disturbance of the dispersion of dioxins in the atmosphere occurs. However, the stacks of general waste incinerators are 40 m high or more; therefore, it is not necessary to take into account the influence of disturbances of the air stream which occur due to the surrounding buildings. For the evaluation target range, GIS was used, and a range of calculation of 90 km east-west and 50 km north-south was set, such that the entire area of the Tokyo Metropolis except for the islands was included, and level distribution was output using 500 m mesh units. As the period evaluated, the nine years from 2000 to 2008, for which source data was available, was selected.

Table 2. Average annual emissions of incinerators, classified by scale of facilities (gTEQ/year).

Scale of facilities	Year 2000	Year 2001
Fire bed area of more than 0.5 m^2 and less than 50 kg/h	0.0048	0.0021
50 kg/h—less than 100 kg/h	0.0110	0.0089
100 kg/h—less than 200 kg/h	0.0180	0.0151
200 kg/h—less than 4000 kg/h	-	0.0373

5.2. Calculation Conditions

1) Source data and chemical substance parameters

For ADMER, necessary source data is source location (longitude and latitude), annual emissions and emissions elevation; necessary chemical substance parameters are decomposition coefficient, washout ratio, background levels and dry deposition velocity. Annual emissions were calculated using the calculation method in the previous section. In the ADMER analysis, only incinerators which operated continuously throughout the year were evaluated for each year. Incinerators whose operation was suspended, incinerators which were under construction or newly built, and incinerators which ceased operation during a year were excluded from that year's analysis. Further, the value of the background level, which was necessary as a chemical substance parameter, was set as zero. Usually, concerning the background level, the atmospheric level of a region which is considered to receive hardly any impact from emission sources is used. However, in this study, the environmental risk from dioxins emitted from incinerators was evaluated; therefore, this was done to exclude the influence of sources other than incinerators, such as car exhaust gas and crematoriums.

2) Details of source data

In order to fully understand the dioxin emissions from each incinerator during the evaluation period, annual emissions for each year were aggregated. These results are shown in **Figure 2**. As **Figure 2** shows, incinerators whose total dioxin emissions for the nine year period were considerably higher than those of other incinerators were Setagaya Waste Incineration Plant, at 7.31 gTEQ/year; Hino City Clean Center, at 6.96 gTEQ/year; Ota Waste Incineration Plant, at 5.82 gTEQ/year; and Shinagawa Waste Incineration Plant (formerly Oi Waste Incineration Plant), at 3.93 gTEQ/year.

5.3. Results and Consideration

1) Results of analysis by the atmospheric dispersion model (ADMER) and consideration

Figure 3 shows the results of analysis of atmospheric levels and deposition amounts during the evaluation period that were calculated using ADMER. It can be seen that atmospheric levels and deposition amounts were high particularly in the years 2000 and 2001, and decreased greatly from the year 2002 onwards. It can also be seen that different regions had high atmospheric levels and deposition amounts each year. It is necessary to understand the contamination levels in more detail in Hino City, Ota Ward and Setagaya Ward, which were high-level regions in 2000 and 2001; however, the city and the wards were not identified as high-level regions in the year 2002 and beyond; therefore, it can be said that the amount of dioxin emissions from their general waste incinerators has been reduced. Meanwhile, from the year 2002 and onwards, Akiruno City is identified as a high-level region more frequently than other regions, and there is variation in the dioxin emissions from its general waste incinerators each year; therefore, it can be said that compared to general waste incinerators in other regions, the emission levels of those in Akiruno City have not been improved. Comparing the 23 wards (the eastern part) with the Tama region (the western part) in the Tokyo Metropolis, from the year 2002 onwards, high-level regions were not identified in the wards of Tokyo; therefore, it can be said that the beneficial effects of the measures against dioxins implemented by the Clean Association of TOKYO23 were demonstrated.

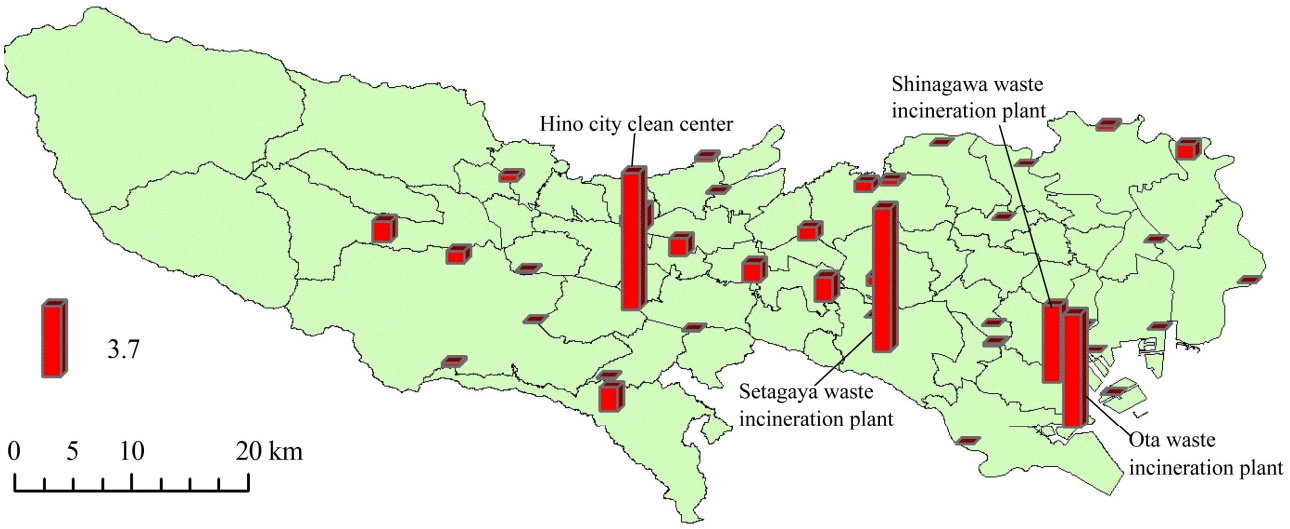
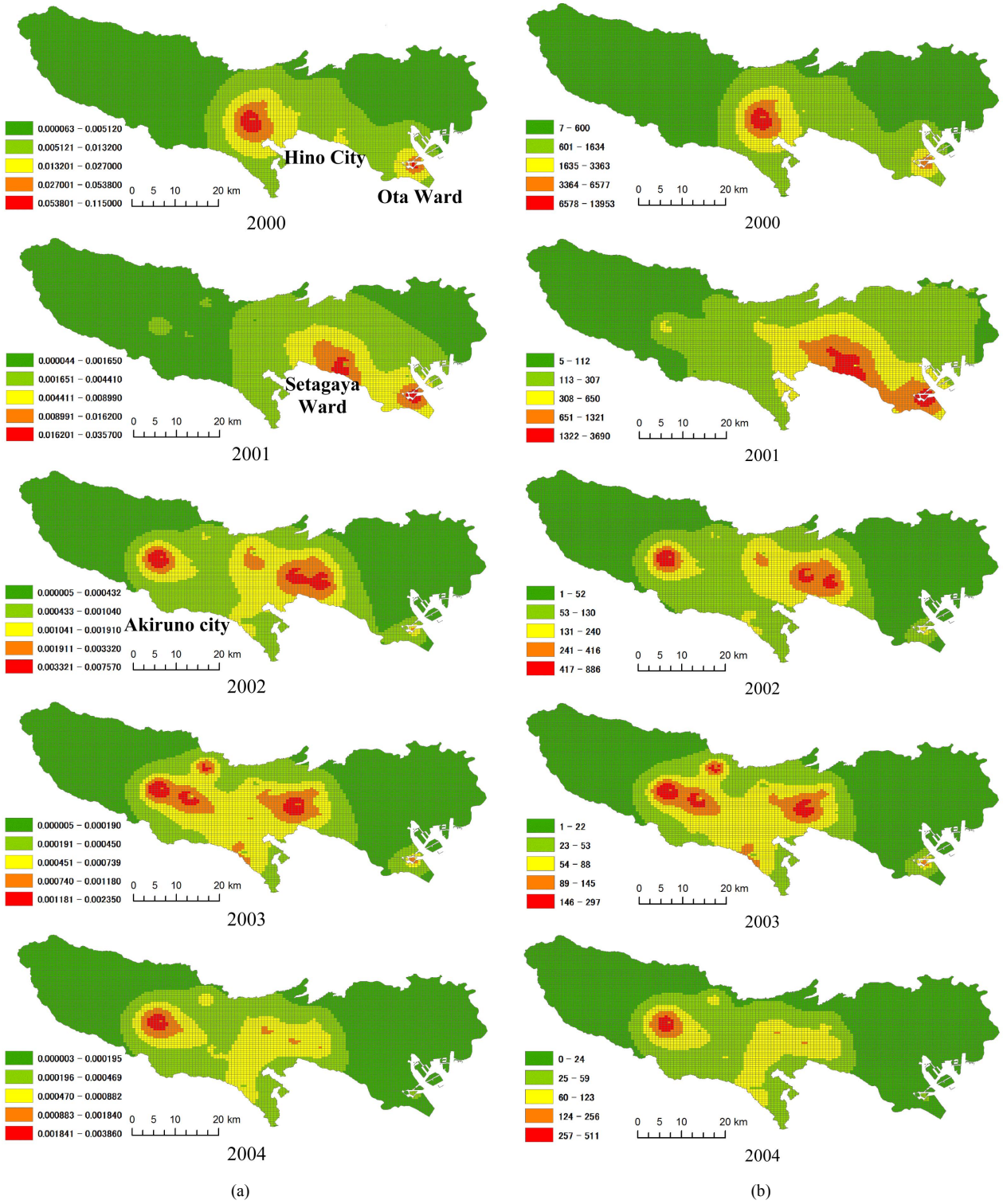


