Chapter-27

MERCURY POLLUTION IN CHINA – AN OVERVIEW

Xinbin Feng

State Key Laboratory of Environmental Geochemistry, Institute of Geochemistry, Chinese Academy of Sciences, Guiyang 550002, P.R. China

MERCURY POLLUTION IN AQUATIC SYSTEM

Mercury pollution in Songhua River, Northeastern China

Songhua River, situated in Northeastern China, is one of the seven largest rivers in China and was seriously contaminated with mercury since the Acetic Acid Plant of Jilin Chemical Company went into operation in 1958. The plant, which was the largest producer of acetaldehyde in 1960s in China, utilized mercury sulfate as a catalyzer to manufacture acetaldehyde. The process is basically the same as that used once by the Chisso Company in Minamata, Japan, which eventually caused painful Minamata disease. Wastewater from the plant containing both inorganic and methylmercury was directly discharged into the Songhua River. The old technique was completely substituted by a new one without utilization of mercury in 1982. From 1958 to 1982, the plant discharged in total113.2 t total mercury and 5.4 t methyl mercury to the Songhua River, which constituted 69.8% and 99.3% of total anthropogenic Hg-tot and methyl mercury input to this river, respectively. In addition to this contamination source, some small plants which also used mercury, such as a chloroethylene plant from Jilin Chemical Company, Jilin Dye Plant, Changchun Meteorological Instrument Plant, a few gold mining companies using the amalgamation method to extract gold,

and a few chlor- alkaline plants, are distributed inside the drainage area of the Songhua River and also discharged mercury into the river.

A few, large research projects related to integrated control and countermeasure of methylmercury pollution of the Songhua River were carried out between 1980s and 1994. The results from these research projects were published in two books (Liu, 1994; Liu et al., 1997) and in a number of papers in Chinese journals (Zhang et al., 1994; Wang and Qi, 1984; Guo, 1990; Zhang et al., 1993; Hou et al., 1994; Yu et al., 1994; Zhai et al., 1991; Yu et al., 1994; Lin, 1995; Zhang and Li, 1994; li et al., 2001; Wang et al., 1986; Zhang et al., 1985; Wang et al., 1985).

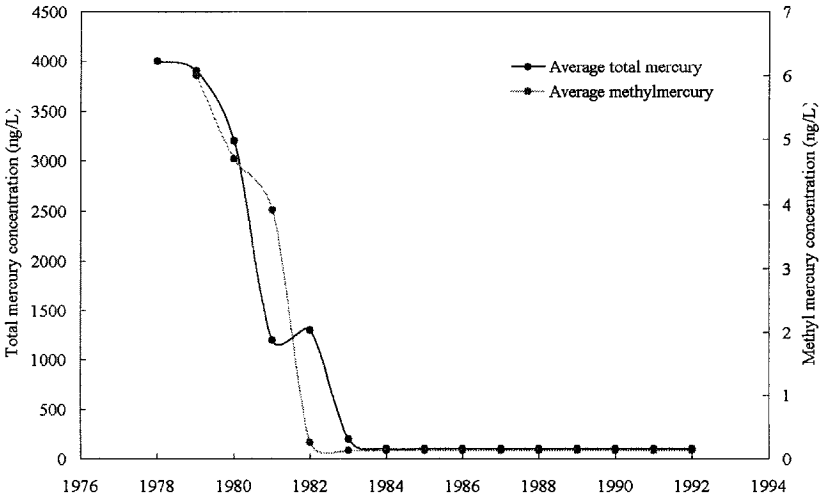

Figure 1. Total and methyl mercury distributions in river water from Jilin City to Shanzakuo section from 1978 to 1992 (adopted from Yu et al., 1994).

After the largest mercury emission source, the acetic acid plant of Jilin Chemical Company, completely terminated discharge of mercury to the Songhua River in 1982, total mercury and methylmercury concentrations in river water decreased significantly as shown in Figure 1 (Yu et al., 1994).

Total mercury concentrations in surface sediment from different sampling sites of the Songhua River in 1973 and 1986 were compared, as shown in Figure 2.

Even though the total mercury concentrations in surface sediment declined significantly after the major mercury discharge source to the river was completely stopped, mercury concentrations in sediment at certain

