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**Zuo-Wen Zhang · Chan-Seok Moon · Takao Watanabe
Shinichro Shimbo · Feng-Sheng He · Yi-Qun Wu
Shun-Fu Zhou · De-Ming Su
Jiang-Bin Qu · Masayuki Ikeda**

Background exposure of urban populations to lead and cadmium: comparison between China and Japan

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Abstract *Objectives*: To assess and compare the background exposure of the general population to lead (Pb) and cadmium (Cd) in China and in Japan. *Methods*: Food duplicates and peripheral blood samples were collected from nonoccupationally exposed subjects, viz 202 Chinese women in four Chinese cities (Beijing, Shanghai, Nanning, and Tainan) and 72 Japanese women in three Japanese cities (Tokyo, Kyoto, and Sendai) in the years 1993–1995. Wet-ashing and graphite furnace atomic absorption spectrometric methods were used for the determination of Pb and Cd levels in food and blood samples. *Results*: Geometric mean (GM) dietary Pb intake (25.8 μg/day) and the GM Pb concentration in blood (56.7 μg/l) in Chinese were significantly higher than in Japanese women (11.6 μg/day in food and 32.1 μg/l in blood), whereas Cd in food (32.1 μg/day) and Cd in blood (1.92 μg/l) in Japanese were significantly higher than in Chinese women (9.9 μg/day in food and 1.07 μg/l in blood). The intake of Pb and Cd via boiled rice accounted for 3.6% and 31.1% of the total dietary burden in Chinese, and 12.1% and 32.7% in Japanese, respectively. The Cd burden was acquired almost exclusively through the dietary route, whereas the Pb burden came from both air and food, especially in the case of the Chinese population. *Conclusions*: The background Pb exposure in the Chinese population was higher than that in the Japanese population, whereas Cd exposure was lower in Chinese women than in their Japanese counterparts.

Key words Lead · Cadmium · Food · Blood · Chinese women · Japanese women

Z.-W. Zhang · C.-S. Moon · M. Ikeda (✉)
Department of Public Health, Kyoto University Faculty of Medicine, Kyoto 606-01, Japan

T. Watanabe
Miyagi University of Education, Sendai 980, Japan

S. Shimbo
Department of Food and Nutrition, Kyoto Women's University, Kyoto 605, Japan

F.-S. He · Y.-Q. Wu
Institute of Occupational Medicine, Chinese Academy of Preventive Medicine, Beijing 100050, People's Republic of China

S.-F. Zhou
Shanghai Institute of Labour Hygiene and Occupational Diseases, Shanghai 200003, People's Republic of China

D.-M. Su
Sanitary and Anti-Epidemic Station of Guangxi, Nanning 530021, Guangxi, People's Republic of China

J.-B. Qu
Department of Hygiene, Faculty of Public Health, Shandong Medical University, Jinan 250012, Shandong, People's Republic of China

Introduction

Lead (Pb) and Cadmium (Cd) are two heavy metals recognized to be ubiquitous in the environment and toxic to humans. The two metals are persistent in the environment once discharged and stay in the human body for a long time when absorbed (International Programme on Chemical Safety 1977, 1989, 1992a, b). It has therefore been considered that the Pb and Cd levels in foods and blood are good long-term markers of general environmental pollution.

Although a number of studies have been conducted to measure or estimate the intensity of exposure of the general population to these two toxic pollutant metals in several areas, including China and Japan (Buchet et al. 1983; Watanabe et al. 1985a, b, 1987a, b, 1989a, b, 1994; Han 1988; Cikrt et al. 1992; Sartor et al. 1992; Chen and Gao 1993; Lagerkvist et al. 1993; Qu et al. 1993; Chia et al. 1994; Müller and Anke 1994; Wietlisbach et al. 1995), comparisons between countries by means of comparable methods of sampling and analysis are limited (e.g., Qu et al. 1988; Watanabe et al.

1989a; Tang et al. 1990). The present study was initiated to investigate the Pb and Cd burdens in Chinese and Japanese by means of dietary intakes (Pb-F and Cd-F) and blood levels (Pb-B and Cd-B) and to draw a baseline for expected changes in the near future in China.