Figure 2. Total dioxin emissions from general waste incinerators for the period 2000 to 2008.

2) Environmental risk evaluation results and consideration

a) The risk to humans

The atmospheric levels estimated using ADMER and the population data were superimposed using the GIS, and thereby, levels of exposure to dioxins were grasped, and the risk to humans was evaluated. In Japan, national censuses are carried out every five years, and data was available for the years 2000, 2005 and 2010. Therefore, for each year in the period evaluated, population data of the census year closest to that year was used. It was possible to confirm that from the year 2003 onwards, in the



(a) (b)

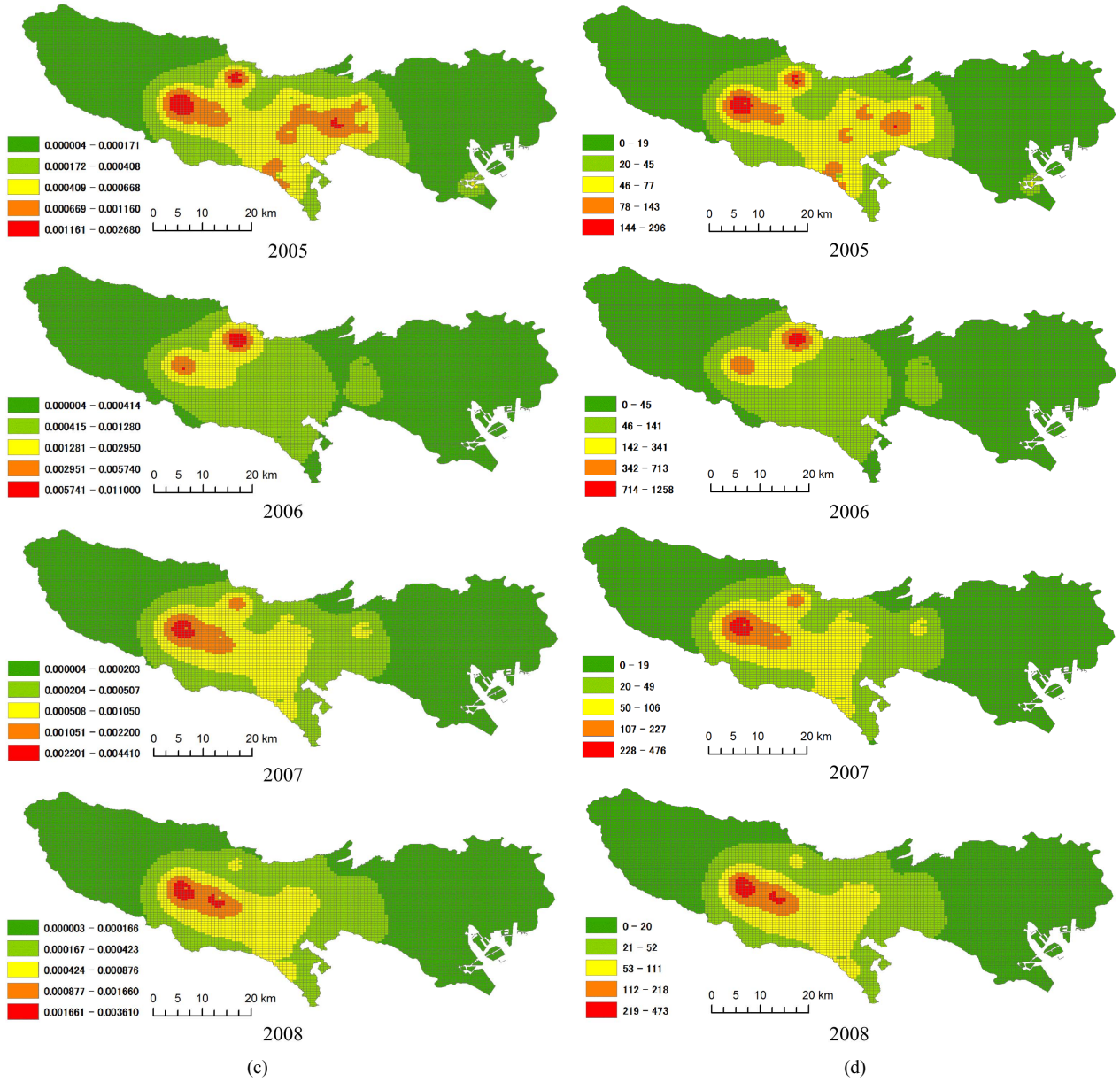
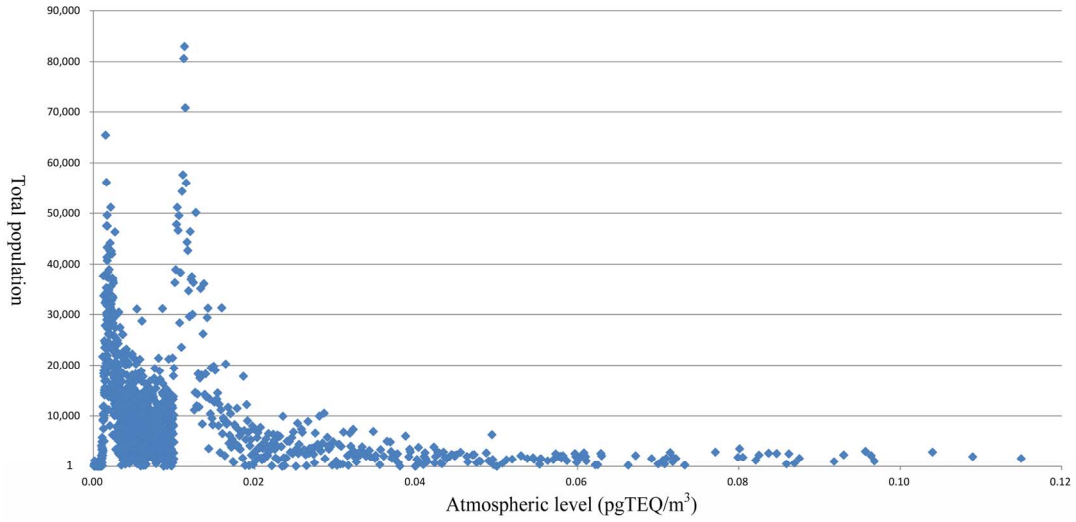