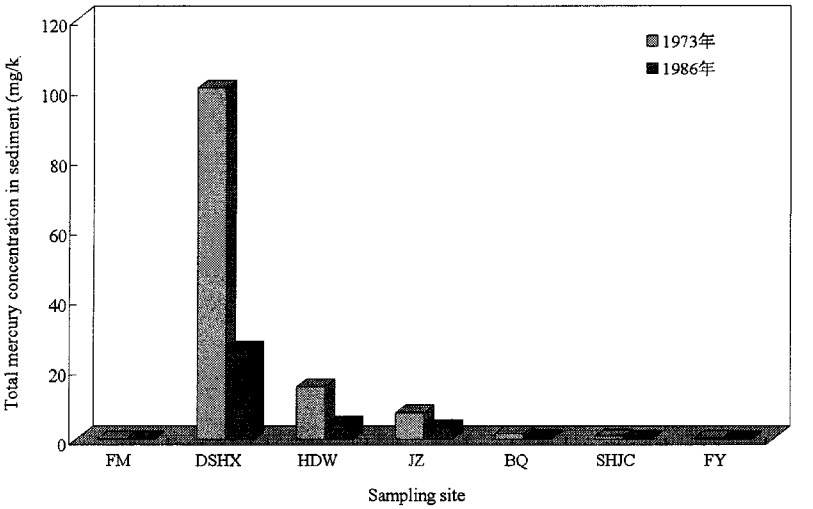

Figure 2. Comparison of mercury concentrations in sediments between samples collected in 1973 and those in 1986. FM: Fengman; DSHX: Dongshihaoxian; HDW: Hadawan; JZ: Jiuzhan; BQ: Baiqi; SHJC: Songhuajiang Chun; FY: Fuyu (Adopted from Zhang et al., 1994).

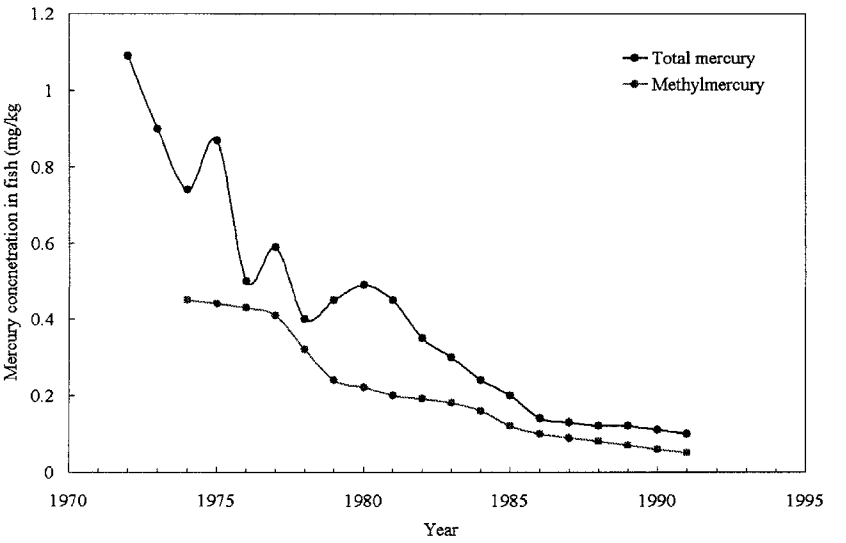

Figure 3. Temporal changes of mercury concentration in fishes from 1972 to 1991 (from Yu et al., 1994).

highly contaminated sites were still elevated compared to the background mercury concentration in sediment of the Songhua River, which is 0.14 mg/kg (Zhang et al., 1994). Both average total mercury and methyl mercury concentrations in fish from the Songhua River were monitored from 1974 to 1991 (Yu et al., 1994).

It is also shown that mercury concentrations in fish decreased gradually after mercury discharge to the river was completely stopped in 1982, as shown in Figure 3.

The health impacts of methylmercury pollution in the Songhua River to local fishermen and inhabitants were intensively investigated (Lin, 1995; Zhang et al., 1993; Hou et al., 1994; Zhai et al., 1991; Li et al., 2001). Lin (1994) systemically reviewed the research progress on the health impact to the local fishermen from methylmercury pollution in the Songhua River from 1970s to late 1980s. The average mercury concentrations in hair of fishermen reached 13.5 mg/kg in 1970s, and some fishermen showed the symptoms of Minamata disease, including concentric constriction of the visual field, loss of sensation in hands and feet, hearing impairment, and ataxia. A survey carried out in 1997 showed that the average total mercury concentration in hair of fishermen dropped to 1.8 mg/kg (Li et al., 2001), and it is demonstrated that the threat of health impact to local fishermen from methylmercury pollution in Songhua River dramatically declined since the major pollution source stopped discharging mercury to the river.

Mercury pollution to Jiyun River, Eastern China

The Jiyun River, located in Tianjin, eastern China, was seriously contaminated with mercury since the 1960s when a Chloralkali plant started to operate at the middle reaches of the river. Before 1977, mercury contaminated wastewater from the plant had been directly discharged into the river. After 1977, a mercury control project went into operation and mercury concentration in wastewater decreased tremendously (Zhang et al., 1981). Many studies (Zhang et al., 1981; Lin et al., 1983, 1984; Lin and Kang, 1984; Kang, 1986; Peng et al., 1983; Yang et al., 1982) were carried out during late 1970s and early 1980s to elucidate mercury distribution and speciation in water and sediment, and the mercury methylation rate in the sediment of this river. It was revealed (Zhang et al., 1981) that methylmercury concentrations in water and sediment varied from below detection limit to 64 ng/L and from below detection limit to 0.35 mg/kg, respectively, and that total mercury concentrations in river water and sediment ranged from 20 to 24,000 ng/L, from 0.03 to 845 mg/kg,

respectively. Unfortunately, no studies on mercury concentration in fish from Jiyun River have been reported.

