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Dispersion modeling and health risk assessment of dioxin emissions from a municipal solid waste incinerator in Hangzhou, China*

Xiao-jun MA, Xu-guang JIANG, Yu-qi JIN†‡, Hong-mei LIU, Xiao-dong LI, Tong CHEN, Jian-hua YAN

(*State Key Laboratory of Clean Energy Utilization, Institute for Thermal Power Engineering, Zhejiang University, Hangzhou 310027, China*)

†E-mail: jinyuqi@zju.edu.cn

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Abstract: The emission of dioxins from municipal solid waste incinerators (MSWIs) has become a widespread concern. The effect of meteorological parameters (wind speed, atmospheric stability and mixing height) on the hourly ground level concentration (GLC) of dioxins was estimated using air dispersion models. Moreover, the health risks of dioxin exposure were evaluated for children and adults using the Nouwen equation. The total environmental exposure via air inhalation and food ingestion was calculated, based on linear fit equations. The results indicate that potentially severe pollution from dioxins occurs at a wind speed of 1.5 m/s with atmospheric stability class F. In addition, local residents in the study area are exposed to severe weather conditions most of the time, and the risk exposures for children are far higher than those for adults. The total exposure for children far exceeds the tolerable daily intake of dioxin recommended by the World Health Organization (WHO) of 1–4 pg TEQ/(kg·d) under severe weather conditions. Results from modeling calculations of health risk assessment were consistent with dioxin levels obtained during actual monitoring of emissions.

Key words: Dioxins, Meteorology, Air dispersion model, Health risk assessment, Dioxin exposure
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1 Introduction

Since the first detection of dioxins in the flue gas of municipal solid waste incinerators (MSWIs) by Olie *et al*. (1977), the emission of these compounds has become a concern worldwide. There are three main ways that dioxins are emitted from MSWIs into the environment, by flue gas, fly ash, or bottom ash (UNEP, 2005). However, atmospheric transport is considered the principal pathway for regional and global movement of dioxins from combustion sources (Brubaker and Hites, 1997; Correa *et al*., 2004). In China, MSWIs are considered to be the third in importance as a source of dioxins (UNEP, 2005). A national standard of dioxin emission from MSWIs established in China (GB 18485–2001) is generally less stringent than those in developed countries. Currently, the Ministry of Environmental Protection (MEP) of the People's Republic of China is proposing tighter guidelines on emissions of dioxins from MSWIs.

Due to their persistence, fat-solubility, and accumulation, dioxins are found widely in the environment, in air, soil, water, and animal and plant fats (Koester and Hites, 1992; Lohmann and Jones, 1998; Welsch-Pausch and McLachlan, 1998; Kanematsu *et al*., 2006; Kao *et al*., 2007; Yan *et al*., 2008), ultimately endangering human health through the food chain (Päpke, 1998). Emissions of dioxin into the environment and their potential effects on human health and the environment in the vicinity of MSWIs have thus been the subject of extensive research (Chang and Lee, 1998; Lorber *et al*., 1998; Basham and Whitwell, 1999; Lorber *et al*., 2000; Ma, 2002;

‡ Corresponding author
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Xu *et al*., 2008). In this study, the impacts of wind speed, atmospheric stability, mixing height, and plume rise on the ground level concentration (GLC) of dioxin emissions from MSWIs were investigated. Also, air dispersion models coupled with a health risk assessment were used to assess environmental exposure to dioxins under severe weather conditions.

2 Models and methods

2.1 Incinerator and emission characterization

The MSWI plant is located in the northeast of Hangzhou city, China. The plant is equipped with three circulation fluidized bed incinerators (CFBIs) with a total daily capacity since 2003 of 800 t municipal solid waste (MSW). Flue gases are purified by means of a semi-dry scrubber sprayed with $Ca(OH)_2$ slurry and the injection of activated carbon, and are then voided to the atmosphere via a single stack. The stack height, diameter, exit gas temperature, and exit gas velocity of the MSWI are 60 m, 3.2 m, 398 K, and 12 m/s, respectively. The dioxin emission concentrations of the MSWI measured during full operation in 2003 varied from 0.054 to 0.1961 ng I-TEQ/Nm^3 (Yan *et al*., 2006), and the average dioxin emission levels of three CFBIs were in the range of 0.083–0.795 ng TEQ/Nm^3, with a maximum level of 1.51 ng TEQ/Nm^3 under unusual combustion conditions (Yan *et al*., 2008). Therefore, the emission concentration of 1.0 ng TEQ/Nm^3 was selected as the emission parameter for our simulation model.