Materials and methods

Study population and sample collection

In 1993–1995, 202 women in four cities in China and 72 women in three cities in Japan volunteered to take part in the present study. In practice, the volunteers in China were 50 persons from Beijing (the capital of China), 50 from Shanghai (the largest industrialized city in China), 50 from Nanning (a medium-size city located in south China near Vietnam), and 52 from Tainan (south of Taiwan). All subjects from the three cities in mainland China are on the staffs of medical institutes. Subjects in Tainan city were university staff members, hospital nurses and local inhabitants, and the values are cited from Ikeda et al. (1996). In Japan, the women taking part were 39 from Tokyo (capital of Japan), 17 from Kyoto (a city in the middle of Japan) and 16 from Sendai (a city in northeast Japan). All Japanese subjects were housewives. Thus, both Chinese and Japanese subjects were all urban inhabitants. For locations of the study sites, see Fig. 1.

Occupational exposure to any toxic substances, including Pb and Cd, was ruled out in a medical interview. Only nonsmoking adult women who were not habitual drinkers were selected as study subjects, because it is known that dietary metal intake and blood metal concentration differ between the two sexes even in persons who live together and that smoking and drinking can elevate blood metal concentrations (Watanabe et al. 1982; 1983; Ikeda et al. 1989). The average age was 38.4 years (range 20–66 years) in Chinese subjects and 53.3 (37–68) years in Japanese subjects. A preliminary study by multivariate analysis showed that age did not affect Pb-F, Cd-F, Pb-B and Cd-B in the Japanese population and affected only Pb-B significantly ($P < 0.05$), though weakly, among Chinese participants. Thus, effects of age were not taken into account in the further analysis.

The study was conducted by the same research group following the same protocols at all study sites as described previously (Ikeda et al. 1996). Samples of 24-h duplicates of food, including three meals, any snacks and drinks (even plain water if taken), were collected for a whole day immediately before the survey day. After collection from the donors, the food constituents (after elimination of nonedible portions) of each duplicate were separated for weighing. All edible components were pooled and homogenized with an electric blender. Blood samples were taken from the cubital vein of each subject between 10:00 a.m. and 12:00 noon into Venoject vacuum blood collection tubes (Terumo, Tokyo), due care being taken to avoid possible metal contamination as previously described (Watanabe et al. 1982). Both blood samples and homogenates were chilled on site after collection, kept iced during transportation, and then kept at $-20\,^{\circ}C$ until analysed in the laboratory.

Analyses of metals in food and in blood

The determinations of Pb and Cd in food duplicates and blood samples were performed in the same laboratory and by the same researchers. Part of each sample (1 ml of each blood sample or 6 g of each food duplicate sample) was taken into a Teflon tube and wet-ashed by gradual heating up to 160–180 °C for 8–10 h in a block digester (Watanabe et al. 1982) in the presence of mineral acids (15 ml nitric acid + 0.2 ml sulfuric acid + 2 ml perchloric acid in case of food homogenates, and 2 ml nitric acid + 0.25 ml perchloric acid in the case of blood samples). The final wet-ash of food homogenate was diluted to 10 ml and that of blood to 5 ml with deionized water and 0.1 ml hydrochloric acid for preservation.

One portion of the wet-ash was analysed for Pb and Cd using a graphite furnace atomic absorption spectrometer (Hitachi Model Z-8100) connected with an automated liquid sampler (Hitachi Model SSC-200), the Zeeman effect being exploited for background correction. Ammonium nitrate (final concentration 6.6%) and ammonium phosphate (2%) were used as matrix modifiers for Pb and Cd analysis, respectively. Pb and Cd in food duplicates and in blood were measured at 283.3 nm (7.5 mA) and 228.8 nm (7.5 mA), respectively, with the standard addition method. Other analytical conditions were as previously described in detail (Watanabe et al. 1983, 1985a, 1996). Daily dietary intake of Pb and Cd was calculated from the total weight of food (g/day) and the metal concentration in the wet-ash (μg/l).

In addition, 244 boiled rice samples were collected from the subjects. The samples were wet-ashed and analysed by the same method of food analysis. Dietary intake of Pb and Cd via boiled rice (PB-BR and Cd-BR) were calculated from the Pb or Cd concentration in boiled rice samples and the daily consumption of boiled rice.

Fig. 1 Locations of study sites in China and Japan

Quality assurance

The reliability of the method used was tested by analysis of certified reference materials. Results of these tests for Pb and Cd are summarized in Table 1. There was good agreement of the analytical results with the certified values. Recovery experiments were conducted by adding standard Pb and Cd solutions to redistilled water and food homogenate (5 samples, respectively) at the final spiking concentrations (in wet-ash) of 15.4 μg/l for Pb and 1.5 μg/l for Cd, and the spiked samples were subjected to wet-ashing followed by atomic absorption spectrometry. Deionized water samples and food homogenate samples without Pb or Cd addition (5 samples each) were treated similarly. When the Pb and Cd increment in food homogenate due to Pb and Cd addition (A) was compared with that in deionized water (B), the recovery (A/B in %) was 90% (86–97%) for Pb and 102% (91–110%) for Cd (mean percentage recovery and the range in brackets).