Figure 3. Dioxin atmospheric level distribution (pgTEQ/m^3, left) and deposition amount distribution (pgTEQ/m^2, right) for the period 2000 to 2008.

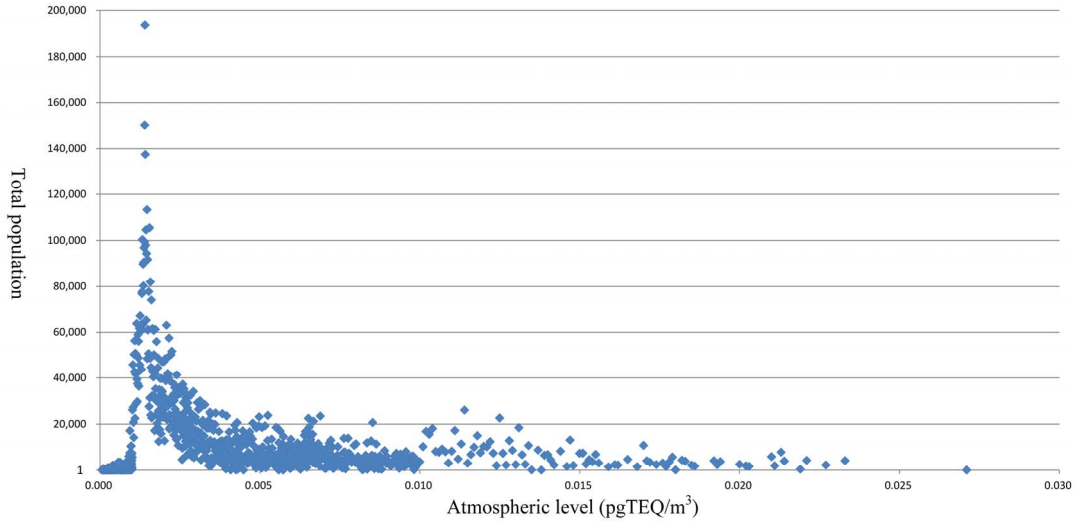
entire Tokyo Metropolis, a tendency continued for population to be concentrated in places with levels in the range of 0.001 pg TEQ/m^3 or less. Therefore, **Figure 4** shows total population figures aggregated for each atmospheric level for the period 2000 to 2002. Further, looking at **Figure 4**, it can be seen that the population was concentrated in places with levels in the range of 0.05 pgTEQ/m^3 or less in the year 2000, 0.025 pg TEQ/m^3 or less in the year 2001, and 0.005 pgTEQ/m^3 or less in the year 2002. Therefore, because the atmospheric levels of dioxins in the entire Tokyo Metropolis during the nine year period were much less than the Japanese environmental standard of 0.6 pgTEQ/m^3 (Central Environment Council, 1999) [32], it can be determined that risk to humans from the amount of exposure to dioxins released from general waste incinerators is not an issue. However, these evaluation results are based on analysis performed using a 500 m mesh unit spatial scale and on estimations of atmospheric diffusion of dioxins released from general waste incinerators only.

b) Risk to the environment

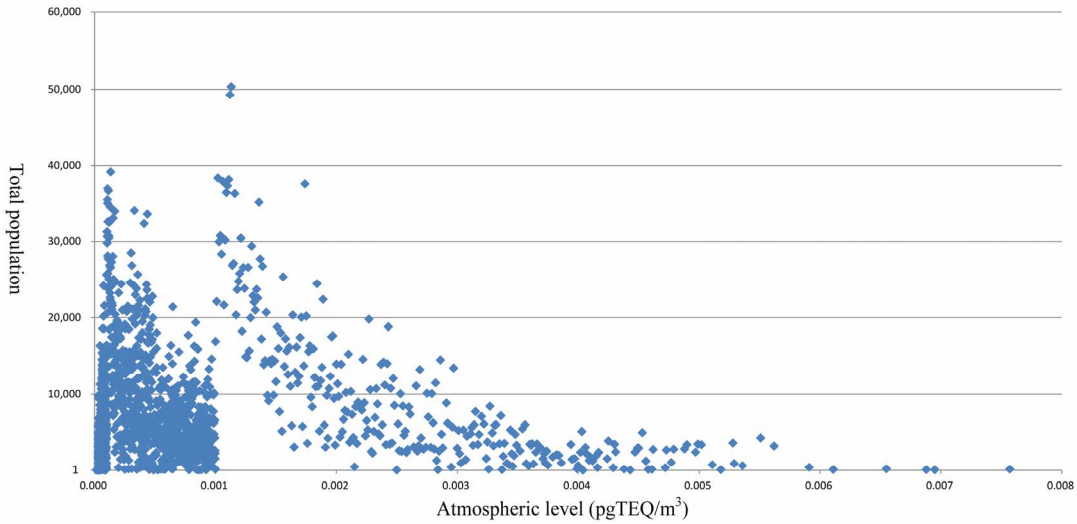
The deposition amounts estimated using ADMER and the land-use data were superimposed using the GIS, and thereby, accumulation levels in each land environment were grasped, and the risk to the environment was evaluated. The deposition amounts used in the superimpose-



2000



2001



2002

Figure 4. Total population figures per atmospheric level for the period 2000 to 2002.

tion were the aggregate amounts for the deposition amounts which accumulated during the nine-year evaluation period. Here, because dioxins have a long half-life in the environment, the fact that once a dioxin has deposited, it exists in the environment for a long time was taken into consideration, and rather than performing superimposition of deposition amounts for each year, superimposition was performed with the aggregate amounts which accumulated over the nine years. Concerning land use subdivision mesh data, land-use environment data from 2006, the year closest to the evaluation period, was used.

