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Mercury in leaf litter in typical suburban and urban broadleaf forests in China

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Abstract

To study the role of leaf litter in the mercury (Hg) cycle in suburban broadleaf forests and the distribution of Hg in urban forests, we collected leaf litter and soil from suburban evergreen and deciduous broadleaf forests and from urban forests in Beijing. The Hg concentrations in leaf litter from the suburban forests varied from 8.3 to 205.0 ng/g, with an average (avg) of (49.7 ± 36.9) ng/g. The average Hg concentration in evergreen broadleaf forest leaf litter (50.8 ± 39.4) ng/g was higher than that in deciduous broadleaf forest leaf litter (25.8 ± 10.1) ng/g. The estimated Hg fluxes of leaf litter in suburban evergreen and deciduous broadleaf forests were 179.0 and 83.7 mg/(ha·yr), respectively. The Hg concentration in organic horizons (O horizons) ((263.1 ± 237.2) ng/g) was higher than that in eluvial horizons (A horizons) ((83.9 ± 52.0) ng/g). These results indicated that leaf litterfall plays an important role in transporting atmospheric mercury to soil in suburban forests. For urban forests in Beijing, the Hg concentrations in leaf litter ranged from 8.8–119.0 (avg 28.1 ± 16.6) ng/g, with higher concentrations at urban sites than at suburban sites for each tree. The Hg concentrations in surface soil in Beijing were 32.0–25300.0 ng/g and increased from suburban sites to urban sites, with the highest value from Jingshan (JS) Park at the centre of Beijing. Therefore, the distribution of Hg in Beijing urban forests appeared to be strongly influenced by anthropogenic activities.

Key words: mercury; broadleaf forests; leaf litter; soil; flux

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Introduction

Mercury (Hg) is a global contaminant, likely spreading across the world through long-range atmospheric transport. Every year, thousands of tons of Hg are emitted into the atmosphere from natural and anthropogenic sources. Atmospheric Hg may be deposited into a forest canopy in gaseous and aerosol forms, and deposited Hg^0 and reactive gaseous Hg (RGM) may be taken up by leaf stomata (Lindberg et al., 1979). Current research has identified that the strength of known atmospheric Hg sources is greater than previous estimations (Gustin et al., 2000; Engle et al., 2001; Pirroni et al., 2001), and vegetation is regarded as the missing sink in global Hg mass balance (Gustin et al., 2008). It is estimated that aboveground vegetation sequesters over 1000 tons Hg from the atmosphere every year (Obrist, 2007).

Litterfall is a major component of both energy flow and nutrient recycling in forest ecosystems. Global flux of Hg by way of litterfall is estimated to be 2400–6000 tons/yr, and is suggested to be the largest Hg flux to forest floors (Lindberg et al., 2004). Leaves account for 50%–84% of litterfall biomass (Silva-Filho et al., 2006) and their Hg concentrations are the highest among litterfall components (Nóvoa-Muñoz et al., 2008). Consequently, leaves are a significant determinant factor of litterfall flux and studies about the role of leaf litter on the Hg cycle in forests are necessary. Researchers from countries such as the United States of America (Rea et al., 1996, 2002; Sheehan et al., 2006), Canada (St Louis et al., 2001; Graydon et al., 2008), Sweden (Iverfeldt, 1991; Munthe et al., 1995), Spain (Nóvoa-Muñoz et al., 2008), and Brazil (Roulet et al., 1999; Fostier et al., 2003; Silva-Filho et al., 2006), have studied the role of litterfall in transferring atmospheric Hg to soils in boreal, temperate, and tropical forests. In China, Wang et al. (2009) and Fu et al. (2010) reported the Hg concentrations and fluxes of litterfall in southwestern subtropical forests. However, regional and even nationwide investigations on Hg concentrations in leaf litter and soil in Chinese forests have not been conducted.

