# Chapter 3
# Mercury Emissions from Industrial Sources in China

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**Summary** In this chapter, we reviewed mercury emissions from industrial sources in China. The industrial sources included fuel oil for stationary sources; gasoline; diesel and kerosene; biofuel combustion; Grassland/ savannah burning; waste and residue burning; cement production; iron and steel production; caustic soda production; non-ferrous metal smelting (Zn, Pb, Cu, and Au); mercury mining; and battery and fluorescent lamp production. Mercury emission factors from most source categories were obtained according to measurement data from Europe and North America. The mercury emission factor for zinc smelting, which was believed to be the largest industrial source, was adopted from the data of recent studies in China. We used the information published in the literature to estimate the emission of different mercury species. The total mercury emission from industrial sources in China was 253.07 Mg in 1999. Non-ferrous metal smelting (including zinc, lead, copper and gold smelting) is the largest industrial mercury emission source in China and the total mercury emissions reached 167.8 Mg. The total mercury emissions from industrial sources in China in 1995 was 296.4 Mg, increasing to 360.5 Mg in 2003, at an average annual growth rate of 2.90%. Due to lack of field measurement data to quantify mercury emission factors for most of industrial sources, a large uncertainty is associated with the current emission inventory. A number of studies need to be undertaken to reduce the uncertainties. Surveys are needed to evaluate mercury contents in raw materials of different industrial categories. Mercury balance studies are necessary for representative plants of different industrial sources. The speciation of mercury emissions from different industrial sources are also urgently needed in order to better understand the atmospheric fate of mercury emitted from these sources.

## 3.1 Introduction

Global emissions of anthropogenic mercury to the atmosphere have been estimated to be 1900 Mg in 1995 (Pacyna and Pacyna, 2002), of which 77% was from coal combustion, with the remainder divided among non-ferrous metals production, cement production, and waster disposal. However, the scenario of mercury emissions from China may be quite different. Mercury emissions from industrial sources

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other than coal combustion contribute a significant portion of the total emissions (Streets et al., 2005; Wu et al., 2006).

China has been regarded as one of the largest atmospheric mercury emission sources from a global perspective (Pirrone et al., 1996; Pacyna and Pacyna, 2002) due to rapid economic development. The increasing mercury emissions in China have resulted in the elevation of mercury concentrations in ambient air both in urban and rural areas in China. Average concentrations of total gaseous mercury (TGM) in Guiyang, Guizhou Province, have been measured in the range of 5–15 ng $m^{-3}$ (Feng et al., 2002; 2003; 2004a; 2004b), attributed to uncontrolled coal-burning in the residential and industrial sectors. In Beijing, Liu et al. (2002) measured TGM concentrations in the range of 6–10 ng $m^{-3}$ during winter. Fang et al. (2001) measured average particulate mercury concentrations of about 0.5 ng $m^{-3}$ in North eastern Changchun City, Jilin Province, rising to as high as 2 ng $m^{-3}$ during the heating season. Xiu et al. (2005) measured somewhat lower levels of Hg in Total suspended particulate (TSP) in Shanghai, in the range of 0.2–0.5 ng $m^{-3}$. The average TGM concentrations in rural areas such as the Gongga Mountain area in South western China (Fu et al., 2008) and the Changba Mountain area in North eastern China (Wan et al., 2008) reached 3.98 and 3.22 ng $m^{-3}$, respectively, which were significantly elevated compared to the values of 1.5 to 2.0 ng $m^{-3}$ measured in rural areas in Europe and North America (Ebinghaus et al., 2002; Schroeder et al., 2001; Lindberg et al., 2007). These measurement data demonstrated that mercury emissions from anthropogenic activities have resulted in elevated TGM concentrations in ambient air in China. There are, however, tremendous uncertainties in mercury emission inventory estimates for China simply because of the lack of direct measurement data to establish reliable emission factors for different anthropogenic sources. Currently, the mercury emission factors from different anthropogenic sources are generally adopted from the data obtained from the studies conducted in Europe and North America. However, mercury measurements of major emission sources in China are known to have been taken or are underway, even if the data are, as yet, unpublished and unavailable, e.g., the cement plant measurements made by US-EPA in cooperation with Chinese entities and others, and power plants measurements made by US-DOE and Chinese universities and authorities. In Chapter 2 of this report, Streets et al. (2008) reviewed mercury emission from the coal combustion sector in China, while mercury emissions from industrial sources will be evaluated in this chapter.

The major industrial mercury emission sources in China include fuel oil for stationary sources; gasoline; diesel and kerosene; biofuel combustion; Grassland/ savanna burning; waste and residue burning; cement production; iron and steel production; caustic soda production; non-ferrous metal smelting (Zn, Pb, Cu, and Au); mercury mining; and battery and fluorescent lamp production.

A model has been developed to calculate mercury emissions from different industrial sources in China (Streets et al., 2005; Wu et al., 2006). The basic concept of the mercury emissions calculation is described by the following equation:

$$E_t = \sum\sum\left[ef_{i,j,t} A_{iij,j,t} F_{RELj,t} \left(1 - F_{REMj,t}\right)\right] \quad (1)$$

where $E_t$ is the mercury emission; $ef_{i,j,t}$ is emission factor for other fuels or non-combustion processes; $A_{i,j,t}$ is the amount of fuel consumption or production yield of non-combustion processes; $F_{RELj,t}$ is the fraction of mercury emitted to the atmosphere; $F_{REMj,t}$ is the fraction of mercury removed by emission control devices; j is the type of combustion with/without control devices; i is the province; and t is the year.