Table 3 shows the accumulated amounts of dioxins in each land environment. Using the GIS, deposition amounts (pgTEQ/m^2) in 100m mesh units and area (m^2) were calculated; these were multiplied to find the accumulated amounts (gTEQ), and accumulated amounts of dioxins were aggregated according to land use. As shown in **Table 3**, no land environment had an accumulated amount of dioxins equal to or greater than the Japanese environmental standard for dioxins in soil, which is 0.51 gTEQ per hectare [33]; therefore it can be determined that risk to the environment is not a problem. However, it was found that the land environment with the highest accumulation of dioxins in the entire Tokyo Metropolis was land for buildings. This is because the Tokyo Metropolis is a region containing a large-scale urban area, so land for buildings occupies about half of its area. Concerning forest and land for other uses (sports grounds, parks, airports, racecourses and so on), between which there is a fourfold difference in total area, a similar amount of accumulated dioxins was found. These results show that in the Tokyo Metropolis there is a tendency for dioxins to accumulate more easily in land environments such as land for buildings, sports grounds, parks and so on than in forests. Further, in the Tokyo Metropolis, forests are mostly concentrated in the western part; therefore, it can be said that dioxins released from general waste incinerators in this region have a small impact on land environments. Accumulation of dioxins was also found in rice fields (paddy fields which are moist all year due to improper irrigation, dry rice fields and so on), other farming land (wheat fields, orchards and so on), riverland, lakes and marshes (watercourses and natural lakes), which can be pathways of dioxin ingestion for humans. However, the accumulated amounts were not as great as those for land for buildings, forests and land for other uses.

6. Evaluation of the Vicinity of Sources

6.1. Evaluation Targets

The evaluation targets of this section were incinerators with a fire bed area of 0.5 m^2 or greater, or an incineration capacity of more than 50 kg/h or greater which are required by the Law Concerning Special Measures against Dioxins to conduct independent measurements once a year or more. In addition to the general waste incinerators which were evaluation targets in the previous section, large-scale sludge incinerators with an incinera-

Table 3. Accumulated amounts of dioxins in each land environment.

Land use type	Area (ha)	Accumulated amount (gTEQ)	Accumulated amount per unit of area (gTEQ/ha)
Rice fields	685	0.018	2.6E−05
Other farming land	5788	0.093	1.6E−05
Forest	58,233	0.270	4.6E−06
Wasteland	1518	0.016	1.1E−05
Land for buildings	84,698	1.252	1.5E−05
Land for arterial traffic	4414	0.066	1.5E−05
Land for other uses	14,558	0.305	2.1E−05
Riverland, lakes and marshes	6301	0.090	1.4E−05
Seaside	3	1.7E−04	5.7E−05
Seawaters	878	0.013	1.5E−05
Golf courses	1558	0.028	1.8E−05

tion capacity of more than 4000 kg/h managed by the Tokyo Metropolitan Government Bureau of Sewerage, and medium-sized and small incinerators with an incineration capacity of less than 4000 kg/h were also targets of evaluation. The METI-LIS analysis covers limited ranges which are the vicinity of sources, and it can evaluate environmental risk focusing on all incinerators, including incinerators that do not operate continuously throughout the year, small incinerators with low stacks and so on. **Figure 5** shows distribution maps of incinerators that are targets of evaluation in this section for the years 2000 and 2001, which were years of high dioxin contamination levels, according to the analysis results of the previous section.

Similarly, METI-LIS was used to conduct level distribution analysis which mainly focused on the vicinity of general waste incinerators in the three areas of the northern part of Hino City, the northeastern part of Ota Ward and the southwestern part of Setagaya Ward, for which the analysis results of the previous section showed particularly high contamination levels. Concerning the evaluation target range, a range of calculation of 6 km east-west and 6 km north-south was set such that general waste incinerators of the areas concerned would be included, and level distribution was output using a 100 m mesh unit. Because METI-LIS output results are output using point data (a level for each grid point), the GIS was used to store and display the output results in polygon data with a 100 m mesh unit. In order to make the position of buildings and rivers within the range of calculation easy to identify, structure perimeter lines of buildings and water edge lines from map basis information for digital map data were displayed.

The evaluation target period was set as the years 2000 and 2001, years for which the rate of contamination by dioxins was high, according to the analysis results of the previous section. For the year 2000, the northern part of Hino City, the northeastern part of Ota Ward and the southwestern part of Setagaya Ward were evaluated. For the year 2001, the northeastern part of Ota Ward and the southwestern part of Setagaya Ward were evaluated. In 2001, the atmospheric levels greatly decreased in the