Mercury in surface soil is an indicator of soil quality, while Hg in foliages at urban sites is an indicator of air quality (Barghigiani et al., 1991; Zhang et al., 2006). In addition, mercury in surface soil and foliage is a potential source of atmospheric and soil Hg, respectively (Lindberg et al., 2004). During the past three decades, Beijing has been undergoing fast economic development and urban construction. Soil and atmospheric Hg pollution has been

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studied in some urban areas (Liu et al., 2002; Wang et al., 2006; Zhang et al., 2006; Chen et al., 2010). It is also necessary, however, to study the distribution of Hg in leaf litter and soil in urban forests. Therefore, the goals of this study were to: (1) investigate the Hg concentrations in leaf litter and soil of evergreen and deciduous broadleaf forests in the suburban mountains of Chinese cities; and (2) investigate Hg distribution in leaf litter and surface soil within different forms of land use in Beijing urban forests.

1 Materials and methods

1.1 Site location

The study sites were divided into suburban and urban forests according to the influential extent of human activities. Suburban forests included both evergreen and deciduous broadleaf forests. Twenty-two sites were located at the suburban mountainous area of 19 cities in 8 provinces in China. Urban forest sites in Beijing were situated in the vicinity of landfills (outside the 5th Ring Road), universities, parks and streets, which contained 4–5 sites for each type of land use. The abbreviations and locations of these sites are presented in Table 1 and Fig. 1.

1.2 Sample collection

Samples of leaf litter and soils in eluvial horizons (A horizons) were collected from every suburban forest, and samples in organic horizons (O horizons) were also collected from some forests. The main tree species in the suburban evergreen broadleaf forests include *Castanopsis carlesii* (Hemsl.) Hayata, *Schima superba* Gardn. et Champ, *Castanopsis fargesii* Franch, *Cyclobalanopsis myrsinaefolia* (Bl.) Oerst, *Lithocarpus glaber* (Thunb.) Nakai, *Cinnamomum camphora* (Linn.) Presl and *Tsoongiodendron odorum* Chun. In deciduous broadleaf forests, the main tree species were *Fraxinus mandschurica* Rupr., *Populus davidiana* Dode, *Betula platyphylla* Suk., *Ulmus pumila* L., *Quercus liaotungensis* Koidz. and *Acer truncatum* Bunge.

In Beijing urban forests, broadleaf trees *Sophora japonica* L., *Populus tomentosa* Carr., *Ginkgo biloba* L. were selected. For comparison, coniferous trees *Sabina chinensis* L. and *Pinus tabulaeformis* Carr. were also selected.

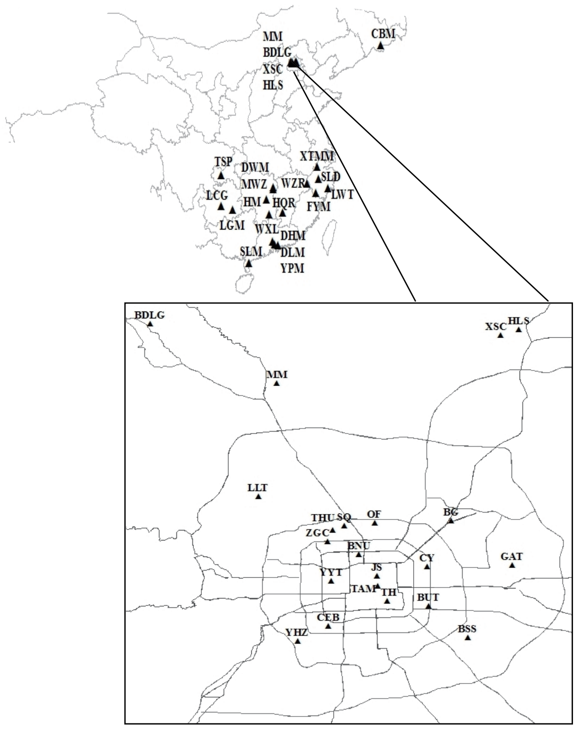

Fig. 1 Location of suburban forest sites in China and urban forest sites in Beijing. Abbreviations of sites are presented in Table 1.