2.2 Meteorological data

Sequential hourly surface meteorological data from 2002 to 2007 were obtained from the Meteorological Bureau of Hangzhou. Atmospheric stability was classified using the Pasquill stability classification method, as severely unstable, unstable, weakly unstable, neutral, stable, and extremely stable, respectively denoted as A, B, C, D, E, and F (Pasquill, 1961). The air dispersion models require meteorological data for hourly wind speed, wind direction, temperature, atmospheric stability, and mixing height, and emission characteristics including stack height, inner diameter, emission rate, exit gas velocity, and exit gas temperature. Atmospheric chemistry is ignored and wind coordinate systems are converted to geographic coordinates using a Cartesian system prior to performing the dispersion calculations.

2.3 Air dispersion models

If wind speeds were equal to or greater than 1.5 m/s (U≥1.5 m/s), a Gaussian continuous elevated point source air dispersion model was used to predict air pollution. This model is widely used for predicting the dispersion of continuous, buoyant air pollutant plumes originating from ground-level or from an elevated source (Turner, 1994). It assumes that the average wind speed along the x direction, specifically the turbulent diffusion rate of the average wind along the x direction, is much faster than the advection transport rate of the average wind speed. The model algorithm is described as follows:

$$C(x,y,0,\mathrm{He})=\frac{Q}{2\pi U\sigma_y\sigma_z}\exp\left(-\frac{y^2}{2\sigma_y^2}\right)\cdot F, \tag{1}$$

where C is the concentration of emission (g/m^3); Q is the source pollutant emission rate (g/s); U is the average wind velocity at the stack outlet (m/s); y is the horizontal transversal distance vertical to the average wind direction (m); σ_y is the horizontal transversal dispersion parameter vertical to the average wind direction (m); and σ_z is the dispersion parameter in the vertical direction; He is the effective plume rise; and F is given by

$$F=\sum_{n=-k}^{k}\left\{\exp\left(-\frac{(2nh-\mathrm{He})^2}{2\sigma_z^2}\right)+\exp\left(-\frac{(2nh+\mathrm{He})^2}{2\sigma_z^2}\right)\right\}, \tag{2}$$

where h is the mixing height (m), and the value k in Eq. (2) is taken as 4.

If wind speeds are less than 1.5 m/s, the diffusion along the x direction cannot be neglected. The Gaussian puff dispersion model is used as referred to Gu and Li (2002):

$$C(x,y,0,\mathrm{He})=\int_0^{+\infty}\frac{2Q\mathrm{d}t}{(2\pi)^{1.5}\sigma_x\sigma_y\sigma_z}\exp\left[-\frac{(x-Ut)^2}{2\sigma_x^2}\right]\cdot\exp\left(-\frac{y^2}{2\sigma_y^2}\right)\cdot\exp\left(-\frac{\mathrm{He}^2}{2\sigma_z}\right), \tag{3}$$

where σ_x is the diffusion parameter along the wind direction (m), t is the puff dispersion time (s). A wind speed of 0.1 m/s or less was considered as a no-wind condition in this study.

In the above two models, the wind speed at the stack-top is used for calculating transport and dilution. It is assumed that the wind profile is reasonably well approximated as a power-law of the form:

$$U_z = u_s (Z/10)^p, \tag{4}$$

where U_z is the scalar mean wind speed at height Z above ground level, u_s is the scalar mean wind speed at some reference height Z_r, typically 10 m, and p is the power-law exponent.

All model parameters for both runs, with the exception of the details on source-receptor locations around the MSWI, are listed in Table 1. Further details on modeling algorithms for air dispersion models can be found in the standards of GB/T 3840-91 (1991) and HJ/T 2.2-93 (1993).

2.4 Health risk assessment model

Nouwen *et al*. (2001) provided the equation for the calculation of the dioxin daily dose for adults/children as follows:

$$\mathrm{INH} = V_r \times C_{air} \times f_r \times t_f \,/\, \mathrm{BW}, \tag{5}$$

where INH is the dioxin daily inhalation exposure in ng I-TEQ/(kg·d); V_r is ventilation rate (20 m^3/d for an adult, 7.6 m^3/d for a child) (Van Hall Instituut, 1997); C_{air} is the concentration in air expressed in pg TEQ/m^3; f_r is the alveolar fraction retained in the lungs (0.75 for adults and children) (Van Hall Instituut, 1997); t_f is the time fraction (0.616 for an adult, 0.457 for a child); and BW is the body weight (70 kg for an adult, 15 kg for a child).