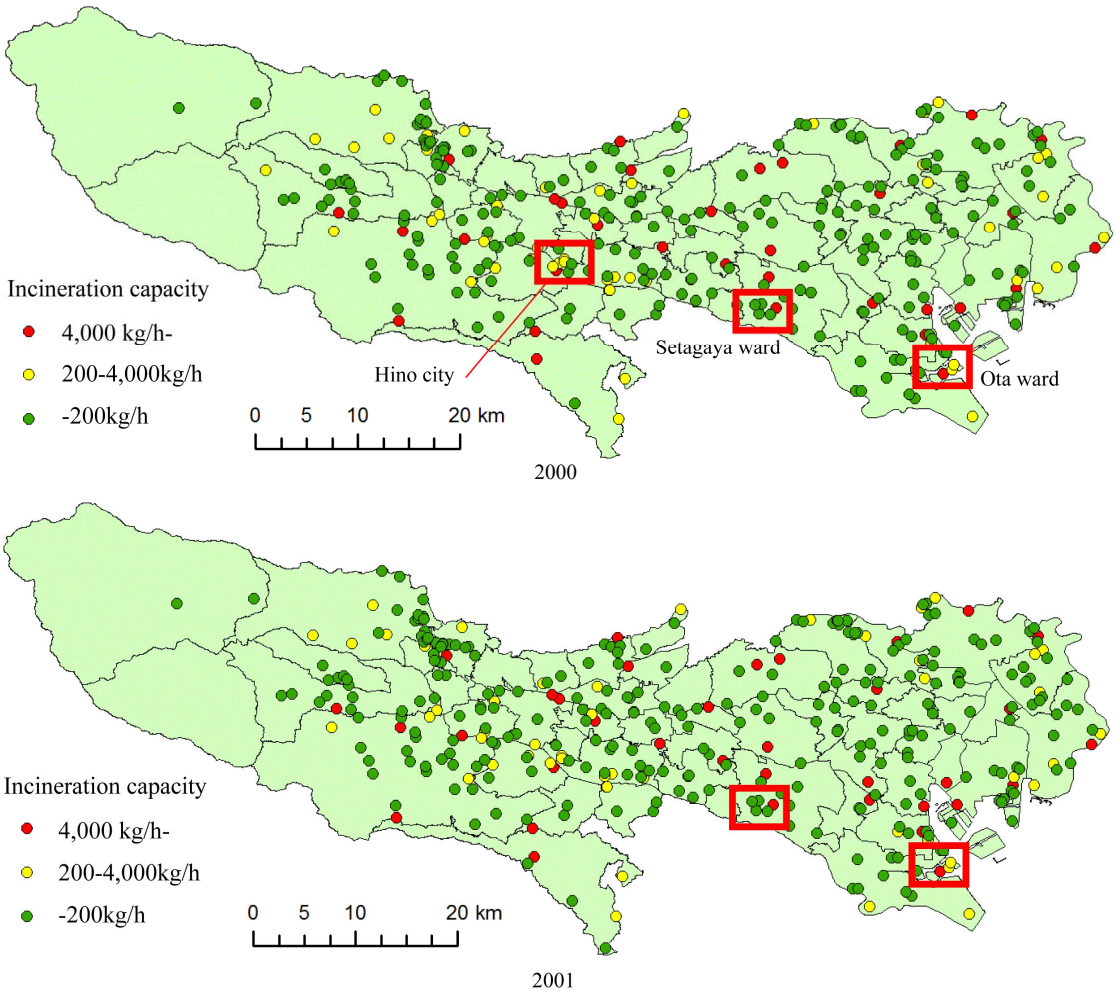


Figure 5. Distribution of all incinerators in the years 2000 and 2001.

northern part of Hino City; therefore, evaluation of this area was not conducted for the year 2001.

6.2. Calculation Conditions

1) Source data and chemical substance parameters

In order to use METI-LIS to conduct analysis of level distribution, data concerning sources, chemical substance parameters and data concerning buildings in the calculation range (refer to Section 6.2 3)) are necessary. Source data is source location (longitude and latitude), annual emissions and emissions elevation; chemical substance parameters are the molecular weight and form of the chemical substance. Concerning the dioxin parameters, referring to the related research previously mentioned [1, 4,5,7-23] and the Ministry of the Environment (2004) [3], the molecular weight of a dioxin was assumed to be 388, the molecular weight for PeCDF, which occurs in the greatest percentage at incineration plant exhaust gas dioxin levels, and the form of a dioxin was entered into METI-LIS as a particulate spherical shape.

2) Details of source data

Concerning the incinerators for evaluation, incinerators with an incineration capacity of less than 200 kg/h were considered small; those with an incineration capacity of 200 kg/h to 4000 kg/h were considered medium-sized; and those with an incineration capacity of more than 4000 kg/h were considered large. Calculations were performed assuming large-scale incinerators to be total continuous type incinerators which operated 24 hours a day, and medium-sized and small incinerators to be incinerators with a daily combustion time of eight hours (the normal daily incineration time for incinerators of such scales, according to Miyoshi (2004) [34]). As stack heights of each incinerator, for incinerators whose stack height was publicly available (mainly general waste incinerators), the published stack heights were entered. In the case of all the incinerators whose stack height was not publicly available (mainly small-scale incinerators managed by business establishments), Miyoshi (2004) was referred to, and the average value for the stack height of small-scale incinerators, 10 m, was input. Further, emissions calculations and operating conditions settings were performed assuming that an incinerator which went into disuse during a year of the evaluation period operated up till the day before the date that it went into disuse.

3) Building data

In analysis using METI-LIS, apart from data for sources of evaluation targets, it is also necessary to input data concerning the width and height of buildings which affect dispersion. Because practically speaking it was impossible to input all data for buildings within the calculation range, reference was made to Japan Environmental Management Association for Industry (2012) [35], and buildings for which it was possible that the effects of downwash, which is a consequence of disturbance of air current due to a building, may appear (mainly buildings with a height of 20 m or more) were identified in the surroundings of incinerators, and the height of those buildings was input.

6.3. Results and Consideration

1) Results of the analysis using the atmospheric dispersion model (METI-LIS) and consideration

Results of analysis of atmospheric levels and deposition amounts calculated using METI-LIS are shown in **Figures 6** to **8**. Because METI-LIS output results are shown using point data, setting was performed such that each piece of point data was displayed in the center of gravity of 100 m mesh unit data (in this section, land use subdivision mesh data); and using GIS, the METI-LIS analysis results were stored in the 100 m mesh unit data. Therefore, the analysis results are shown using a 100 m mesh unit level distribution. Through carrying out these steps, 100 m mesh unit atmospheric level and deposition amount level distributions were grasped for areas estimated to have a high risk of contamination by the ADMER evaluation results of the previous section, and it was possible for the impact on population and land environments to be considered in more detail for these areas.

Values of more than 0.6 pgTEQ/m^3, which is the Japanese environmental standard for dioxin atmospheric levels, were estimated for the northeastern part of Ota Ward. An example of a cause for why levels exceeding the environmental standard were estimated is the impact of buildings near large-scale incinerators. The height of the stack of the large-scale incinerator emitting the most dioxins into the atmosphere in this area is 41 m. However, because buildings in the vicinity of this incinerator are about 30 m high, a downwash, which is a disturbance in air current caused by a building, occurs, and dioxins which should be dispersed into the atmosphere accumulate in the surroundings of the stack of the incinerator; thus, very high dioxin levels which exceeded the environmental standard were estimated in places.

In the evaluation using METI-LIS described in this section, besides general waste incinerators, large-scale sludge incinerators managed by the Tokyo Metropolitan Government Bureau of Sewerage, and medium-sized and small incinerators were also evaluation targets; however, the region with the highest levels in **Figures 6** to **8** is in the neighborhood of a general waste incinerator shown in **Figure 2**; therefore, it was found that atmospheric levels are greatly dependent on general waste incinerators. However, according to the Tokyo Metropolitan source data shown in **Table 1**, no incinerator included in the evaluation target range of this section had a level of dioxins in exhaust gases which exceeded the environmental

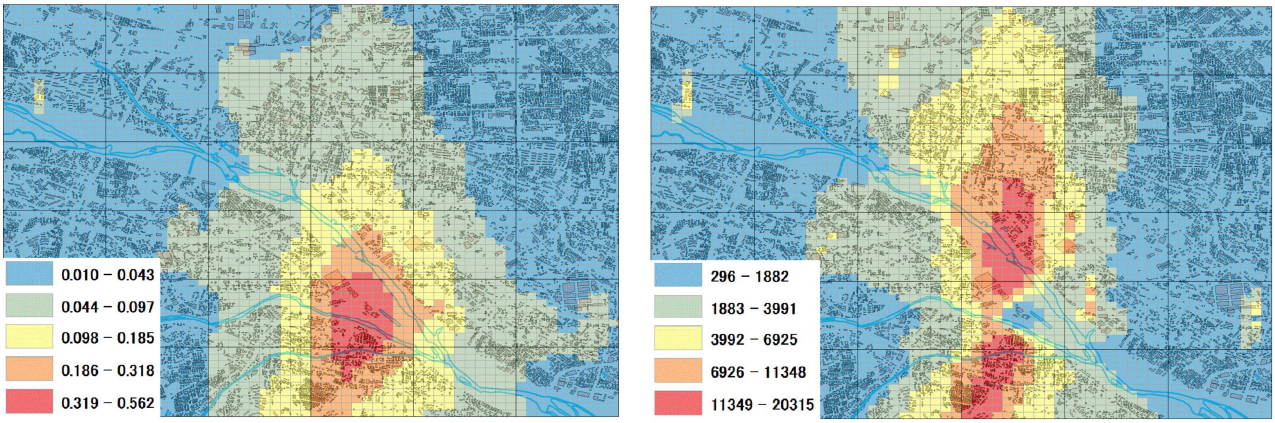


Figure 6. Dioxin atmospheric level distribution (pgTEQ/m^3, left) and deposition amount distribution (pgTEQ/m^2, right) in the northern part of Hino city for the year 2000.