Table 1 Abbreviations of suburban forest sites in China and urban forest sites in Beijing

Suburban forests in China				Urban forests in Beijing	
Forest type	Site	City/County	Province	Land use	Site
Evergreen broadleaf	Tieshanping (TSP)	Chongqing	Chongqing	Landfill	Gaoantun (GAT)
	Daling Mountain (DLM)	Dongguan	Guangdong		Beishenshu (BSS)
	Yinping Mountain (YPM)	Dongguan	Guangdong		Yonghezhuang (YHZ)
	Sanling Mountain (SLM)	Zhanjiang	Guangdong		Liulitun (LLT)
	Dinghu Mountain (DHM)	Zhaoqing	Guangdong	University	Tsinghua University (THU)
	Luchongguan (LCG)	Guiyang	Guizhou		
	Leigong Mountain (LGM)	Leishan	Guizhou		Capital University of Economics and Business (CEB)
	Maweizao Reservoir (MWZR)	Liuyang	Hunan		
	Daowu Mountain (DWM)	Liuyang	Hunan		
	Heng Mountain (HM)	Hengyang	Hunan		Beijing University of Technology (BUT)
	Wangxianling (WXL)	Chenzhou	Hunan		
	Hongqi Reservoir (HQR)	Ganzhou	Jiangxi		Beijing Normal University (BNU)
	Wangzhai Reservoir (WZR)	Shangrao	Jiangxi		
	Xitianmu Mountain (XTMM)	Hangzhou	Zhejiang	Park	Olympic Forest (OF)
	Shuanglongdong (SLD)	Jinhua	Zhejiang		Yuyuantan (YYT)
	Fengyang Mountain (FYM)	Longquan	Zhejiang		ChaoYang (CY)
	Longwantan (LWT)	Yongjia	Zhejiang		Temple of Heaven (TH)
Deciduous broadleaf	Changbai Mountain (CBM)	Antu	Jilin		Jing Shan (JS)
	Mang Mountain (MM)	Changping	Beijing	Street	Zhongguan Cun (ZGC)
	Badaling Greatwall (BDLG)	Yanqing	Beijing		Beigao (BG)
	Xisancun (XSC)	Huairou	Beijing		Shuangqing (SQ)
	Hongluosi (HLS)	Huairou	Beijing		Tiananmen (TAM)

Leaf samples were collected from five parallel individuals at each site. Surface soil (1–2 cm) was collected at each site from all four directions (north, south, west, and east) with a stainless steel shovel, and then was mixed together as the sample of one site.

Samples of leaf litter and soil were put in plastic zip-lock bags and transported to the laboratory immediately. Leaf litter was first rinsed by tap water, and then rinsed by ultra pure deionized water in laboratory. Leaf litter samples were dried at 55°C, ground, and then stored at –4°C for analysis. Soil samples were air-dried, ground, sieved and then stored at –4°C for analysis.

1.3 Sample analysis

The Hg concentrations in leaf litter and soil samples were measured by an RA-915$^+$ Hg analyzer attached with PYRO-91 thermal decomposition accessory (Lumex Inc., Russia). The samples were thermally decomposed in an atomizer chamber at 750°C with aided catalytic action, and then the RA-915$^+$ analyzer detected the Hg^0. Each sample was analyzed three times and then averaged. In this study, instrument calibration curves covering the appropriate concentrations were confirmed by soil standards (GBW07404, 590 ng/g) and peach foliage standards (GBW08501, 40 ng/g), and control standard samples were checked every ten samples. The instrument detection limit was 5 pg for solid samples. The precisions, obtained from ten replicated determinations of standards, were 3.47% for soil and 2.83% for peach foliage.