Mercury pollution to other aquatic systems

The Guizhou Organic Chemical Factory (GOCF), a state-owned company, has been producing acetic acid from acetaldehyde synthesized by addition reaction of acetylene and water using mercury as a catalyst. The production of acetic acid in the plant continued from 1971 to 2000 in the eastern urban section of Qingzhen City, Guizhou Province, Southwestern China. The total loss of mercury into the environment was 134.6 t for the 30 year period. The wastewater from the plant was directly discharged into the Dongmenqiao River, which is the water source for paddy rice fields of about 120 ha farmland. Yasuda et al. (2004) investigated mercury distribution in the farmland, and the average total mercury concentration is 15.73±42.98 mg/kg with the maximum concentration of 321.38 mg/kg. GOCF is located about 10 km upstream of Baihua Reservior, which was dammed in 1966. Mercury contaminated wastewater finally reached the reservoir through the Dongmenqiao River. Total mercury concentrations in the sediments of the reservoir ranged from 5 to 40 mg/kg, and total mercury concentrations in water varied from 25.5 to 70.0 ng/l (Yan et al., 2003 and Yan et al., unpublished data). Since the bedrocks of the drainage areas of the reservoir are limestone and dolomite, it is an alkaline reservoir and the pH of the water is around 8. Though the reservoir is seriously contaminated with mercury, total and methylmercury concentrations in fish are usually very low and less than 0.15 mg/kg (wet weight). A research project is on-going in our research group to elucidate the biogeochemical cycling of mercury in the reservoir (e.g. Feng et al., 2004b).

An et al. (1982) reported mercury pollution from the Yanguoxia Chloralkali Plant in Yongjing County, Ganshu Province, Northwestern China. The total mercury concentrations in the air of the workshop varied from 23 to 111 $\mu g\ m^{-3}$.Total mercury concentrations in hair of workers from the chloralkali plant ranged from 6.1 to 202.8 mg/kg and averaged out to 45.4 mg/kg, which were significantly elevated compared to the average total mercury concentration of 0.25 mg/kg in hair from people at a reference site. The plant discharged approximately 4.8 t mercury to the Yellow River each year during late 1970s, and total mercury concentrations in river water of the Yellow River within 14 km down reach of the pollution source varied from 0.04 to 3480$\mu g\ m^{-3}$. Mercury concentrations in wheat grain cultivated around the chloralkali plant varied from 0.02 to 0.11 mg/kg and averaged at 0.068

mg/kg, which already exceeded the national advisory limit of mercury in food, which is 0.02 mg/kg. No further report on mercury pollution status in this area was found from the open literature since 1982. Zhang (1993) reported fish mercury concentrations from the Yellow river in the section of Ningxia Province, which is downstream of Ganshu Province. Total mercury concentrations in fish varied from 0.09 to 0.90 mg/kg in 1982 and 1983, and ranged from 0.45 to 0.98 mg/kg in 1988. Thirty-two percent of the fish in theYellow River in Ningxia section had mercury concentrations exceeding the national advisory limit for fish, which is 0.3 mg/kg in 1982, while 100% of fish exceeded the limit in 1988. Though other mercury pollution sources could not be ruled out, the Yanguoxia Chlor alkali Plant may be responsible for the elevation of mercury in fish in the Ningxia section of the Yellow River.

Table 1. The statistical data of mercury production, imports and exports, and total consumption from 1980 to 1995.

Year	Total national production (ton)	Total import (ton)	Total export (ton)	Total consumption (ton)
1950-79	31432		17156	14276
1980	894		985	
1981	826		946	
1982	912		476	
1983	895		300	320
1985	800			
1990	930			
1991	768	200	63	905
1992	580	538	52	1066
1993	515	344	53	806
1994	466	909	48	1327
1995	779	799	16	1562

ANTHROPOGENIC MERCURY EMISSION INVENTORY IN CHINA

China is one of the largest mercury production and consumption countries, as shown in Table 1. Before 1991, China was a large mercury exporter, but after that became a large mercury importer because the resources in most Chinese mercury mines were exhausted, on the one hand,

and on the other hand, the national demand for mercury has been increasing since the early 1990s when artisanal gold mining activities started.

Asia, especially China, has been regarded as the largest atmospheric mercury emission source from a global perspective (Pirrone et al., 1996, Pacyna and Pacyna, 2002). There is, however, tremendous uncertainty on mercury emission inventory estimates for Asia, especially for China because of lack of direct measurement data to establish reliable emission factors for different anthropogenic sources. The efforts to establish an accurate anthropogenic mercury emission inventory in China are underway (Feng and Hong, 1997; Feng, 1997; Qi, 1997; Feng and Hong, 1999; Wang et al., 2000; Feng et al., 2002). In this section, an estimation of anthropogenic mercury emissions in China in 1995 is made, depending on recent domestic measurements and literature data.

Table 2. Emission factors and total mercury emissions from different anthropogenic sources.