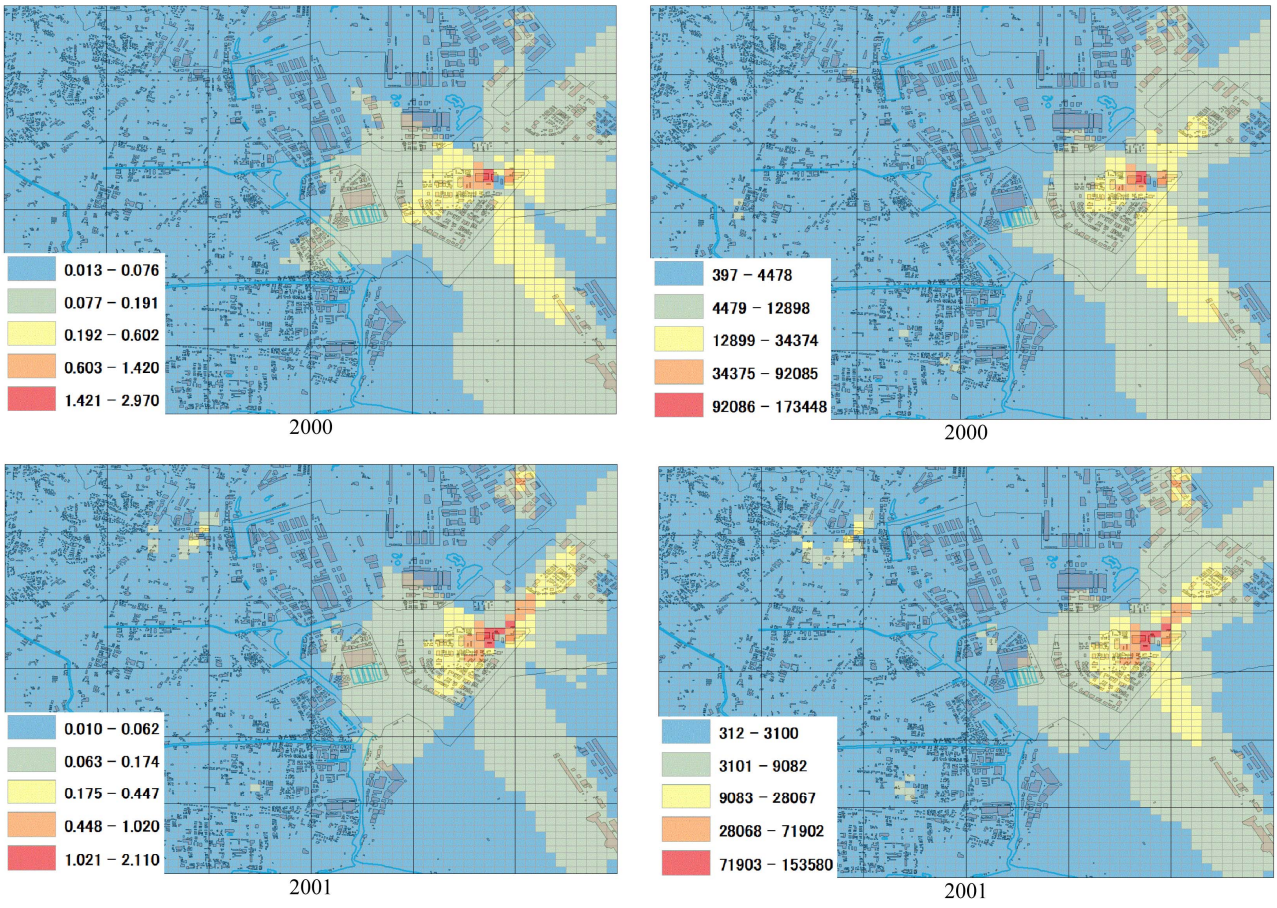


Figure 7. Dioxin atmospheric level distribution (pgTEQ/m^3, left) and deposition amount distribution (pgTEQ/m^2, right) in the northeastern part of Ota ward for the years 2000 and 2001.

standard.

2) Environmental risk evaluation results and consideration

a) Risk to humans

By superimposing the atmospheric levels estimated using METI-LIS and the population data using GIS, dioxin exposure levels were grasped, and the risk to humans was evaluated. In the national census, there is no 100m mesh unit data; therefore, in this section, as population data, subregion unit (district unit) data is used, rather than 500 m mesh unit data. In three regions (the northern part of Hino City, the northeastern part of Ota