The sources and the purities of the standard Pb and Cd solutions and mineral acids for digestion were as previously described (Watanabe et al. 1983, 1985a, 1996). Before use, both plastic containers and plastic- or glassware were washed with detergent,

Table 1 Measurements of standard reference samples for quality control (arithmetic mean ± SD; $n = 3$)

Metal	Reference sample	Unit	Certified values (A)	Values analysed in our laboratory (B)	B/A (%)
Pb					
	Total diet[a]	μg/g	0.05[d]	0.045 ± 0.006	90.0%
	Bovine liver[b]	μg/g	0.129 ± 0.004	0.118 ± 0.019	91.5%
	Human whole blood[c]	μg/100 ml	8.9 (7.6 − 10.3)	8.04 ± 0.14	90.3%
Cd					
	Total diet[a]	μg/g	0.028 ± 0.004	0.026 ± 0.002	92.9%
	Bovine liver[b]	μg/g	0.50 ± 0.03	0.54 ± 0.02	108.0%

[a] Standard reference material 1548, National Institute of Standards and Technology, USA
[b] Standard reference material 1577b, National Institute of Standards and Technology, USA
[c] Whole blood control (human), level 1-73011, Bio-rad, ECS Division Test Kits, Lyphochek, USA
[d] Not certified; listed for information only

immersed in dilute nitric acid for over 24 h and then rinsed with deionized water to make them metal leakage free. The preparation of deionized water was as described elsewhere (Watanabe et al. 1996).

Statistical analysis

Distributions of Pb and Cd in biological materials were log-normal rather than normal both in previous studies (Watanabe et al. 1983, 1985a, b; Ikeda et al. 1989) and in the present study (data not shown). Thus, dietary metal intakes and blood metal levels, and also metal intakes via boiled rice were all expressed in terms of a geometric mean (GM) and a geometric standard deviation (GSD). ANOVA, the multiple comparison test (Scheffe), the U-test and Student's t-test (unpaired) were used to detect any significant differences in means after logarithmic conversion of the measured values.

Results

Comparison of dietary intakes of Pb and Cd between Chinese and Japanese

Pb-F and Cd-F are shown by study site in Table 2. Women both in China and in Japan took a little more than 2000 g food daily, with no significant difference between two countries. GMs of Pb-F ranged from 17.0 μg/day (Shanghai) to 37.3 μg/day (Nanning) in Chinese populations and 9.3 μg/day (Tokyo) to 15.6 μg/day (Sendai) in Japanese. The grand GM for the total of 202 Chinese women (25.8 μg/day) was significantly ($P < 0.01$) higher than the grand GM for the 72 Japanese women (11.6 μg/day). The multiple comparison test made it clear that Pb-F in Tokyo (9.3 μg/day) was significantly ($P < 0.01$) lower than that in Beijing, Nanning and Tainan in China, and Kyoto in Japan.

In contrast to Pb, Cd-F among Japanese populations was significantly higher ($P < 0.01$, t-test) than that among Chinese populations (Table 2). Thus, Cd intake by Japanese is about three times that by Chinese. ANOVA followed by multiple comparison among the seven cities showed that Cd-F was about the same in all three Japanese cities and significantly ($P < 0.01$) higher than in the Chinese cities, except Nanning.

Among Chinese cities, Nanning had the highest Cd-F, with a value of 25.0 μg/day, which was significantly ($P < 0.01$) higher than the values for any of the other three Chinese cities, with no statistical difference ($P > 0.10$) from the levels in Japanese cities. Beijing and Shanghai had the lowest Cd-F levels, with no difference between them. Dietary intake in Tainan people was significantly different from that in the other six cities: higher than in Beijing and Shanghai ($P < 0.05$) and lower than in Tokyo, Kyoto, Sendai and Nanning ($P < 0.01$).

Comparison of Pb and Cd levels in blood between Chinese and Japanese

The trend in Pb-B was similar to that in Pb-F (Table 3), in which a grand GM for the Chinese women (56.7 μg/l) was significantly ($P < 0.01$) different from and about 1.8 times as high as the grand GM for Japanese women (32.1 μg/l). The highest Pb-B level was in Shanghai (although Pb-F was in the lower level) with a value of 79.0 μg/l, which was significantly higher than that in any of other six cities ($P < 0.01$) when assayed by a multiple comparison test. Unexpectedly, Pb-B in Kyoto (45.6 μg/l) was at the same level as in three Chinese cities, although lower than that in Shanghai.