Source category	**Emission factor**	**Total Hg emission (tonnes)**
1. Coal combustion	0.12 (g/T)	145
2. Non-ferrous metal production		27
-Primary Cu	10 (g/T)	6
-Primary Pb	3 (g/T)	2
-Primary Zn	20 (g/T)	19
3. Pig iron and steel production	0.04 (g/T)	8
4. Cement	0.1 (g/T)	48
5. Gold extraction		107
-Large scale	0.68 (g/g)	21
-Artisinal	15 (g/g)	86
6. Hg mining	45 (g/kg)	35
7. Chlor-alkali production	18 (g/T NaOH)	2
8. Battery, electrical light, thermal meter (do you mean thermometer?)	5% mercury used	20
9. Others		10
Total		**402**

Emission factors of different sources

The major anthropogenic mercury emission sources are categorized as: coal combustion; copper production; lead production; zinc production; pig iron and steel production; cement production; gold extraction; mercury

mining; chlor-alkali production; battery, electrical light, thermometer production; and other sources. Emission factors from different sources are listed in Table 2. The emission factor from coal combustion is synthesized from the literature (Feng, 1997; Qi, 1997; Feng et al., 2002; Wang et al., 2000; Zheng et al., 2004). There is large uncertainty on the estimation of the mercury emission factor from coal combustion in China. Wang et al. (2000) reported an emission factor of 0.177 g/T based on the assumption that average mercury concentration in Chinese coal is 0.22 mg/kg. An intensive investigation of mercury concentrations in Chinese coals was carried out in collaboration between the Institute of Geochemistry, the Chinese Academy of Sciences and the US Geological Survey, and it was revealed that the average mercury concentration in Chinese coal is about 0.15 mg/kg (personal communication with Prof. Zheng Baoshan, 2004), which is much lower than the value used by Wang et al. (2000). Therefore we recalculated the mercury emission factor, which is 0.12 g/T, as listed in Table 2.

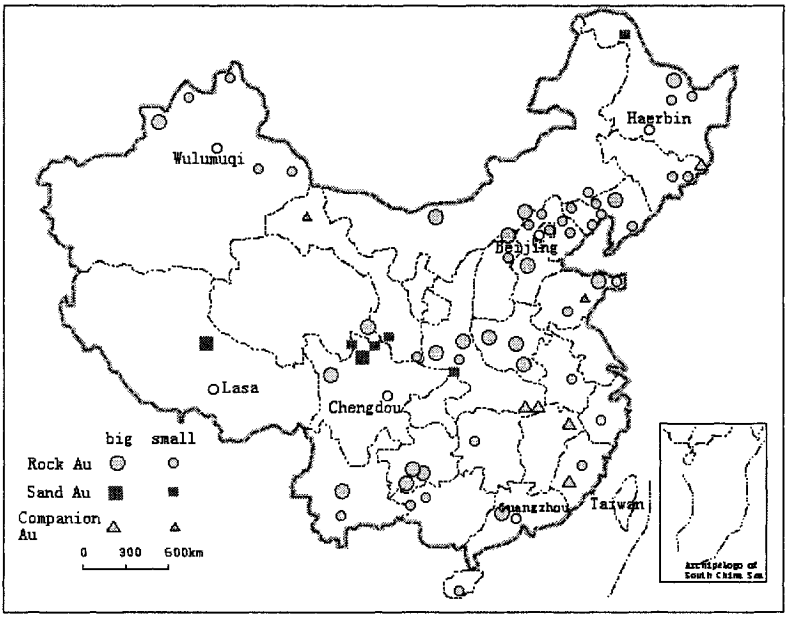

Figure 4. Distribution of gold mines in China.

Gold mining is also one of the most important atmospheric mercury emission sources in China. Gold mines are distributed in most provinces as shown in Figure 4. It is estimated that about 1/3 of annual gold production in

China was produced using the mercury amalgamation technique during the1980s and 1990s. In the last decade, many surveys were conducted to investigate health impacts to gold miners from different provinces (Wang et al., 1999; Xi et al., 1996; Wang, 2001; Duan et al., 2001; Zhong and Lou, 2000; Guan et al., 2002; Zhang et al., 1999), but only a few studies regarding mercury pollution to the local environment were carried out (Lin et al., 1997; Dai et al., 2003). There are basically two different techniques regarding gold extraction using mercury in China, namely large scale mining and small scale, or artisanal mining. For large scale mining activities, mercury is well recovered, and the emission factor was estimated to be 0.68 g Hg/ g gold produced (Qi, 1997). However, mercury is poorly recovered in the artisanal mining process, and the mercury emission factor is estimated to be 15 g Hg/ g gold produced (Qi, 1997). The annual production of gold from artisanal mining generally constituted about 16% of gold produced using amalgamation techniques during the 1980s and 1990s. From 2000, artisanal gold mining activities were officially banned, but a few illegal artisanal mining workshops are still operating in some remote areas.