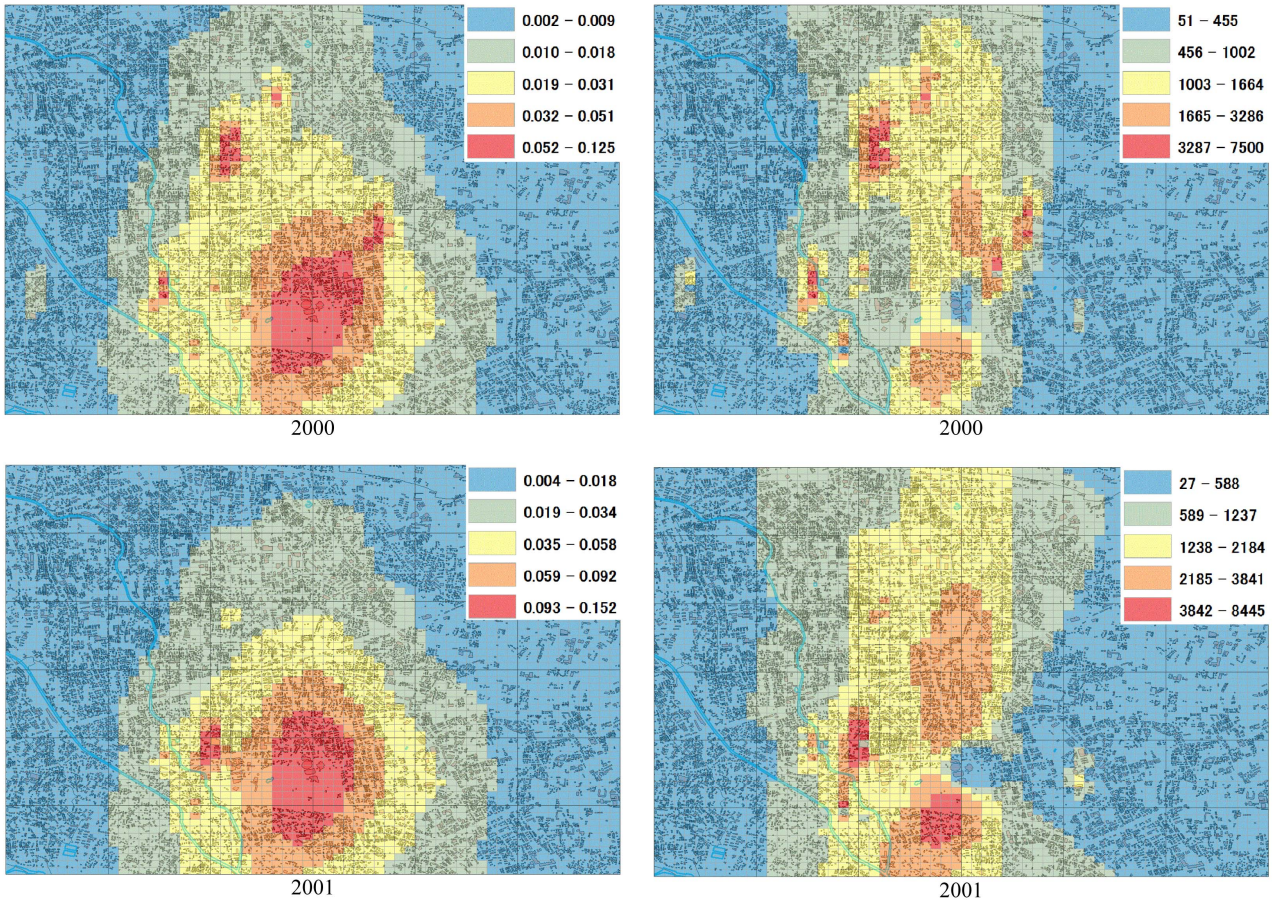


Figure 8. Dioxin atmospheric level distribution (pgTEQ/m^3, left) and deposition amount distribution (pgTEQ/m^2, right) in the southwestern part of Setagaya ward for the years 2000 and 2001.

Ward and the southwestern part of Setagaya Ward), subregions within the calculation range of METI-LIS described in Section 6.1 were selected. The combined total of atmospheric levels was found for these regional units, and the amount of exposure with respect to the total population was found. Concerning the population data, considering the evaluation target period, data for the year 2000 was used.

From the above-mentioned three regions, subregions in which the maximum atmospheric level exceeded 0.1 pgTEQ/m^3 were selected, and these are shown in **Figure 9**. Looking at **Figure 9**, it can be seen that in regions which have no residents or an extremely small population, the atmospheric levels are high. A result which was an exception is that in the northeastern part of Ota Ward, maximum atmospheric levels exceeded 0.6 pgTEQ/m^3, the environmental standard value for atmospheric levels, in five districts in the year 2000 and three districts in the year 2001. However, these subregions are industrial districts, so they do not have residents. Therefore, it can be confirmed that consideration is being given such that incinerators which emit many dioxins are not set up in residential areas and the like, and people are not exposed to high levels of dioxins.

However, Ochikawa of Hino City is unique in that although it is a subregion with a population of over 5000, it was found that the people there were exposed to high atmospheric levels in the year 2000. For this region, an estimation of a maximum level of 0.56 pgTEQ/m^3 was obtained, and because this is close to 0.6 pgTEQ/m^3, the Japanese environmental standard for dioxin atmospheric levels, the risk to the environment from dioxins caused by incinerators is high. This result is consistent with the fact that in the results of investigation of dioxins in the environment for the year 2000 for the Tokyo Metropolis [36] and in the results of [5], it was shown that Ochikawa in Hino City was a region with a very high level of dioxins, and that among those dioxins there was a high proportion of PCDD which originates from incinerators.

b) Risk to the environment

The deposition amounts estimated using METI-LIS and the land use data were superimposed using the GIS, and thereby, accumulation levels in each land environment were grasped, and the risk to the environment was evaluated. As land use data, land use subdivision mesh data from 1997, the year closest to the evaluation target

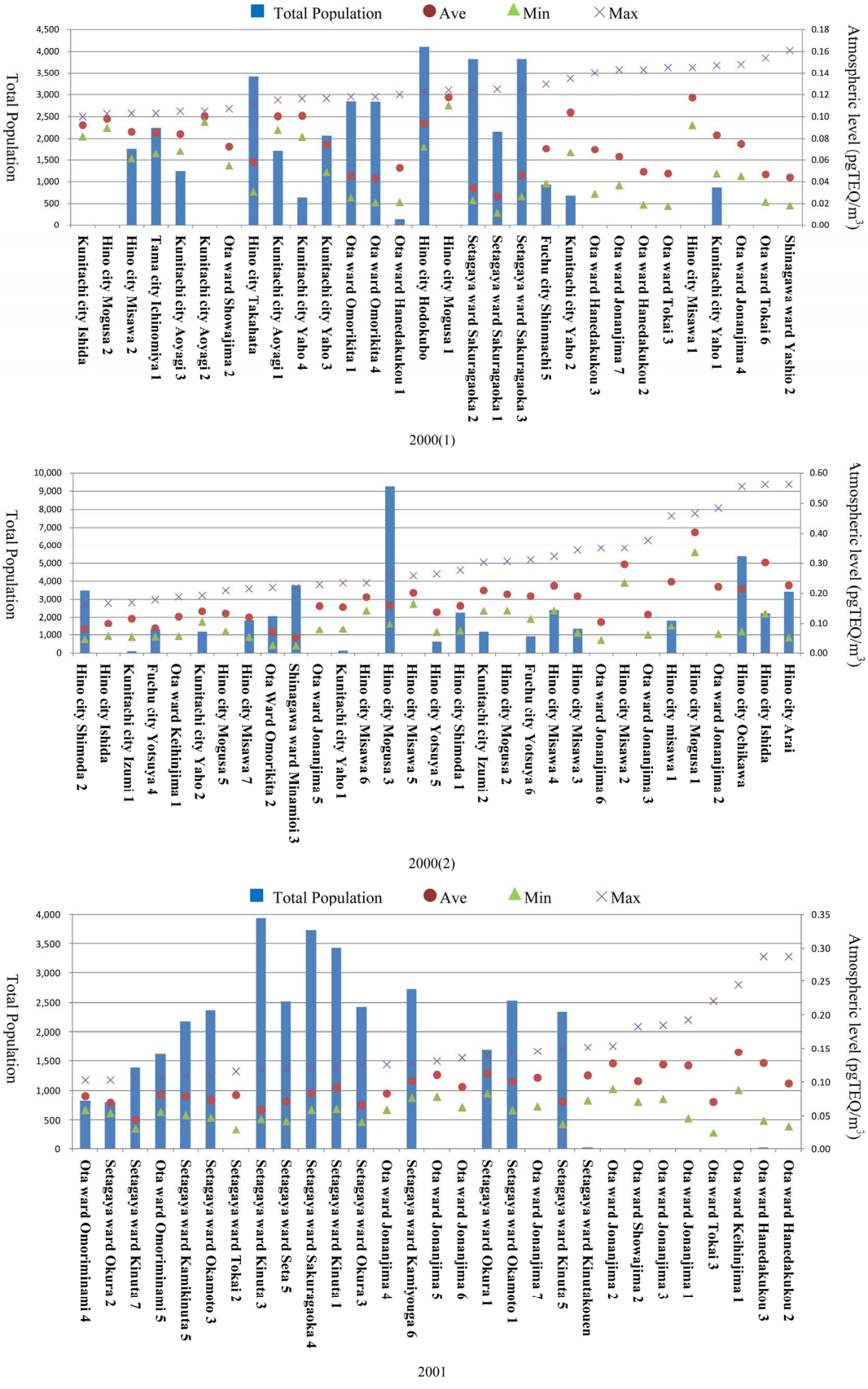


Figure 9. Subregions with a maximum atmospheric level of 0.1 pgTEQ/m^3 or more.