1.4 Data analysis

Analysis of variance (ANOVA) was performed using SPSS 11.5 for windows. Data were presented as mean and standard deviation, and statistical significance was considered at $p < 0.05$.

2 Results and discussion

2.1 Hg concentrations in leaf litter and soil in Chinese suburban forests

The average Hg concentration in evergreen broadleaf forest leaf litter ((50.8 ± 39.4) ng/g) was about two times the concentration in deciduous broadleaf forest litter ((25.8 ± 10.1) ng/g). This may be the result of a longer growth period for evergreen plants, with previous studies indicating that Hg concentrations in foliage increased with time (Ericksen et al., 2003; Millhollen et al., 2006). The Hg concentration at SLD ((119.9 ± 30.3) ng/g) was the highest among all the sites, with LCG ((91.8 ± 78.2) ng/g) presenting the second highest concentration (Table 2). In this study, the Hg concentrations in leaf litter in Chinese suburban forests were 8.3–205.0 (avg 49.7 ± 36.9) ng/g, within the range of 8–250 ng/g reported in the United States of America, Canada, Brazil, Sweden, and French Guiana (Table 3), but lower than those reported in Chinese southwest forests (Wang et al., 2009).

The soil Hg concentrations in A horizons and O horizons in Chinese suburban forests were 23.7–181.7 (avg 83.9 ± 52.0) ng/g and 95.3–860.7 (avg 263.1 ± 237.2) ng/g, respectively. Most soil Hg concentrations in the A horizons were lower than the Natural Standard Value for Soil Environment Quality I of Hg (150 ng/g, GB15618–1995) and concentrations in 11 sites were lower than the background soil Hg value (65 ng/g) in China (SEPAC, 1990), which indicated that most A horizon soils in the suburban forests were not polluted by Hg. The concentrations in the O horizons were 1.2–19.1 times those in the A horizons. Nóvoa-Muñoz et al. (2008) also reported Hg concentrations in O horizons (216.9–248.2 ng/g) higher than those in A horizons (108.1–195.0 ng/g) in Spain. Wang et al. (2009) reported that soil Hg concentrations decreased with soil depth in Chinese southwest forests. Larssen et al. (2008) reported the organic horizon and the

Table 2 Hg concentrations in leaf litter and soil at each suburban forest site

Evergreen broadleaf forest				Deciduous broadleaf forest			
Site	Leaf litter (ng/g)	Soil (ng/g)		Site	Leaf litter (ng/g)	Soil (ng/g)	
	mean ± SD (range)	O horizon	A horizon		mean ± SD (range)	O horizon	A horizon
DLM	22.0 ± 12.7 (13.0–31.0)	–	31.0 ± 4.2	CBM	27.0 ± 8.8 (21.5–40.0)	860.7 ± 24.2	45.0 ± 1.4
DWM	45.5 ± 18.3 (23.0–68.0)	221.0 ± 7.0	181.7 ± 9.0	MM	21.3 ± 8.6 (13.3–33.0)	–	63.0 ± 1.4
DHM	46.6 ± 10.8 (34.0–64.0)	–	170.5 ± 2.1	BDLG	31.7 ± 25.0 (14.0–49.3)	–	40.0 ± 2.1
FYM	30.7 ± 7.5 (22.0–35.0)	99.3 ± 5.5	66.5 ± 3.5	XSC	27.4 ± 9.0 (15.0–36.3)	–	32.0 ± 1.4
HM	68.8 ± 41.4 (32.0–125.0)	130.7 ± 0.6	95.7 ± 3.1	HLS	25.0 ± 9.3 (15.7–39.0)	–	37.3 ± 2.1
HQR	25.0 ± 10.4 (18.0–37.0)	–	58.5 ± 3.5				
LGM	69.6 ± 55.5 (25.0–187.5)	323.3 ± 17.6	177.7 ± 11.7				
LWT	57.4 ± 15.6 (31.0–69.0)	–	61.0 ± 2.6				
LCG	91.8 ± 78.2 (34.0–205.0)	283.3 ± 7.8	121.3 ± 8.4				
MWZR	57.0 ± 13.8 (41.0–69.0)	95.3 ± 1.5	23.7 ± 1.5				
SLM	31.5 ± 21.9 (16.0–47.0)	–	39.5 ± 4.9				
SLD	119.9 ± 30.3 (91.5–147.0)	–	71.5 ± 2.1				
TSP	41.3 ± 12.4 (29.0–57.0)	–	79.0 ± 1.4				
WXL	67.5 ± 21.1 (41.0–86.0)	182.3 ± 4.2	155.0 ± 2.0				
WZR	21.1 ± 13.1 (8.3–47.0)	–	150.5 ± 0.7				
XTMM	43.0 ± 49.2 (9.7–151.0)	172.3 ± 3.2	94.3 ± 0.6				
YPM	16.9 ± 5.9 (9.4–22.0)	–	51.5 ± 2.1				