Similar to Cd-F, levels of Cd-B (Table 3) in Japanese populations (1.92 μg/l) was nearly twice ($P < 0.01$) that in Chinese populations (1.07 μg/l). ANOVA and multiple comparison revealed that levels in Tokyo, Kyoto and Sendai were all significantly higher than in any of the four Chinese cities. Beijing (0.79 μg/l) was the lowest in Cd-B, which was also significantly lower there than in other Chinese cities. No differences were found among Shanghai, Nanning and Tainan or among the three Japanese cities ($P < 0.10$).

Table 2 Dietary intakes of Pb and Cd by Chinese and Japanese women (*GM* geometric mean, *GSD* geometric standard deviation, *ADI* acceptable daily intake, *PTWI* provisional tolerable weekly intake)

Site	Total food intake	Pb		Cd	
	AM[b] ± ASD[b]: *n*	GM[c] (GSD): *n*	%ADIs[d]	GM[c] (GSD): *n*	%ADIs[d]
China					
Beijing	2249 ± 408: 50	31.4 (2.31): 50	8.0	6.3 (1.73): 50	11.5
Shanghai	1698 ± 499: 50	17.0 (1.72): 50	4.3	6.1 (2.01): 50	11.1
Nanning	2332 ± 480: 50	37.3 (1.99): 50	9.5	25.0 (2.05): 50	45.5
Tainan[a]	2069 ± 550: 52	22.4 (1.92): 52	5.7	10.1 (1.70): 52	18.4
Total	2063 ± 558: 202	25.8 (2.12): 202**	6.6	9.9 (2.33): 202**	18.0
Japan					
Tokyo	2343 ± 363: 39	9.3 (4.39): 39	2.4	33.4 (2.08): 39	60.7
Kyoto	2479 ± 418: 17	14.6 (3.60): 17	3.7	37.0 (1.55): 17	67.3
Sendai	2411 ± 339: 16	15.6 (1.75): 16	4.0	24.8 (1.88): 16	45.1
Total	2390 ± 371: 72	11.6 (3.63): 72**	3.0	32.1 (1.93): 72**	58.4

** $P < 0.01$ for difference between China and Japan
[a] Cited from Ikeda et al. (1996)
[b] Grams per day
[c] Micrograms per day (GSD is dimensionless)
[d] Based on WHO/FAO's PTWIs for lead and cadmium of 50 and 7 µg/kg body weight/week (WHO/FAO 1972, 1989), respectively, the ADIs for a 55-kg woman was calculated as 393 µg/day for lead and 55 µg/day for cadmium

Table 3 Blood levels of Pb and Cd in Chinese and Japanese women

Site	Pb	Cd
	GM[a] (GSD): N	GM[a] (GSD): N
China		
Beijing	53.2 (1.41): 50	0.79 (1.54): 50
Shanghai	79.0 (1.50): 50	1.18 (1.40): 50
Nanning	56.0 (1.47): 50	1.25 (1.45): 50
Tainan[a]	44.5 (1.28): 52	1.11 (1.39): 52
Total	56.7 (1.50): 202**	1.07 (1.51): 202**
Japan		
Tokyo	30.6 (1.62): 39	1.82 (1.57): 39
Kyoto	45.6 (2.05): 17	1.99 (1.45): 17
Sendai	25.3 (1.48): 16	2.08 (1.84): 16
Total	32.1 (1.75): 72**	1.92 (1.60): 72**

** $P < 0.01$ for difference between China and Japan
[a] Cited from Ikeda et al. (1996)
[b] Micrograms per litre (GSD is dimensionless)

Contribution of boiled rice as the source of Pb and Cd

The role of boiled rice as the source of Pb and Cd was evaluated by comparing the Pb and Cd in boiled rice with Pb and Cd in total food duplicates, respectively. The results are summarized in Table 4. On average, the consumption of boiled rice was 389 g/day for Chinese and 317 g/day for Japanese, which accounted for about 18.8% and 13.3% (w/w) of the total daily food intake, respectively. Pb intake from boiled rice by Japanese (1.4 µg/day) accounted for 12.1% of Pb in the total food, and this account was over three times that among the Chinese subjects (0.9 µg/day, 3.6%), because total dietary Pb burden was smaller for Japanese (11.6 µg/day) than for Chinese (25.1 µg/day). The contribution of boiled rice as a source of Cd in total food was 31.1% for Chinese, and 32.7% for Japanese, the amount being almost the same, although Cd intake via rice among Japanese women (10.5 µg/day) was three times that among Chinese subjects (3.3 µg/day).