period, was used. For the northeastern part of Ota Ward and the southwestern part of Setagaya Ward, the evaluation target period was 2000 to 2001; therefore, the deposition amount was the total amount that accumulated in these two years. Deposition amount was aggregated according to land use for each region, and using a similar calculation method to that of the previous section, the accumulated amounts of dioxin in each land environment of each region were grasped. **Table 4** shows the accumulated amounts in each land environment of the three regions. None of the three regions had a land environment with an accumulated dioxin amount which exceeded the Japanese environmental standard for dioxins in soil, which is 0.51 g TEQ per hectare, so it can be determined that there is no risk to the environment.

However, in land environments where dioxins accumulate, different characteristics were observed for each of the three regions. In the northern part of Hino City, land for buildings occupies the majority of the area; however, because the area of "Rice fields", "Other farming land" and "Riverland, lakes and marshes" in Hino City is large compared with the other two regions, the accumulation amounts for these land uses were also large. These are land environments which must be focused on as routes of ingestion to the human body, and it is possible that in rice fields and other farming land, dioxins will be absorbed from roots via soil and accumulate in agricultural products. Further, in riverland, lakes and marshes, dioxins may move long distances with the flow of water and contaminate environments of other regions, and there is a risk that organisms which inhabit water bodies will take dioxins in the water and in bottom material into their bodies, and these dioxins will be biomagnified in large fish via the food chain.

Although land for buildings occupies the largest area in the northeastern part of Ota Ward, the amounts of dioxin accumulation in industrial areas ("Land for other uses" in **Table 4**) and bodies of seawater are higher relative to area. The surroundings of the large-scale incinerator within the industrial area which showed a maximum dioxin level which exceeded the environmental standard for atmospheric levels mentioned above is close to Tokyo Bay; therefore, there is a strong possibility that the dioxins released from it have also accumulated in the bottom material of Tokyo Bay. However the air flow in the atmosphere flows towards Tokyo Bay; therefore the impact of dioxins on land for buildings (mainly residential areas) in this area is low. Therefore, in this area, there is a stronger necessity to conduct a more detailed survey involving actual measurements in the industrial areas, rather than the residential areas.

In the southwestern part of Setagaya Ward, a large amount of dioxins has accumulated in land for buildings. This is because approximately 80% of the land in this area that is in the range of calculation is classified as land for buildings. In the estimation results of this section, it was determined that risk to the environment is not a problem; however, it is possible that people could ingest dioxins orally from soil in gardens, parks and so on in the neighborhood of residential areas in land for buildings. In particular, since children spend more time enjoying themselves outdoors than adults do, it is necessary to take into consideration the risk that they may ingest a larger amount of dioxins.

7. Conclusion and Future Research Topic

The conclusion of this study can be summarized into the

Table 4. Amounts of accumulated dioxins in land environments in each of the three regions.

Land use type	Northern part of Hino City			Northeastern part of Ota Ward			Southwestern part of Setagaya Ward		
	Area (ha)	Deposition amount (gTEQ)	Accumulated amount per unit of area (gTEQ/ha)	Area (ha)	Deposition amount (gTEQ)	Accumulated amount per unit of area (gTEQ/ha)	Area (ha)	Deposition amount (gTEQ)	Accumulated amount per unit of area (gTEQ/ha)
Rice fields	210	0.006	2.8E−05	26	0.002	6.2E−05	1	1.1E−05	1.0E−05
Other farming land	247	0.008	3.1E−05	-	-	-	131	0.002	1.8E−05
Forest	71	0.003	4.2E−05	-	-	-	47	0.001	2.2E−05
Wasteland	25	5.2E−04	2.1E−05	1	1.7E−05	1.6E−05	5	1.3E−04	2.4E−05
Land for buildings	2104	0.051	2.4E−05	1947	0.081	4.1E−05	3112	0.043	1.4E−05
Land for arterial traffic	187	0.004	2.2E−05	307	0.013	4.2E−05	100	0.002	1.6E−05
Land for other uses	476	0.013	2.7E−05	996	0.098	9.8E−05	340	0.006	1.7E−05
Riverland, lakes and marshes	446	0.014	3.1E−05	82	0.002	3.0E−05	29	3.1E−04	1.0E−05
Seaside	-	-	-	3	2.9E−04	9.4E−05	-	-	-
Seawaters	-	-	-	408	0.042	1.0E−04	-	-	-

following three points:

1) A method of evaluating environmental risk in two stages was developed. In the method, two atmospheric dispersion models and GIS were used to create source data, which were used in the two stages of evaluation. In the first stage of evaluation, ADMER was used to conduct a wide-area evaluation which covered the entire area of the evaluation target region. In the second stage, METI-LIS was used to conduct a detailed limited-area evaluation which targeted the vicinity of sources.

2) In this study, incinerators were selected as sources and dioxins were selected as harmful chemical substances. The area selected for evaluation was the Tokyo Metropolis in Japan. The evaluation method proposed in this study was used to evaluate environmental risk. Through the use of atmospheric dispersion models and GIS, the behavior of dioxins emitted into the atmosphere from incinerators was estimated. By superimposing atmospheric levels and population data, the amounts of dioxins that humans exposed to were found. By superimposing deposition amounts and land use data, the amounts of dioxins accumulated in each land environment were found. Conducting these steps enabled the impact of dioxins on humans and the environment to be grasped quantitatively and visually, and the risk that dioxins emitted from incinerators pose to the environment to be evaluated.

3) In the evaluation method developed in this study, through the creation of data concerning sources, a wide area which consists of the entire area of an evaluation target region can be evaluated, and limited areas which consist of the vicinity of sources in areas with high contamination risks can be evaluated in detail. Therefore, if data concerning sources is available, the evaluation method can also be applied to harmful chemical substances other than dioxins. In particular, through using the evaluation method of this study, areas with high levels of harmful chemical substances which should be subjected to surveys involving actual measurements can be identified based on evaluation of impact on land environments, and risk communication between local residents and business people can be stimulated based on evaluation of impact on humans; therefore, the evaluation method can effectively support harmful chemical substance countermeasures and measures to improve the environment.

Future research topic is to verify the usefulness of the evaluation method developed in this study by using the method in evaluations concerning harmful chemical substances which have other sources.

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