– indicates no samples.

Table 3 Comparison of Hg concentrations and fluxes in different countries

Country	Hg concentration (ng/g)	Hg flux (mg/(ha·yr))	Reference
China	leaf litter: 8.3–205, 49.7 ± 36.9	leaf litter: 179.0, 83.7	This study
	litter: 104.8 ± 18.6–135.1 ± 31.7		Wang et al., 2009
America	green foliage: 34.2 ± 7.2, litter: 53.2 ± 11.4	litter: 130	Rea et al., 1996
	litter: 36 ± 8	litter: 158 ± 19, 114 ± 28	Rea et al., 2002
	litter: 31.6–58.8	litter: 100, 101	Sheehan et al., 2006
Canada	litter: 33–79	litter: 110–220	St. Louis et al., 2001
	litter: 28.9–69.2	litter: 86–105	Graydon et al., 2008
Sweden	litter: 97.4–140.6, 20–80	litter: 250	Iverfeldt, 1991
	litter: 33–140		Munthe et al., 1995
Brazil	green foliage: 40–152, leaf litter: 56–140		Roulet et al., 1999
	litter: 72–100		Fostier et al., 2003
	litter: 20–244	litter: 1220	Silva-Filho et al., 2006
French Guiana	leaves: 32.4–114	litter: 450 ± 100	Mélières et al., 2003

illuviation horizon have clearly higher Hg concentrations than do other horizons in mineral soil in Norway. However, Roulet et al. (1999) reported that the rapid turnover of organic matter at the surface of tropical soil in the Brazilian Amazonian ecosystems limited the accumulation of Hg in the organic horizon, which was much lower than that in the underlying mineral horizons. The present study showed that the Hg concentration in the O horizon at CBM (860.7 ng/g) was much higher than that in the other evergreen broadleaf forests. Slow decomposition of litter due to low temperatures at CBM led the Hg in litterfall accumulated in the O horizon. Guo et al. (2006) reported that 95% of leaf and twig decomposition takes about 4.5–8.0 and 7.8–29.3 years at CBM, respectively.

2.2 Distribution of Hg in leaf litter and surface soil in Beijing forests

The Hg concentrations in leaf litter in Beijing were 28.1 ± 16.6 (8.8–119.0) ng/g, nearly the same as those in suburban deciduous broadleaf forests ((25.8 ± 10.1) ng/g), but lower than those in suburban evergreen broadleaf forests ((50.8 ± 39.4) ng/g). This may ascribe that the tree species in Beijing are mainly deciduous, and their leaf growth periods are shorter than those of evergreen plants. The Hg concentrations in leaf litter did not correlate significantly with soil Hg concentrations for each species ($p > 0.05$), so Hg in leaves may originate mainly from the atmosphere. For *S. japonica*, the Hg concentrations at the streets, gardens, and universities were higher than those near landfills and suburban areas. The same results were found for *P. tomentosa*, *G. biloba*, *S. hinensis*, and *P. tabulaeformis* (Table 4). De Temmerman et al. (2007, 2009) showed that foliar Hg concentrations linearly correlate with atmospheric Hg concentrations, while Liu et al. (2002) showed that atmospheric Hg concentrations at the centre of Beijing (7.9–34.9 ng/m^3) are higher than those in suburban (5.3–12.4 ng/m^3) and rural (2.5–5 ng/m^3) areas. Therefore, the difference of Hg concentrations in leaf litter for each species might roughly indicate the atmospheric Hg pollution among different land use forms.