Battery production is an important mercury consumption industry, and Table 3 lists the annual estimated mercury consumption in battery production from China in 1992-1999. It is rather difficult to estimate the mercury emission rate from battery, light and thermometer production, and Qi (1997) estimated that, in general, 5% of the used mercury will be emitted to the air. Mercury emission factors from mercury mining and chloralkaline plants were adopted from Qi (1997), and the emission factors from other sources, such as non-ferrous metal production, pig iron and steel production, and cement production are adopted from Pacyna and Pacyna (2002).

Table 3. Estimated mercury consumption (ton) in battery production in China in 1992-1999.

Battery type	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**
Ordinary Zn-Mn	24.2	32.4	26.1	37.3	21.2	23.2	21.3	12.7
Alkaline Zn-Mn	10.3	18.7	24.4	55.2	76.0	137.6	199.1	325.8
Hg Oxide	150.2	300.5	344.3	369.3	237.9	494.5	570.0	463.2
Total	184.7	351.6	394.8	461.8	355.1	655.3	790.4	801.7

Estimates of total mercury emissions in 1995

The 1995 Chinese emissions of total mercury from anthropogenic sources are presented in Table 2. About 402 metric tons of total mercury were emitted in 1995. Coal combustion (including power plant and domestic

uses) is the largest source, which accounted for 36.1% of total anthropogenic emissions. Meanwhile, our mercury group is investigating mercury speciation in flue gases in coal combustion using the Ontario Hydro method, which will result in a more precise mercury emission factor from coal combustion in China. All those efforts will definitely improve the precision of estimating mercury emissions from coal combustion in China in the near future.

Gold extraction using mercury may also be an important atmospheric mercury emission source, especially artisanal gold mining activity (Lin et al., 1997). Fortunately the artisanal gold mining activities were completely banned by the government recently, which will significantly reduce mercury emissions from gold extraction from now on. In addition, cement and non-ferrous metal production are still important atmospheric mercury emission sources.

Waste incineration was not employed in China to dispose of waste in 1995, though some metropolitan areas, such as Shanghai, Beijing and Guangzhou, are now using this technology to dispose of municipal waste. The total amount of waste incinerated is, however, still limited in China. Thus, mercury emission from waste incineration, one of the major atmospheric mercury emission sources in western countries, was ignored in our estimation. So far, landfills have been the major way to dispose of municipal waste in China. Though studies showed that landfills are important atmospheric mercury emission source (Lindberg and Price, 1999), we do not yet have enough information to estimate the emission rate from this source category. A research project is, however, on-going in our group to study mercury emission flux from landfills in China, and hopefully the knowledge gap will be bridged soon.

Uses of mercury in chloralkali production, in battery production, and in production of measuring instruments and electrical lights were not banned but decreased in China until 2002, which is in contrast with Western countries, where uses of mercury in these industries have already ceased. However, emissions of mercury from these sources were quite low compared to emissions from other sources.

MERCURY CONCENTRATIONS IN AMBIENT AIR OF URBAN AREAS IN CHINA

Using an acidified, sulfurized cotton trap to pre-concentrate mercury coupled with CVAFS detection method, Yu (1985) for the first time

measured total gaseous mercury (TGM) concentrations in ambient air at some tourist attraction areas in Beijing. TGM concentration varied from 1 to 87 ng m^{-3}, and at most sampling sites, the average TGM concentration exceeded 20 ng m^{-3}. Quality assurance (QA) and quality control (QC) information of the method applied is, unfortunately, not available from his work. Using an automated mercury vapor analyzer (Tekran 2537A), Liu et al. (2000) monitored TGM concentrations at one industrial, two urban, three suburban, and two rural sampling locations in January, February and September of 1998 in Beijing, China. In the three suburban sampling stations, mean TGM concentrations during the winter sampling period were 8.6, 10.7, and 6.2 ng m^{-3}, respectively. In the two urban sampling locations, mean TGM concentrations during winter and summer sampling periods were 24.7, 8.3, 10, and 12.7 ng m^{-3}, respectively. In the suburban-industrial and the two rural sampling locations, mean mercury concentrations ranged from 3.1-5.3 ng m^{-3} in winter to 4.1-7.7 ng m^{-3} in summer. It is clear that TGM concentrations in ambient air of Beijing are elevated compared to the global background values that are believed to be within the range from 1.0 to 1.5 ng m^{-3}. Wang et al. (1996) investigated preliminarily TGM concentrations in ambient air in metropolitan Chongqing and its suburb and TGM concentrations varied from 9.2 to 101.5 ng m^{-3} with an average of 34.4 ngm^{3}.