Discussion

The present study made it clear that the levels of lead and cadmium both in total daily food and in blood are quite different between Chinese and Japanese. In accordance with the results from other studies (Zheng 1984; Watanabe et al. 1985a, b, 1987b, 1989a, 1992, 1994, 1996; Qu et al. 1988, 1993; Tang et al. 1990; Xu 1994; Ikeda et al. 1996) comparison (Tables 2, 3) showed that Pb intake via food (25.8 µg/day) and Pb concentration in blood (56.7 µg/l) were significantly larger in Chinese women ($P < 0.01$ for both) than in Japanese women (11.6 µg/day via food and 32.1 µg/l in blood), whereas Cd intake via food (32.1 µg/day) and Cd in blood (1.92 µg/l) were significantly ($P < 0.01$) larger in Japanese than in Chinese (9.9 µg/day by food and 1.07 µg/l in blood, respectively).

For estimation of dietary intake of heavy metals there are several methods available, e.g., food duplicate collection, memory recall, market basket, among which the food duplicate method seems to be the most reliable (Louekari 1992). Nevertheless, studies on levels of dietary intake in the general Chinese population are limited, and none has been based on the food duplicate method. The reported values are: 86.3 and 57.5 µg Pb/day (AM) for adult men and children, respectively, 13.8 µg Cd/day (AM) for adult men (Chen and Gao 1993), 36.0–93.0 µg Cd/day for farming women

Table 4 Contribution of boiled rice as the source of lead and cadmium

Study site	Number of subjects	Pb intake (μg/day) Boiled rice (A)	Pb intake (μg/day) 24-h food (B)	A/B (%)	Cd intake (μg/day) Boiled rice (A)	Cd intake (μg/day) 24-h food (B)	A/B (%)
		GM (GSD)	GM (GSD)		GM (GSD)	GM (GSD)	
China							
Beijing	24	2.4 (2.21)	31.8 (3.12)	7.5	0.9 (2.37)	5.8 (1.87)	15.5
Shanghai	50	0.5 (3.21)	17.0 (1.72)	2.9	2.0 (2.08)	6.1 (2.01)	32.8
Nanning	50	2.4 (2.39)	37.3 (1.99)	6.4	9.6 (3.61)	25.0 (2.05)	38.4
Tainan	48	0.3 (2.55)	22.2 (1.95)	1.4	3.4 (2.02)	10.2 (1.73)	33.3
Total[a]	172	0.9 (3.74)	25.1 (2.21)	3.6	3.3 (3.46)	10.6 (2.42)	31.1
Japan							
Tokyo	39	0.6 (4.57)	9.3 (4.39)	6.5	11.1 (2.48)	33.4 (2.08)	33.2
Kyoto	17	4.9 (2.53)	14.6 (3.60)	33.6	10.6 (2.08)	37.0 (1.55)	28.6
Sendai	16	2.7 (2.41)	15.6 (1.75)	17.3	8.8 (3.00)	24.8 (1.88)	35.5
Total[b]	72	1.4 (4.71)	11.6 (3.63)	12.1	10.5 (2.48)	32.1 (1.93)	32.7

[a] Beijing, Shanghai, Nanning and Tainan combined
[b] Tokyo, Kyoto and Sendai combined

(Han 1988), and 47 μg Cd/day (AM) for adult (men and women combined) farmers (Chen 1994). It is difficult to compare these results with our present study results, partly because they were obtained with different study methods and at different study locations. It should be noted, however, that the values cited above were all 3–10 times those in the present study. It might be possible to analyse the causes of the observed difference if blood levels were also reported. Unfortunately, no blood data were available in these studies with the one exception of the study by Cai et al. (1990), in which a Cd intake of 64.9 μg/day (6 times that in the present study) was reported in combination with a blood Cd level of 1.34 μg/l as GM (a little higher than in the present study) in nonsmoking farming women.