3 Results and discussion

3.1 Study area and data used

3.1.1 Meteorological background

Frequency distributions of wind direction, wind speed, and atmospheric stability are the primary data which are necessary to determine the dispersion direction and concentration of pollutants in the atmosphere (Manju *et al*., 2002; Rama Krishna *et al*., 2005).

Table 1 Air dispersion model input assumptions and parameters

Description and model input	Parameter value and comment
Source characterization	2003 stack test: 0.054–0.1961 ng I-TEQ/Nm^3, with average emission average concentration from tested stack (Yan *et al*., 2006); 2007 stack test: 0.083–0.795 ng TEQ/Nm^3, with average emission average concentration from tested stacks (Yan *et al*., 2008); National emission standard: 1.0 ng TEQ/Nm^3 was selected as emission concentration
Dispersion parameter	
Terrain	Suburban
Regulatory default option	Yes
Buoyancy induced dispersion	Yes
Wind profile exponents	Regulatory defaults
Vertical potential temperature gradient	Regulatory defaults
Decay coefficient	0 (no decay of pollutant in plume)
Wind speed	From calm winds to 20 m/s, the local average wind speed is 2.08 m/s
Stability category	Regulatory defaults: class A, B, C, D, E, and F
Particle-phase deposition	Yes
Vapor-phase deposition	No
Stack height, diameter, and temperature	60 m, 3.2 m, and 398 K
Exit velocity	12 m/s
Atmospheric concentrations	2006–2007 tests: 0.059 and 3.03 pg TEQ/m^3 with an average and median value of 0.495 and 0.295 pg TEQ/m^3 (Xu *et al*., 2009)

The annual average temperature, precipitation, pressure, and relative humidity were 16.2 °C, 1500 mm, 101.1 kPa, and 76%, respectively, in the study area. Fig. 1 illustrates wind rise for the MSWI plant operating during the period from 2002–2007. The prevailing wind direction was westward and calm winds occurred in only 6.06% of total wind classes. The main annual wind classes (1.0 to 3.9 m/s classes) accounted for about 92.1% of the total, and the average wind speed was 2.08 m/s. The atmospheric stability (Table 2) was dominated by classes E and F, accounting for about 86.9% of the total, especially class F, which accounted for about 68.4%.

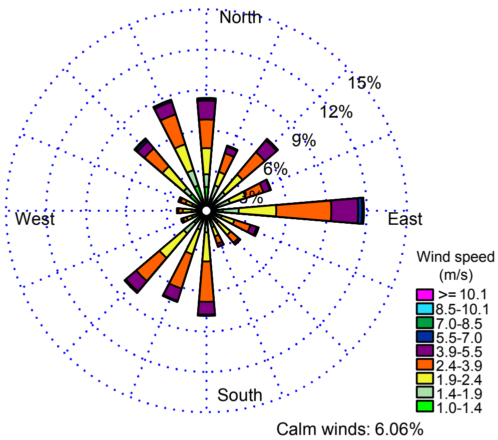

Fig. 1 Annual wind rose for the MSWI operation period (2002–2007)

Table 2 Stability classes frequency distributions (2002–2007)

Class	Ratio	Class	Ratio
A	0	D	5.53
B	0.073	E	18.51
C	1.48	F	68.34

3.1.2 Mixing height

Mixing height is defined as the height of a layer adjacent to the ground at which an emitted or entrained inert non-buoyant tracer will be mixed within a time scale of about one hour or less (Beyrich, 1997). Mixing height plays a significant role in atmospheric dispersion. In particular, a low mixing height resulting from a rather low wind speed leads to a higher concentration of pollutants near the ground (Lohmann and Jones, 1998; Rama Krishna *et al.*, 2005). Fig. 2 illustrates the effect of wind speed on mixing height under different atmospheric conditions. According to the relationship between solar elevation angle and wind speed, stability class C always occurs at a wind speed of more than 1.9 m/s. The mixing height increases with wind speed in a positive linear relationship when the stability class is in the range from unstable A to neutral D. A parabolic relationship exists between mixing height and wind velocity under class E and class F conditions. For stable atmospheric (E and F) classes and low wind speeds, the mixing height is very low, suggesting that a stable atmosphere is not conducive to atmospheric pollutant dispersion. Hence, ground level pollution would be significant in the study area. In addition, the mixing height increases as the transition from stable (class F) to unstable atmosphere (class A) occurs under the same wind speed conditions.