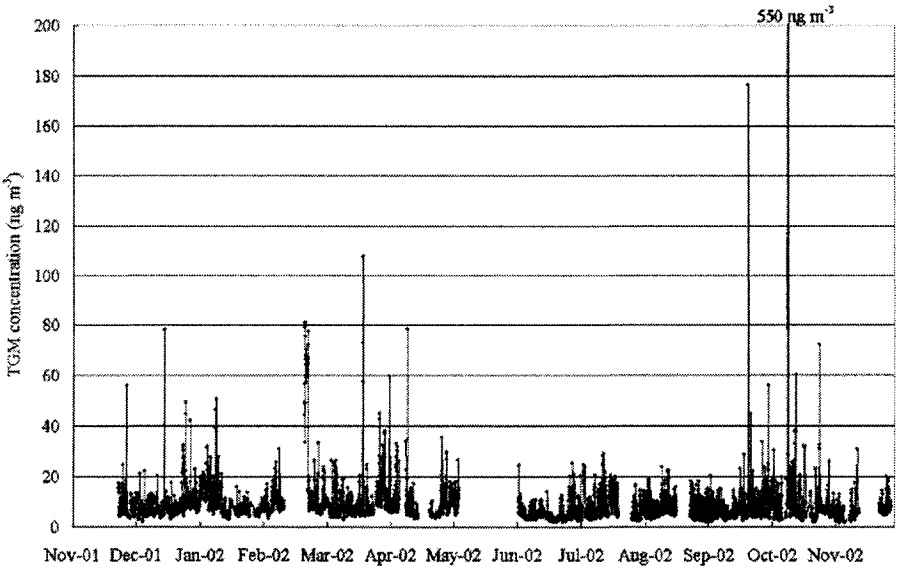

Figure 5. Hourly averaged total gaseous mercury (TGM) concentration in the air of Guiyang, China from November 23,2001 to November 30, 2002 (From Feng et al., 2004).

TGM concentrations in ambient air were occasionally monitored at an urban site in Guiyang between 1996 and 2000 (Feng et al., 2002; 2003). The sampling site where the Institute of Geochemistry, Chinese Academy of Sciences is located is a dense residential area, and a number of industries are located southwest of the sampling site within 20 km.

The average TGM concentration in the ambient air was 11±4 ng m^{-3} in December 1996, and was 13±9 ng m^{-3} in October 1999 (Feng et al., 2002). Four measurement campaigns were carried out to monitor TGM at this site in the following periods: April 19-30, 2000; February 26- March 14, 2001; June 26- July 20, 2001 and October 9 – November 22, 2001, respectively. High temporal resolved data were obtained by using automated mercury analyzers Gardis 1A and Tekran 2537A.

The geometric means of TGM from different seasons were 8.56, 7.45, 5.20 and 8.33 ng m-3 in spring 2000, winter 2001, summer 2001 and fall 2001, respectively (Feng et al., 2003).

A more intensive TGM measurement campaign was carried out at this sampling site from 23 November 2001 to 30 November 2002 using a high temporal resolution 5 min mercury vapor analyzer (Tekran 2537A) (Feng et al., 2004a). A total of 77,541 individual data points were collected and the hourly averaged TGM concentrations over the sampling period are depicted in Figure 5.

TGM concentrations follow lognormal frequency distribution pattern and the mean TGM concentration is 8.40 ng m^{-3} on the basis of one year observation. The elevation of TGM in Guiyang is attributed to coal combustion from both industrial and domestic uses. A seasonal distribution pattern of TGM, with a descending order of winter, spring, fall and summer, was observed. The highest TGM concentration in winter is attributed to household heating using coal.

Only a few data are published on reactive gaseous mercury (RGM) concentrations in ambient air in China. Using the KCl coated tubular denuder technique (Feng et al., 2000), Feng et al. (2002) carried out a short-term measurement campaign in Guiyang in October 1999. The average RGM concentration was 453.8 pg m^{-3}. Shang et al. (2003) also conducted a short-term measurement campaign at the same sampling site in March 2002 using the same technique and the average RGM concentration was 37.5 pg m^{-3}. Coal combustion in Guiyang was demonstrated to be RGM emission source (Tang et al., 2003), and these limited data showed that RGM concentrations in ambient air of Guiyang were significantly higher than the background RGM concentrations < 10 pg m^{-3} measured in Northern America and Northern Europe (Lindberg et al., 1998; Sommar et al., 1999). The significant discrepancy of RGM concentrations between the two campaigns

is mainly attributed to the weather conditions. It is dry season in October and it is relatively wet season in March in Guiyang. Since RGM consists of water soluble mercury species such as $HgCl_2$ and $Hg(NO_3)_2$, the lifetime of RGM in ambient air is strongly dependent on the relative humidity. Therefore, the dry season favors the retention of RGM in the air, and RGM concentration could be very high since there are RGM emission sources in the city.