At most sites, the Hg concentrations in *S. japonica*, *P. tomentosa* and *G. biloba* were higher than those in *S. chinensis* and *P. tabulaeformis*. It appeared, therefore, that the leaves of broadleaf trees accumulated more Hg than those of coniferous trees. This may result from the fewer stoma in the leaves of coniferous trees.

The Hg concentrations in surface soil in Beijing varied from 32.0 to 25300.0 ng/g, with a median of 129.5 ng/g and an average of 1502.1 ng/g. Li et al. (2010) reported that the Hg concentration in Beijing topsoil ranges from 12.1

Table 4 Hg concentrations in surface soils and leaf litter at different land use types in Beijing

Type	Site	Soil (ng/g)	Avg (ng/g)	Leaf litter (ng/g)									
				S. japonica	Avg	*P. tomentos*	Avg	*G. biloba*	Avg	*S. chinensis*	Avg	*P. tabulaeformis*	Avg
Suburban	MM	63.0 ± 0.4	44.1	33.0 ± 1.4	28.3	22.0 ± 1.4	29.2	–	–	13.3 ± 0.5	14.7	16.7 ± 0.5	21.4
	XSC	32.0 ± 1.4		30.7 ± 0.5		36.3 ± 0.9		–		15.0 ± 0.8		27.7 ± 0.9	
	HLS	37.3 ± 1.7		21.3 ± 2.1		29.3 ± 2.1		–		15.7 ± 0.5		19.7 ± 0.5	
Landfill	GAT	89.0 ± 5.4	99.4	28.7 ± 0.5	33.1	37.0 ± 0.0	28.0	11.7 ± 1.2	12.6	16.3 ± 0.9	14.8	13.7 ± 1.2	13.8
	LLT	94.3 ± 17.6		69.0 ± 1.6		26.0 ± 1.4		–		23.7 ± 1.2		21.0 ± 0.8	
	BSS	101.0 ± 12.6		22.3 ± 0.5		22.0 ± 0.8		13.7 ± 0.9		8.8 ± 0.8		9.4 ± 0.4	
	YHZ	113.3 ± 11.8		12.3 ± 0.5		27.0 ± 1.6		12.3 ± 0.9		10.3 ± 0.5		11.0 ± 0.8	
University	THU	255.0 ± 49.3	384.6	67.0 ± 2.2	44.8	42.3 ± 1.7	36.8	23.7 ± 1.2	18.8	21.0 ± 0.8	17.3	57.3 ± 0.5	29.6
	CEB	308.7 ± 30.0		40.0 ± 2.2		35.0 ± 0.8		15.3 ± 0.9		15.0 ± 0.8		19.0 ± 0.8	
	BUT	442 ± 55.7		32.0 ± 0.0		36.0 ± 0.8		15.7 ± 0.5		17.3 ± 0.5		19.0 ± 0.0	
	BNU	532.7 ± 49.3		40.0 ± 0.0		33.7 ± 0.5		20.7 ± 0.5		15.7 ± 0.5		23.0 ± 0.8	
Park	OF	117.0 ± 16.3	5420.2	49.3 ± 0.9	45.3	28.7 ± 1.2	37.6	11.3 ± 0.5	25.5	13.0 ± 0.8	16.7	52.3 ± 2.1	28.9
	YYT	137.3 ± 8.8		39.0 ± 0.0		44.0 ± 0.0		27.7 ± 1.2		13.0 ± 0.8		21.0 ± 1.4	
	CY	250.0 ± 43.3		37.0 ± 0.8		47.7 ± 1.2		23.3 ± 1.2		12.0 ± 0.8		16.7 ± 0.9	
	TH	1296.7 ± 9.4		41.0 ± 0.8		33.0 ± 0.8		27.7 ± 1.7		20.3 ± 0.5		22.0 ± 0.8	
	JS	25300.0 ± 408.2		60.0 ± 0.0		34.7 ± 0.5		37.3 ± 2.6		25.0 ± 1.4		32.3 ± 2.1	
Street	ZGC	81.3 ± 10.7	217.9	42.3 ± 1.2	46.3	32.3 ± 1.2	34.0	12.3 ± 0.5	18.7	13.3 ± 0.9	20.8	20.0 ± 1.4	45.0
	BG	121.7 ± 10.7		39.3 ± 0.5		30.0 ± 0.8		21.3 ± 1.2		23.0 ± 0.0		21.3 ± 0.5	
	SQ	262.0 ± 15.1		75.0 ± 1.6		38.0 ± 1.6		26.7 ± 0.5		34.0 ± 0.8		119.0 ± 0.8	
	TAM	406.7 ± 27.4		28.3 ± 1.2		35.7 ± 0.5		14.3 ± 0.5		12.7 ± 0.5		19.7 ± 0.5	