In 1984, Zheng et al. (1984) reported the results of a large-scale study on blood Pb and Cd levels in the general populations of nine provinces and cities in China, following the guidelines of a WHO/UNEP programme (Vahter 1982). Most of the study participants were teachers of both sexes in high and elementary schools. The results based on 2017 Pb analyses and 2020 Cd analyses showed that the grand GM for Pb-B was 67 μg/l. The local GMs were in the range of 59–82 μg/l, one of the highest being in Shanghai. The GM for Cd-B in different provinces and cities varied from 0.4 to 1.3 μg/l (highest in Shanghai), with a grand GM of 0.79 μg/l. It should be noted that the smoking ratio has been high among men and low among women in general Chinese population. Therefore, when only women were selected, the grand GMs were lower, i.e., 59 μg/l (range 51–75 μg/l) for Pb-B and 0.69 μg/l (range 0.3–1.2 μg/l) for Cd-B. In a recent report (Xu 1994), the local GM for Pb in the blood of 7600 women in 28 cities was distributed over a range of 28–112 μg/l, with a grand GM of 60.6 μg/l. Thus, the present study results on Pb-B (56.7 μg/l) and Cd-B (1.07 μg/l) are comparable to the values reported by Zheng et al. (1984) and Xu (1994).

A number of studies have been conducted on dietary intake and blood levels of Pb and Cd in various sites in the world. The results on dietary intake and blood levels published in the literature since 1990 are summarized in Tables 5 and 6, respectively. For better comparison, attention was directed at women because all the subjects in the present study were women and it is known that both dietary intake and blood concentrations of metals differ between the two sexes (Watanabe et al. 1982, 1985a; Ikeda et al. 1989; Zheng 1984). Simple comparison of the Pb-F in the present study shows that the Chinese appear to be have one of the highest Pb-F values in Asia, in particular higher than that for the Japanese, whereas, in contrast, the Chinese fall in the group with the lowest values for Cd-F.

International comparison of Pb and Cd in blood suggests that the Pb-B in Chinese is the third highest (the Chinese have the second highest Pb-F), whereas for Cd-B the Chinese are in the intermediate group and this value is much lower than the values for Japanese populations.

FAO/WHO have established provisional tolerable weekly intakes (PTWIs) for Pb and Cd of 50 (FAO/WHO 1972) and 7 μg/kg body weight (FAO/WHO 1989), respectively. Thus, the recommended acceptable daily intakes (ADIs) for a woman weighing 55 kg (the average body weight of subjects in this study was about 55 kg, both for the total study population and for those of the individual cities) can be calculated as 393 μg/day for Pb and 55 μg/day for Cd. Table 2 summarizes the percentage daily intakes referred to the ADIs in each of the cities studied in the two countries. With a mean (GM) daily intake of 25.8 μg, corresponding to 7.2% of the ADI for Pb, the Pb intake by Chinese women is more than twice as high as that by

Table 5 Daily dietary intake of lead and cadmium found in food duplicate studies or estimated in women

Metal	Study site	Daily dietary intake (μg/day)	GM/AM	Reference
Lead				
	China	25.8	GM	Present study
	Finland	50.4 (52.9)[a]	GM (AM)	Louekari et al. (1991)
	Japan	11.6	GM	Present study
	Japan	8.4	GM	Watanabe et al. (1996)
	Korea	20.0	GM	Moon et al. (1996a)
	Malaysia	10.0	GM	Moon et al. (1996b)
Cadmium				
	China	9.9	GM	Present study
	Finland	14.2 (14.5)[a]	GM (AM)	Louekari et al. (1991)
	Germany	6.9 (8.8)[a]	GM (AM)	Müller and Anke (1994)
	Japan	32.1	GM	Present study
	Japan	29.9	GM	Watanabe et al. (1996)
	Korea	21.0	GM	Moon et al. (1996a)
	Malaysia	7.0	GM	Moon et al. (1996b)
	Sweden	8.3 (8.5)[a]	GM (AM)	Vahter et al. (1992)

[a] Estimated from the original AM (in the brackets), and ASD by the moment method (Sugita and Tsuchiya 1995)