Fang et al. (2001) studied total particulate mercury (TPM) concentrations in ambient air at five sampling sites representing the tourism district, an ordinary industrial district, a scattering heating residential district, a special industrial district and a reference area in Changchun City, Northeastern China from July 1999 to January 2000. The daily average TPM concentrations ranged from 22 to 1984 pg m^{-3}. A significant correlation was obtained between the TPM concentrations and the total suspended particulate concentrations. It is demonstrated that coal combustion and wind-blown soil material are the two main sources of particulate mercury in Changchun City. Wang et al. (2002) reported TPM measurement data at three sampling sites in Beijing, and TPM concentrations ranged from 360 to 440 pg m^{-3}. Duan and Yang 1995 reported TPM concentrations in ambient air at six sampling locations in Lanzhou City, Northwestern China. TPM concentrations varied from 100 to 1000 pg m^{-3}, and a clearly seasonal TPM distribution pattern was obtained showing that TPM concentrations during winter and summer are usually higher than during spring and fall. Again, the authors claimed that coal combustion emissions and wind blown dust were the main sources of particulate mercury in Lanzhou City. The limited reported data showed that TPM concentrations in urban ambient air in China are much higher than that of the background value of 1 to 86 pg m^{-3} (Keeler et al., 1995).

MERCURY POLLUTION TO SOIL COMPARTMENT

A survey conducted by the China National Environmental Monitoring Center 1990 showed that the national background mercury concentration in soil is 0.038 mg/kg. Zhang and Zhu 1994 reported that the average mercury concentration in soil from Tibet, which is the less impacted by human activities, is 0.022 mg/kg. Figure 6 shows the distribution of mercury in soil over the whole of China (China National Environmental Monitoring Center, 1994).

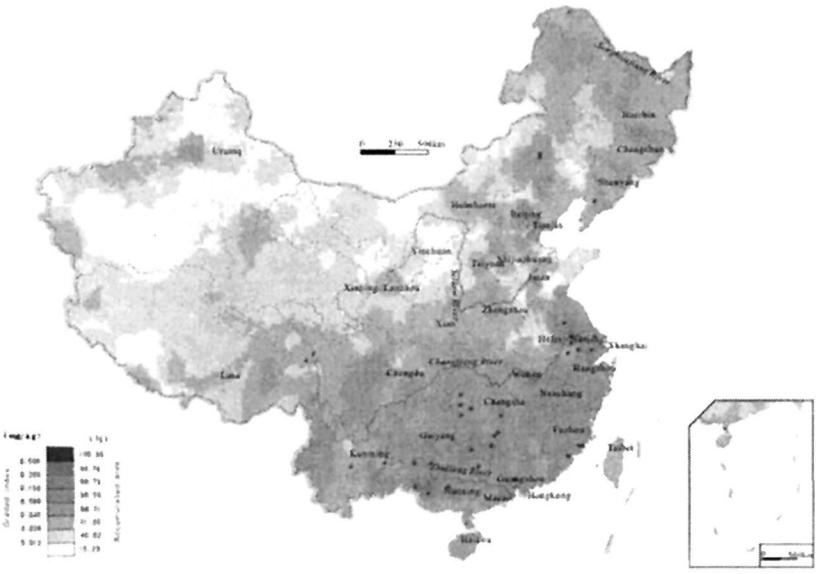

Figure 6. Distribution of mercury in surface soil of China (From China National Environmental Monitoring Centre, 1994).

We can see that mercury concentrations in soil are not evenly distributed over China, and in certain areas, such as southwestern and southern China, mercury concentrations are quite elevated. These elevations are mainly related to the geological background since the Global Circum- Pacific Mercury Belt crosses these areas. Of course, mercury emissions from human activities also caused the elevation of mercury concentrations in urban soils.

Tao et al. (1993) investigated mercury concentrations in soil from Shenzhen area, South China. The average mercury concentration in soil is 0.068 mg/kg, which is significantly higher than the national background value of < 0.01 mg/kg. The utilization of fertilizers containing high mercury during the 1970s is believed to be the main cause of the elevation of mercury in soil (Tao et al., 1993). Of course, the deposition of atmospheric mercury emitted from industrial activities could not be totally ruled out. Guo et al. (1996) reported that average mercury concentration in soil from Taiyuan City is 0.110 mg/kg, and Wang et al. (2003) attributed the elevation of mercury in soil to industrial emissions.

MERCURY POLLUTION IN GUIZHOU

Guizhou is located in Southwestern China with an area of 170,000 km^2, accounting for about 1.8% of the total area of China. It is known as the "mercury capital" of China because more than 60 % of total national mercury resources were discovered in this province. Guizhou is located in the Global Circum-Pacific Mercury Belt, and at least 13 large and super large-scale mercury mines have already been discovered in the province. As shown in Figure 5, the background mercury concentrations in soil from this area are very elevated. Meanwhile, Guizhou is one of the largest coal producing provinces and is also one of the areas where acid deposition occurs frequently due to coal combustion emissions. Mercury concentrations in coal from Guizhou are significantly elevated compared to the average mercury concentration in Chinese coal due to low-temperature thermal fluid activities in the Mesozoic and Cenozoic Eras in this area (Feng et al., 2002). Mining of mercury and other metals, coal combustion, and other human activities significantly polluted the environment with mercury. Tremendous national and international attention has been given to mercury pollution in the province (Horvat et al., 2003; Xiao et al., 1998; Tan et al., 2000; Yasuda et al., 2004; Feng et al., 2002, 2003, 2004a, 2004b, 2004c).