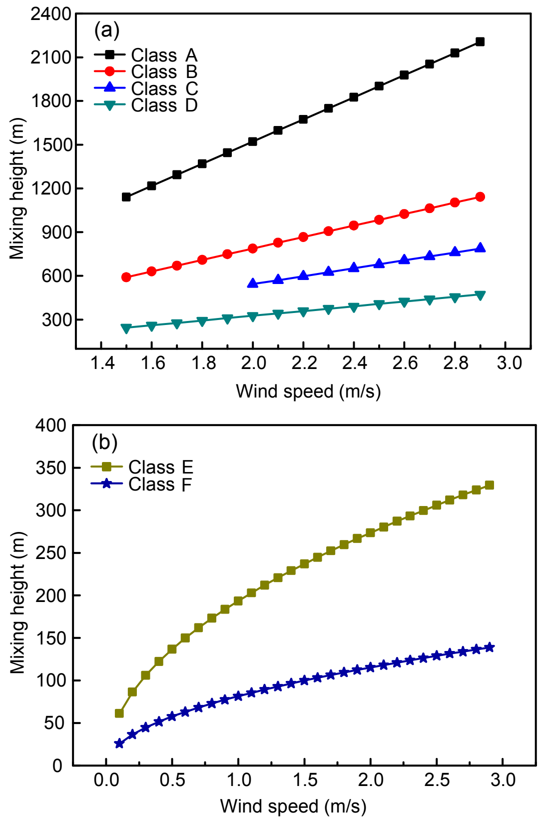

Fig. 2 Effect of wind speed on mixing height under different atmospheric stability classes A–D (a) and E and F (b)

3.1.3 Plume rise

Plume effective height is equal to the sum of the actual stack height and the plume rise. The plume rise is influenced mainly by the vertical adiabatic temperature and the smoke release rate under light wind or calm wind conditions in the Gaussian puff model. However, it is affected mainly by wind speed, smoke release rate, actual stack height, and surface roughness in the Gaussian model. The plume rise is higher under light wind or calm wind (<1.5 m/s) conditions. If the dioxin emission rate is assumed to be constant at 1.0 ng TEQ/Nm^3, the maximum height of the plume rise could reach about 260 m.

Fig. 3 illustrates the effect of wind speed on plume rise, where the wind speed varies from 1.5 to 2.9 m/s with atmospheric stability classes in the range of from A to F. The plume rise decreases inversely with wind speed and with stability class varying from A to F. Similarly, plume rise is very low under stable atmospheric conditions (E and F classes), but the change with wind speed is not obvious.

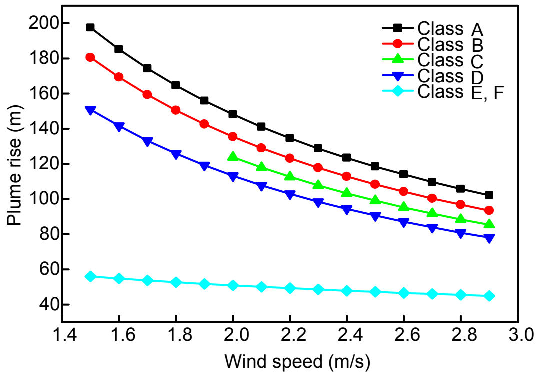

Fig. 3 Effect of wind speed between 1.5 and 2.9 m/s on plume rise under different atmospheric conditions

3.2 Maximum hourly GLC and severe weather conditions

3.2.1 U<1.5 m/s conditions

There were large differences between the hourly distributions of dioxins under different wind speeds with various atmospheric conditions (Fig. 4a). For stable atmosphere A situations, the GLC of dioxin increased with increasing wind speed, and this trend became significant when the wind speed reached 0.4 m/s. The peak dioxin concentration, about 0.391 pg TEQ/m^3, appeared at a wind speed of 0.7 m/s. A similar trend was found for neutral atmosphere D and E situations. The maximum GLC first increased and then decreased with the increase in wind speed, and the highest level occurred at a wind speed of 0.5 m/s. In the case of stable atmosphere F, the GLC remained at a very low level with almost no change with increasing wind speed.