Table 6 Comparison of lead and cadmium levels in blood in women

Metal	Study site	Metal levels in blood (μg/l)	GM/AM	Reference
Lead				
	China	56.7	GM	Present study
	China	72.0	GM	Qu et al. (1993)
	China	76.1	GM	Tang et al. (1990)
	China (Taiwan)	54.7	GM	Liou et al. (1996)
	Czechoslovakia	83.4	GM	Cikrt et al. (1993)
	Japan	32.1	GM	Present study
	Japan	31.8	GM	Watanabe et al. (1985b)
	Korea	44.0	GM	Moon et al. (1996a)
	Malaysia	46.0	GM	Moon et al. (1996b)
	Mexico	102.9 (120.0)[a]	GM (AM)	Rojas-Lopez et al. (1994)
	Sweden	22.8	GM	Lagerkvist et al. (1993)
	Switzerland	51.8	GM	Wietlisbach et al. (1995)
Cadmium				
	Belgium	0.70–1.25	GM	Sartor et al. (1992)
	China	1.07	GM	Present study
	China	0.83	GM	Qu et al. (1993)
	Czechoslovakia	0.99	GM	Cikrt et al. (1992)
	Japan	1.92	GM	Present study
	Japan	1.84	GM	Watanabe et al. (1993)
	Korea	1.30	GM	Moon et al. (1996a)
	Malaysia	0.70	GM	Moon et al. (1996b)
	Singapore	0.18–0.33	GM	Chia et al. (1994)
	Sweden	0.45	GM	Lagerkvist et al. (1993)

[a] Estimated from the original AM (in brackets), and ASD by the moment method (Sugita and Tsuchiya 1995)

their Japanese counterparts (3.2%) as a percentage of ADI, although both of them are much lower than the ADI for Pb. In contrast, the daily Cd intake by the Japanese population was about 64.2% of the ADI for Cd, or about three times that by the Chinese women (19.8%).

The respiratory and dietary routes are the only two routes for Pb and Cd intakes. Information on heavy metal concentrations in ambient air of Chinese cities is still scarce. Data available from the literature are compiled in Table 7. A simple comparison between China and Japan shows that Pb in air in China is about ten times that in Japan on average, whereas except in Beijing, the Cd concentration in air in China is lower than that in Japan. Based on these data in combination with the two assumptions that the respiratory volume is 15 m^3/day and that the absorption ratio is 50% both for Pb and Cd when inhaled (Ikeda 1996), it is possible to calculate that Chinese women will absorb about 2.33 μg Pb and 0.05 μg Cd daily via the respiratory route. Assuming that the absorption ratio in the gastrointestinal tract is 7.5% for both Pb and Cd (Ikeda 1996), daily dietary intakes of 25.8 μg Pb and 9.9 μg Cd among Chinese women will result in the uptake of 1.94 μg Pb and 0.74 μg Cd per day. Therefore, the total uptake of Pb and Cd via both inhalation and dietary routes in combination is 4.27 μg/day and 0.79 μg/day, respectively. Corresponding values for Japanese women are 0.27 μg Pb and 0.07 μg Cd/day via inhalation, 0.87 μg Pb and 2.41 μg Cd/day via ingestion, and 1.14 μg Pb (nearly one fourth of the intake by Chinese) and 2.48 μg Cd/day (over three times that in Chinese) via both routes in combination. When the amounts absorbed via each of the two routes of intake (i.e. via the respiratory route and via the dietary route) are compared it becomes obvious that Pb absorption from inspired air is substantial, especially in Chinese subjects (54.6% of the total burden versus 23.7% in Japanese women), whereas the Cd burden acquired via inhalation is essentially negligible for both Chinese and Japanese compared with that via ingestion (6.3% of the total burden in Chinese and 2.8% in Japanese; Table 7).

The question is, then, why Pb has been high both in foods and in blood and Cd has been low in China. Limited data published in the literature for a few cities (Zheng 1984; Cao et al. 1988; Wang 1992; Song et al. 1993; Xu 1994) showed that the Pb levels in air ranged from 0.18 to 0.66 μg/m^3, evidently a higher range than that (0.01–0.14 μg/m^3) in Japanese cities (Environment Agency 1995). Two major sources of exposure may be considered. One is the use of leaded gasoline in traffic, which has been reduced dramatically in Japan since 1969 (Kawada et al. 1994) but still persists in China even now, although the organic lead content in automobile gasoline was less than 1% in many cities in China even in the 1980s (Zheng 1984). Recent studies have shown that the changeover from leaded to unlead gasoline is the major cause of the decline in blood Pb (Wietlisbach et al. 1995; Strömberg et al. 1995).