A study conducted by Xiao et al. (1998) showed that total mercury deposition flux in Fanjing Mountain Nature Reserve (FMNR) area in the Northeast of Guizhou, which is one of over 3000 nature reserves established in the world, was calculated to be 115 $\mu g\ km^{-2}\ y^{-1}$. The deposition flux is very elevated compared to the estimated total deposition rate over Southwestern Sweden, which is about 40 $\mu g\ km^{-2}\ y^{-1}$, implying that mercury pollution in the province is even affecting the biogeochemical process in FMNR (Xiao et al., 1998). Tan et al. 2000 monitored total mercury deposition fluxes at 12 sampling sites in the province, and the total deposition fluxes ranged from 336 to 2340 $\mu g\ km^{-2}\ y^{-1}$. TGM concentrations in ambient air measured at these sites varied from 2.7 to 12.2 $ng\ m^{-3}$ (Tan et al., 2000). Horvat et al. (2003) assessed the level of contamination with mercury in two geographical areas of Guizhou province. Mercury pollution in the areas concerned originate from mercury mining and ore processing in the area of Wanshan, while in Qingzhen, mercury pollution originates from GOCF, as stated in section 1.3. The results of this study confirmed high contamination of Hg in soil sediments and rice in the mercury mining area in Wanshan, and high levels of mercury in soil and rice were also found in the vicinity of GOCF. Mercury contamination in Wanshan is geographically more widespread due to deposition and scavenging of Hg from contaminated

air and deposition on land. In Qingzhen, Hg contamination of soil is very high near the chemical plant, but the levels reach background concentrations at a distance of several km. Even though the major source of Hg in both areas is inorganic mercury, it is observed that active transformation of inorganic Hg to organic Hg species (MeHg) takes place in water, sediments and soils. The concentration of Hg in rice grains can reach up to 569 μg/kg of total Hg, of which 145 μg/kg was in MeHg form. The percentage of Hg as MeHg varied from 5 to 83%. It was concluded in this study that the population mostly at risk is located in the vicinity of smelting facilities, mining activities and close to the waste disposal sites in a wide area of Wanshan.

Feng et al. (2003b) investigated the status of mercury pollution in the groundwater systems in the Wanshan area and found that total mercury concentrations in river waters ranged from 29.7 to 585.8 ng/L. The mercury mining wastes and processing residues are the mercury contamination sources to surface water systems. Feng et al. (2004c) estimated mercury emissions from artisanal zinc smelting using indigenous methods in Hezhang, Guizhou, and mercury emission factors were estimated to be 154.7 and 78.5 g Hg t^{-1} of Zn produced from sulfide ore and oxide ore, respectively. These emission factors are much higher than the literature value used to estimate mercury emissions from zinc smelting in developing countries, which is 25 g Hg t^{-1} of Zn produced. Annual mercury emission rates from zinc smelting in Hezhang from 1989 to 2001 are listed in Table 4. The local surface water, soil and crops were also contaminated with mercury due to zinc smelting activities.

CONCLUSIONS

Mercury pollution to aquatic systems from acetic acid plants and chloralkaline plants, both of which use mercury is an environmental concern in China, and some of the plants are still in operation. The remediation of mercury polluted aquatic systems is an imperative task, and the biogeochemical cycling of this metal and its health impacts to local inhabitants need to be scrutinized as well.

Total anthropogenic mercury emissions in 1995 in China was estimated to be 402 tons. Coal combustion, gold mining and cement production are the most important mercury emission sources.

Table 4. Annual zinc production and mercury emission from artisanal zinc smelting in Hezhang.

Year	Zn production (t)	Hg emission (Kg)
1989	7610.5	1119
1990	11289	1660
1991	11639	1712
1992	11989	1763
1993	17403	2560
1994	23453	3449
1995	26731	3932
1996	23038	3388
1997	22700	3339
1998	31100	4574
1999	45200	6648
2000	48098	7074
2001	32700	4809

Mercury emission from coal combustion is projected to be increasing due to the increasing need of energy with the rapid economic increase since 1995. On the other hand, mercury emission from gold mining has been decreasing since the late 1990s because artisanal gold mining, which is one of the largest mercury emission sources, was totally banned then. Cement production is, and will still be, an important mercury emission source in China. Of course, there are large uncertainties on the estimation of mercury emissions from anthropogenic sources in China because the emission factors from most of sources were obtained only by indirect methods, such as using literature values obtained from other countries. More studies are needed to conduct intensive field measurement of not only total mercury emissions, but also mercury speciation from different mercury emission sources to precisely estimate mercury emissions from anthropogenic sources in China.

From limited data, we can conclude that mercury concentrations in urban air are very elevated. More atmospheric mercury measurement data, especially data from remote background areas in China, are needed to correctly understand the global cycle of mercury in the troposphere. Soils from urban areas are usually contaminated with mercury deposited after being emitted to the air from industrial processes, which poses a threat to terrestrial ecosystems. Mercury pollution in Guizhou is a regional, environmental issue. The biogeochemical cycle of mercury at the regional scale there, and its environmental and health risks, need to be studied.

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