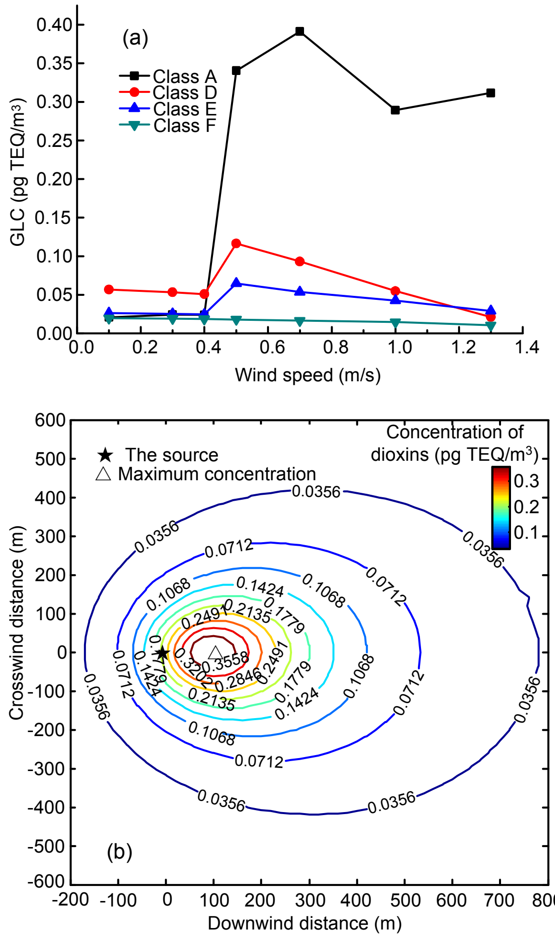

Fig. 4 (a) Effect of wind speeds less than 1.5 m/s on the maximum hourly ground level concentration; (b) Isoline figure of predicted hourly dioxin concentrations at wind speed of 0.7 m/s with extremely unstable A atmosphere

Fig. 4b illustrates the isoline figure of hourly dioxin concentrations at a wind speed of 0.7 m/s with an extremely unstable A atmosphere. The diffusion along the downwind direction cannot be neglected. The main impact area of the MSWI on the atmosphere is in the nearby region around the plant under those conditions. The highest dioxin level occurs at about 100 m from the source. Based on our calculations, the mixing height is low, about 520 m, and the plume height He is about 274 m. Therefore, airborne dioxins will persist for a long time, based on an average dioxin deposition rate in the atmosphere, of 2 m/h (Yoshida *et al*., 2001). In addition, dioxins in the atmosphere are very stable at room temperature. The half-decay of tetra-chlorinated dibenzo-*p*-dioxins (TCDDs) in the atmosphere is 2.0–8.0 d, and the half-decay of octa-chlorinated dibenzo-*p*-dioxins (OCDDs) is about 160 d (Lohmann and Jones, 1998). These results suggest high dioxin concentrations in the atmosphere. As the average emission from the actual monitoring data is in the range of 0.083–0.795 ng TEQ/Nm^3, the hourly GLC will vary from 0.0015 to 0.3111 pg TEQ/m^3 under the severe meteorological conditions. If the emission concentration is 1.51 ng TEQ/Nm^3 (under unusual combustion conditions) under the same meteorological conditions, the hourly GLC will reach the highest point, about 0.59 pg TEQ/m^3.

3.2.2 U≥1.5 m/s conditions

The highest concentration of dioxins, about 0.4037 pg TEQ/m^3, occurs in the area downwind from the MSWI in the case of a wind speed of 1.5 m/s with an extremely stable F atmosphere (Fig. 5a). A decreasing trend is not obvious with increasing wind speeds under atmospheric stability classes A, D and E. Considering the meteorological conditions in this study area, the main wind classes are lower winds with an average wind speed of 2.08 m/s with dominant wind speeds below 2.5 m/s, and atmospheric stability class F accounts for 68.34% of all stability classes. Hence, the residents of the study area are exposed to severe weather conditions most of the time.

Fig. 5b illustrates the hourly concentration distributions of dioxins at a wind speed of 1.5 m/s with an extremely stable F atmosphere. The downwind influence distance is far more than 10 km and the highest level of dioxins appears at 5868 m downwind from the MSWI. Furthermore, the crosswind effect distance is about 3 km, with a very low mixing height, about 100 m. The calculated actual effective plume height He is 116 m. Similarly, the hourly GLC ranges from 8.56×10^{-4} to 0.326 pg TEQ/m^3, when the dioxin emission levels are in the range of 0.083–0.795 ng TEQ/Nm^3 at a wind speed of 1.5 m/s with stable atmosphere (F class). The hourly GLC, 0.619 pg TEQ/m^3, should exceed the guideline of the Japanese Air Quality Standard (JAQS) of 0.6 pg TEQ/m^3 under unusual combustion conditions of the MSWI (dioxin emission levels of 1.51 ng TEQ/Nm^3). Thus, if dioxin emission concentrations are controlled below or equal to 0.1 ng TEQ/Nm^3, the hourly GLC, about 0.04 pg TEQ/m^3, is far below the JAQS of 0.6 pg TEQ/m^3 under the same weather conditions. Based on Xu *et al.* (2009), the average background concentration of dioxin is 0.19 pg TEQ/m^3 in this study area. This MSWI plant has a significant impact on the surrounding environment, considering this background value and the high dioxin emission concentrations. This was confirmed by the levels of dioxin in air during monitoring, which varied between 0.059 and 3.03 pg TEQ/m^3 with an average and median value of 0.495 and 0.295 pg TEQ/m^3, respectively. These levels were among the highest in the world (Xu *et al.*, 2009) from 2007 to 2008. Hence, there would be potential danger for the ecological system downwind from the MSWI under current operating conditions.