Another source is the waste discharge from industries. It has been reported that, in China, 2918.6 tons of Pb were discharged into the air as industrial waste in 22 provinces in 1990 (Xu 1994), but only 101.6 tons of Pb into the industrial waste water in 50 major cities in

Table 7 Comparison of metal intake and uptake via respiratory (*UR*) and dietary (*UD*) routes in China and Japan

Metal	Site	Respiratory burden		Dietary burden		Total Uptake[k] (UR + UD)	UR/ (UR + UD) (%)[b]
		Air conc.[a]	Uptake[h] (UR)	Intake[i]	Uptake[j] (UD)		
		(μg/m^3)	(μg/day)	(μg/day)	(μg/day)		
Lead							
	China	0.31[b]	2.33	25.8	1.94	4.27	54.6
	Beijing	0.51[c]	3.83	31.4	2.36	6.19	61.9
	Shanghai	0.183[c]	1.37	17.0	1.28	2.65	51.7
	Nanning	–	–	37.3	2.80	–	–
	Tainan	0.32[d]	2.40	22.4	1.68	4.08	58.8
	Japan	0.036[e]	0.27	11.6	0.87	1.14	23.7
Cadmium							
	China	0.0073[f]	0.05	9.9	0.74	0.79	6.3
	Beijing	0.026[c]	0.20	6.3	0.47	0.67	29.9
	Shanghai	0.004[c]	0.03	6.1	0.46	0.49	6.1
	Nanning	0.0037[g]	0.03	25.0	1.88	2.18	1.4
	Tainan	–	–	10.1	0.76	–	–
	Japan	0.0095[e]	0.07	32.1	2.41	2.48	2.8

[a] Air concentrations of metals
[b] Geometric means of air concentrations of lead in Beijing, Shanghai and Tainan
[c] Cited from Cao et al. (1988)
[d] Cited from Ikeda et al. (1996)
[e] Cited from Environment Agency (1995)
[f] Geometric means of cadmium concentrations in air in Beijing, Shanghai and Nanning
[g] Cited from Cheng et al. (1987)
[h] Calculated assuming that the daily respiratory volume is 15 m^3 and that 50% of the metal element inhaled will be absorbed in the lungs
[i] Dietary intakes in geometric means
[j] Calculated on assumption that 7.5% of the element ingested will be absorbed in the gastrointestinal tract
[k] Sum of uptakes by respiratory and dietary routes
[l] UR/(UR + UD) in %; contribution of respiratory burden

1992 (Editorial Committee 1993). As for air pollution attributable to traffic, the total amount of lead discharged in automobile exhaust fumes in China per year remains unknown. Zhang et al. (1990) estimated that about 100 tons of tetraethyl lead (ca. 64 tons as Pb) were released into the air by the combustion of leaded gasoline in Taiyuan city, Shanxi Province every year. In the same period, only 10.8 tons and 22.1 tons of Pb were discharged in the whole province into water and air from industrial companies each year, respectively, which together accounted for only half the amount discharged from traffic in Taiyuan city (Xu 1994).

If all this is so, we can extrapolate from these values that 3,200 tons of lead would be discharged into ambient air in the 50 major cities a year (as 64 tons of Pb was discharged in a single city of Taiyuan), or more than 5,000 tons of Pb in 22 provinces (assuming that each province has 4 major cities). A comparison of the pollution resulting from industrial discharge, estimated at about 3,000 tons Pb as industrial waste in 22 provinces (Xu 1994), and the more than 5,000 tons Pb in automobile exhaust fumes suggests that the traffic will be a major source of air pollution by lead in China and that industrial discharge is a second important source of pollution.

This point, however, has been disputed. For example, Zheng (1984) and Wang (1992) believe that the industrial discharge rather than traffic is the leading source, because traffic is generally not heavy and the concentration of organic lead in gasoline is not high (e.g., less than 1%: Zheng 1984). The point apparently deserves further study. In this connection, people in Shanghai had the highest Pb-B levels in China (Table 3) although they had the lowest Pb-F (Table 2) and the Pb in air in Shanghai was also, contrary to expectations, reported to be the lowest (Cao et al. 1988). Such discrepancies should be investigated with reference to Pb in ambient air in such industrial cities as Shanghai.

Exposure of the general population to Cd. is mainly via food stuffs (Table 7), the Cd in food coming from the Soil. A large-scale investigation in China (Editorial Committee 1991) revealed that the background level of Cd in soil was 0.074 mg/kg as a GM, which was obviously lower than that in Japan, where 0.44 mg/kg Cd (as an AM) in soil has been reported (Iimura 1981). This may possibly explain the higher Cd burden via food among the Japanese population than among the Chinese population, although the sources of Cd in soil in Japan apparently still remain to be identified.

In summary, the results of the survey on dietary Pb and Cd intakes and that on Pb and Cd in blood agree with each other, indicating that exposure of the general urban population to Pb is higher in China than in Japan, whereas exposure to Cd is higher in Japan than in China. The levels of exposure to Pb and Cd are, however, well below the FAO/WHO-recommended tolerable intakes and do not appear to suggest any immediate health risk among the populations.

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