to 8487 ng/g, lower than the maximal value of 25300 ng/g in this study, and the median Hg concentration is 471.3 ng/g, much higher than the median concentration of 129.5 ng/g in this study. Most Hg concentrations were lower than the Natural Standard Value for Soil Environment Quality II of Hg (1000 ng/g, GB15618–1995), except the soil Hg concentrations in the TH and JS Parks. The median Hg concentration of 129.5 ng/g in surface soil was 61.9% higher than the reported background concentrations in Beijing (80 ng/g, Environmental Monitoring of China, 1994).

The Hg concentrations in surface soil increased from suburban sites to urban sites, with the highest value in the JS Park at the centre of Beijing. Soil Hg concentrations with different types of land use followed the sequence: classical park (TH, JS) > university > street > vicinity of landfill > suburban. These results indicated that the Hg distribution in Beijing was influenced by anthropogenic activities. Li et al. (2010) reported that the median Hg concentration in different regions occurred in the following order: urban center soil > suburban soil > rural soil, and thought that the regional Hg distribution was closely related to anthropogenic and industrial activities. The Hg concentrations in classical parks were relatively higher than those in other land use forms, and also higher than those in parks in many other countries (Table 5). This may relate to the use of gold amalgamation or cinnabar as paint in classical parks. Chen et al. (2010) also reported soil Hg concentrations in three classical parks (Beihai, Zhongshan, TH) in Beijing as 9400, 6500 and 4200 ng/g, respectively. In this study, the Hg concentrations in universities were relatively high, as reported by Chen et al. (2010) who inferred that the use of Hg in medical and scientific apparatuses in cultural and educational areas was the main reason for this concentration. However, the Hg concentrations at the streets (except TAM) were not actually high. Chen et al. (2010) suggested that traffic was not a main source for Hg in roadside areas. The low Hg concentrations in the vicinity of landfills may ascribe to location, as all landfills were located outside the 5th Ring Road, far from urban areas.