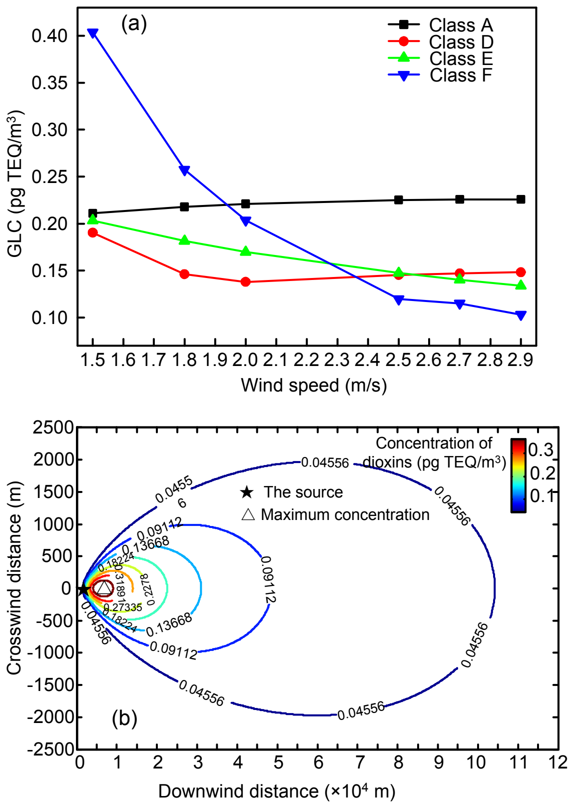

Fig. 5 (a) Effect of wind speed more than and equal to 1.5 m/s on hourly maximum ground level concentration; (b) Isoline figure of predicted hourly dioxin concentrations at wind speed of 1.5 m/s with extremely stable F atmosphere

3.3 Health risk assessment

Due to their special physical-chemical properties, the bulk of dioxins are easily adsorbed by dust and soot particles (Masunaga *et al*., 2003). Dioxins are also associated strongly with carcinogenesis, although, the precise mechanisms involved are not clear. To ensure public safety as far as possible, the tolerable daily intake of dioxins has been set very low. The WHO proposed a tolerable daily intake of dioxin of 1.0 pg I-TEQ/(kg·d), and a warning value of 4.0 pg I-TEQ/(kg·d). The US EPA (Environmental Protection Agency) also recommends that the daily inhalation exposure to dioxins should be no more than of 1.0 pg WHO_{98}-TEQ/(kg·d).

A number of recent health risk assessments using the Nouwen equation have provided details on country, year, pollution sources, daily dioxin intake (including inhalation, dermal contact and ingestion in soil and food) and the relative contribution of different exposure pathways (Table 3). The data show different risk distributions of the daily intake of dioxins via different intake pathways. Ignoring exposure through ingestion of food, the contribution of inhalation to total dioxin exposure for children ranges between 30% and 99.9%, and for adults between 70.1% and 100% (Nouwen *et al*., 2001; Lee *et al*., 2007). In contrast, if exposure through ingestion of food is included, ingestion accounts for more than 95% of total dioxin intake risks. Dioxin exposure via soil is negligible. Generally, dioxin exposure via inhalation for children is slightly lower than that for adults; conversely, exposure via daily food intake for children is higher than that for adults (Table 3).

Table 3 Daily intakes of dioxins in pg TEQ/(kg·d) and relative contribution from emission sources to the exposure scenarios in different countries

Country (area)	Year	Major source	Inhalation		Dermal contact and ingestion soil		Ingestion food		Reference
			Child	Adult	Child	Adult	Child	Adult	
Belgium (Neerland-quarter)	1980	MSWI[a]	6.32×10^{-2} (43.3)	3.56 (70.1)	8.28×10^{-2} (56.7)	1.517×10^{-1} (29.9)	0 (0.0)	0 (0.0)	Nouwen *et al.* (2001)
	1980	MSWI[b]	6.32×10^{-2} (0.6)	3.56 (1.3)	8.28×10^{-2} (0.7)	1.517×10^{-1} (0.6)	11.155 (98.7)	2.6 (98.1)	
	1997	MSWI[a]	1.15×10^{-2} (30.0)	6.51×10^{-3} (72.2)	8.28×10^{-2} (70.0)	1.517×10^{-2} (27.8)	0 (0.0)	0 (0.0)	
	1997	MSWI[b]	1.15×10^{-2} (0.4)	6.51×10^{-3} (0.9)	8.28×10^{-2} (3.0)	1.517×10^{-2} (2.1)	2.63 (96.6)	0.71 (97.0)	
Spain (Catalonia)	1998	MSWI[c]	6.08×10^{-2} (0.53)	4.62×10^{-2} (1.64)	9.9×10^{-2} (0.86)	4.815×10^{-3} (0.17)	11.4 (98.62)	2.77 (98.19)	Domingo *et al.* (2002)
	1998	MSWI[d]	4.97×10^{-2} (0.43)	3.78×10^{-2} (1.34)	1.328×10^{-2} (0.12)	3.114×10^{-3} (0.11)	11.4 (99.45)	2.77 (98.54)	
	2000	MSWI[c]	1.23×10^{-2} (0.32)	9.37×10^{-3} (1.01)	1.424×10^{-2} (0.37)	1.379×10^{-2} (1.49)	3.81 (99.31)	0.903 (97.50)	
	2000	MSWI[d]	1.01×10^{-2} (0.26)	7.66×10^{-3} (0.83)	2.824×10^{-2} (0.73)	1.069×10^{-2} (1.16)	3.81 (99.0)	0.903 (98.01)	
Korean	1999	MSWI	2.89×10^{-1} (98.93)	1.63×10^{-1} (99.58)	3.133×10^{-3} (1.07)	6.89×10^{-4} (0.42)	–	–	Lee *et al.* (2007)
	2002	MSWI	1.46×10^{-1} (99.98)	8.23×10^{-2} (100)	2.50×10^{-5} (0.02)	3.07×10^{-6} (0)	–	–	
	2005	MSWI	3.04×10^{-2} (99.97)	1.71×10^{-2} (99.53)	1.038×10^{-5} (0.03)	8.06×10^{-5} (0.47)	–	–	
China (Pearl River Delta)	2011	Area	0.067 (0.5)	0.051 (1.5)	0.009 (0.001)	0.005 (0.002)	12.137 (99.1)	3.256 (98.0)	Hu *et al.* (2011)
Japan	2003	Area	0.039 (2.3)		0.012 (0.7)		1.629 (95.7)		Government of Japan (2003)
	2009	Area	0.015 (1.5)		0.0038 (0.4)		1.0412 (97.9)		Government of Japan (2009)

Note: a: living with consumption of only commercially sold foods (milk, meat and vegetables); b: living with consumption of 25% vegetables, 50% meat and 100% milk produced in the impact area; c: dioxin exposure for adults and children living 500 m from the MSWI; d: dioxin exposure for adults and children living 1000 m from the MSWI; The value in parentheses indicates the percentage contribution of the daily intake of dioxins via different intake pathways

This may result from the different consumption behaviors and lower body weights of children (Nouwen *et al*., 2001).

By using the fit line of the least square method, the fitting curves of dioxin exposure via inhalation and daily food intake for children and adults were determined (Fig. 6). The positive correlations obtained can be expressed by Eqs. (6) and (7) for children and adults, respectively:

For children:

$$y=160.3x+1.76, \quad R^2=0.95; \tag{6}$$

For adults:

$$y=51.5x+0.38, \quad R^2=0.87, \tag{7}$$

where y is the dioxin exposure via ingestion food, and x is the dioxin exposure via inhalation.

Fig. 6 and Eqs. (6) and (7) show a good linear correlation between inhalation and food ingestion, especially for children. The daily food intake of children is far higher than that of adults. Exposure to dioxin by inhalation is proportional to the dioxin concentration of the air in Eq. (5), thus a high level of airborne dioxin must cause a high dioxin exposure through daily food intake, and children are subjected to significantly higher exposure than are adults.