2.3 Estimations of Hg fluxes of leaf litter in evergreen and deciduous broadleaf forests

In this study, the Hg flux of leaf litterfall was estimated by the biomass of leaf litter multiplied by the Hg concentration in leaf litter (Rea et al., 1996; Sheehan et al., 2006; Silva-Filho et al., 2006; Wang et al., 2009). Because of a lack of information on biomass of leaf litterfall in urban forests, we only estimated the fluxes in suburban evergreen and deciduous broadleaf forests. We averaged the data on biomass of leaf litter in evergreen and deciduous broadleaf forests reported by others, and obtained Hg fluxes of 179.0 and 83.7 mg/(ha·yr), respectively (Table 6). These fluxes were nearly at the same level of those reported in America, Canada and Sweden (Table 3), but lower than those reported in French Guiana (Mélières et al., 2003), and much lower than those reported in the Brazilian Amazon (Silva-Filho et al., 2006) (Table 3). The extremely high values at the two sites may be ascribed to the huge biomass of litterfall in tropical rainforests. The Hg fluxes of leaf litterfall in deciduous and evergreen broadleaf forests were nearly the same with a wet deposition flux of 84 mg/(ha·yr) and a dry deposition flux of 165 mg/(ha·yr) at CBM (Wan et al., 2009), respectively. Therefore, the input

Table 5 Soil Hg concentrations in urban forests at different countries

Site	Hg concentration (ng/g)	Reference
Beijing, China	32.0–25300, classical park: 1297–25300	This study
Beijing, China	16–966 (278)	Zhang et al., 2006
Beijing, China	22–9400, classical park: 1100–9400	Chen et al., 2010
Aveiro, Portugal	Park: 32–130 (59)	Rodrigues et al., 2006
Glasgow, England	Park: 310–5200 (1600)	Rodrigues et al., 2006
Ljubljana, Serbia	Park: 150–860 (410)	Rodrigues et al., 2006
Sevilla, Spain	Park: 110–1300 (420)	Rodrigues et al., 2006
Torino, Italy	Park: 210–900 (480)	Rodrigues et al., 2006
Uppsala, Sweden	Park: 15–1200 (350)	Rodrigues et al., 2006
Palermo, Italy	Park and greenland: 40–56000, media 1850	Manta et al., 2002

Table 6 Hg fluxes of leaf litter in evergreen and deciduous broadleaf forests

Broadleaf forest type	Biomass of leaf litter reported in China (kg/(ha·yr))	Average biomass (kg/(ha·yr))	Average Hg concentrations (ng/g)	Hg flux (mg/(ha·yr))	Reference
Evergreen	3764	3523.3	50.8	179.0	Hou et al., 1998
	2969.1				Lin et al., 1999
	3033, 3499				Yang et al., 2001
	4260				Guan et al., 2004
	1876.18, 3970.88				Chen et al., 2006
	3964				Wu, 2006
	1788.07				Fan et al., 2007
	2350				Wei et al., 2009
	4031.4, 6143.2, 4155.4				Pan et al., 2010
Deciduous	2549	3244.0	25.8	83.7	Zhang et al., 2008b
	4210, 3220, 3350				Zhang et al., 2008a
	2890.8				Liu et al., 2009

of Hg through leaf litterfall is important for the transport of atmospheric Hg to soil in Chinese forests.

The urban forest is an important forest type and is closely related with the daily life of humans. However, studies on litterfall in urban forests are limited, thus it is necessary to conduct research on the role of litterfall on the Hg cycle of urban forests.

3 Conclusions

In Chinese suburban forests, the Hg concentrations and fluxes in leaf litter in evergreen broadleaf forests were higher than those in deciduous broadleaf forests. The Hg concentrations in O horizons were higher than those in A horizons, and the Hg concentration in the O horizon at CBM was the highest among these sites. This data indicated that leaf litterfall is important for the transport of atmospheric Hg to soil in both evergreen and deciduous broadleaf forests.

The Hg concentrations in leaf litter at urban sites were higher than those at suburban sites for each plant. The Hg concentrations in surface soil in Beijing increased from suburban to urban areas, with the highest value at the center of Beijing. It therefore appears that the distribution of Hg in urban forests is strongly influenced by anthropogenic activities.

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