Based on the discussion in Section 3.2.1, as U<1.5 m/s, the highest dioxin exposure occurs at a wind speed of 0.7 m/s with stability class A. Thus, according to Eqs. (6) and (7), the total environmental exposure to dioxins is 12.72 and 3.092 pg TEQ/(kg·d) for children and adults, respectively. Exposure via inhalation is 0.068 and 0.0517 pg TEQ/(kg·d) for children and adults, respectively, assuming that locally produced food is consumed and that the dioxin emission concentration is selected as 1.0 ng TEQ/Nm^3. In addition, based on the results of Section 3.2.2, as U≥1.5 m/s, an adverse dioxin exposure appears at a wind speed of 1.5 m/s with stability class F, where total environmental exposure to dioxins is 13.068 and 3.178 pg TEQ/(kg·d) for children and adults, respectively. Exposure via inhalation is 0.07 and 0.053 pg TEQ/(kg·d) for children and adults, respectively, based on the same assumptions.

According to actual concentrations of airborne dioxin during monitoring (Table 1), the total environmental dioxin exposure for children and adults is in the range of 3.413–86.63 and 0.789–21.38 pg TEQ/(kg·d), respectively, with an average of 15.63 and 3.81 pg TEQ/(kg·d), respectively. Thus, the modeling levels are consistent with the monitoring data. Above all, the total exposure for children under severe weather conditions is significantly high, as it mostly far exceeds the range established by the WHO, 1–4 pg TEQ/(kg·d), as the tolerable daily intake of dioxin for general toxicological effects. The results suggest that the emission of dioxins from the MSWI should be reduced. If emission concentrations are reduced from 1.0 to 0.1 ng TEQ/Nm^3, the maximum GLC of dioxins in the air and thus the total environmental exposure to dioxins decrease accordingly. In particular, for children, the highest total environmental exposure under severe weather conditions reduces from 13.068 to 2.892 pg TEQ/(kg·d), and the total exposure is within the range of 1–4 pg TEQ/(kg·d).

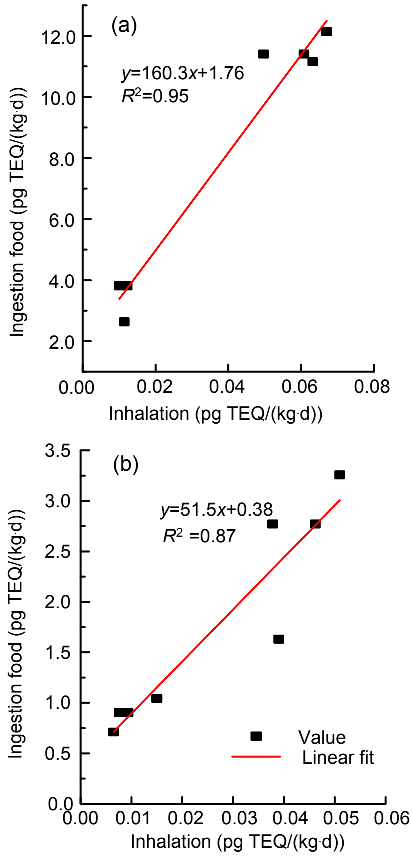

Fig. 6 Fitting curve of inhalation and ingestion food (a) for children and (b) for adults

In this study, the air dispersion models are sensitive to the location of emission releases, model parameters, meteorology, and terrain features, and dioxin health risk models are sensitive to local levels of dioxins in different environmental media (such as air, soil, water and plants) and living habits. Both uncertainty and variability should affect the modeling results. The meteorological data and an independent point source, which are considered separately in this study, may lead to errors in dioxin concentration estimates, especially for the total environmental exposure to dioxin. Thus, further research is necessary to compare long term distribution predictions and monitoring data of dioxins according to local meteorological data, terrain features, and emission releases.

4 Conclusions

Air dispersion models coupled with a health risk assessment method were carried out to assess the environmental impact of an MSWI located in the northeast of Hangzhou city. When $U<1.5$ m/s, the highest GLC and total environmental exposure to dioxins occurs at a wind speed of 0.7 m/s with stability class A. When $U\geq1.5$ m/s, the highest GLC and the total environmental exposure of dioxins appear at a wind speed of 1.5 m/s with stability class F. In accordance with previous studies using the Nouwen equation, a positive linear relationship between dioxin exposure through inhalation and through the ingestion of food was obtained in this study. Based on the actual monitoring data and calculated exposures, local residents in the study area, especially children, are exposed to high health risks.

In 2010, the MEP of China proposed a new emission guideline for large and small MSWI units, in which dioxin emission levels are reduced from 1.0 to 0.1 ng TEQ/Nm^3 for large MSWI units with a design combustion capacity of more than 150 t/d of MSW. The results of modeling and actual monitoring suggest that dioxin reduction is necessary in this study area. The emission level should be reduced to 0.1 ng TEQ/Nm^3 to meet the new discharge standards. However, the long term monitoring of dioxin levels and concentrations in various environmental media, serum and human milk should be considered.

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