## 3.2 Emission Factors from Different Industrial Sources in China

Table 3.1 lists mercury emission factors from various industrial activities in China and the majority of the data are adopted from Streets et al. (2005). Mercury emission factors from the zinc smelting sector which is currently believed to the one of the largest

**Table 3.1** Emission factors for total Hg from industrial sources in China

| Source category | Unit | Emission factor |
|---|---|---|
| 1. Fuel oil for stationary sources (e.g., power plants, industrial use) | $g\ Mg^{-1}$ oil | 0.014[a] |
| 2. Gasoline, diesel, and kerosene | $g\ Mg^{-1}$ oil | 0.058[a] |
| 3. Biofuel combustion | $g\ Mg^{-1}$ biofuel | 0.020[b] |
| 4. Grassland/savanna burning | $g\ Mg^{-1}$ grass burning | 0.080[c] |
| 5. Waste and Residue Burning | | 0.037[d] |
| Agricultural residue | g $Mg^{-1}$ residue 0.037[d] | 2.80[e] |
| Household waste | g $Mg^{-1}$ waste 2.80[e] | |
| 6. Cement production | $g\ Mg^{-1}$ cement | 0.040[f] |
| 7. Iron and steel production | $g\ Mg^{-1}$ steel | 0.04[g] |
| 8. Caustic soda production | $g\ Mg^{-1}$ caustic soda | 20.4[h] |
| 9. Non-ferrous metal smelting | | |
| Zinc (Zn) | g $Mg^{-1}$ Zn | 5.7-155[i] |
| Copper (Cu) | g $Mg^{-1}$ Cu | 9.6[j] |
| Lead (Pb) | g $Mg^{-1}$ Pb | 43.6[j] |
| Gold (Au): large-scale production | $Mg\ Mg^{-1}$ Au | 0.79[j] |
| Gold (Au): artisanal production | $Mg\ Mg^{-1}$ Au | 15.0[k] |
| 10. Mercury mining | $kg\ Mg^{-1}$ Hg | 45.0[k] |
| 11. Battery and fluorescent lamp production | $Mg\ Mg^{-1}$ Hg used | 0.05[k] |

[a] From US EPA (1995).
[b] From Friedli et al. (2003a).
[c] Average emission factor for forests is 0.113 g $Mg^{-1}$ (Friedli et al., 2003b). We assume that grasslands are generally like forests in terms of long-term exposure to Hg, but with typically rather shorter lifetimes for Hg uptake. This value is therefore lowered to 0.080 g $Mg^{-1}$ for grassland burning.
[d] From Friedli et al. (2003b).
[e] From UNECE/EMEP (2004).
[f] Coal related Hg emissions for cement production are excluded from this category. Energy intensity of 0.196 Mg of coal $Mg^{-1}$of cement produced (Zhou et al., 2003) is used here to adjust emission factor of 0.065 g $Mg^{-1}$ of cement (US EPA, 1997) to 0.040 g $Mg^{-1}$ of cement produced.
[g] From Pacyna and Pacyna (2002).
[h] From Qi et al. (2000).
[i] From Li (2007) and Feng et al. (2004).
[j] From Jiang (2004).
[k] From Qi (1997).

mercury emission sources in China is adopted from recent studies by Li (2007) and Feng et al. (2004). It is obvious that no field measurement data in China regarding mercury emission studies is currently available in the open literature for most source categories. Therefore, mercury emission factors from most source categories were obtained according to measurement data obtained in Europe and North America.

Zinc smelting processes in China can be divided into two major types, namely the pyro-metallurgic process (PMP) and the electrolytic process (EP). The pyro-metallurgic process can then be divided into four sub-types, such as the imperial smelting process (ISP), retort zinc smelting process (RZSP), electric zinc furnace (EZF), and artisanal zinc smelting process (AZSP). Total zinc production in China reached 2.71 million Mg in 2004. It is estimated (Jiang, 2006) that 71.8% of total zinc production is based on EP technology, 7.7%, 5.9%, 13%, and 1.6% of total zinc production used ISP, RZSP, EZF and AZSP techniques, respectively.

Using the mass balance method, Feng et al. (2004) calculated mercury emission factors from artisanal zinc smelting using both oxide and sulphide ores and the data are shown in Table 3.2. Applying the same method, Li (2007) investigated mercury emission factors from four large scale zinc smelters using EP with flue gas mercury removal devices and without flue gas mercury removal devices, RZSP and ISP techniques, respectively and one artisanal zinc smelter using oxide ores and the emission factors are listed in Table 3.2. We can see that mercury emission factors varied significantly with different smelting processes and decreased dramatically if mercury removal devices were applied for the smelters. Streets et al. (2005) and Wu et al. (2006) used an averaged emission factor of 86.6 g $Mg^{-1}$ Zn to estimate mercury emission from zinc smelting sources. From Table 3.2, we can clearly see that in most cases mercury emission factors were much lower than the value that is currently used. Therefore, we have applied these mercury emission factors in Table 3.2 to estimate mercury emissions from different zinc smelting factories using different zinc smelting processes in China in this paper.

**Table 3.2** Mercury emission factors from zinc smelting using different smelting processes in China

| Methods | $EF_{Hg}(\mu g\ Mg^{-1})$ | Information Source |
|---|---|---|
| Artisanal Zn Smelting using oxide ore | 75 | Li, 2007 |
| Artisanal Zn Smelting using oxide ore | 79 | Feng et al., 2004 |
| Artisanal Zn Smelting using sulfide ore | 155 | Feng et al., 2004 |
| Electrostatic Process (EP) with mercury removal device | 5.7 | Li, 2007 |
| Electrostatic Process (EP) without mercury removal device | 54 | Li, 2007 |
| Retort Zn Smelting Process RZSP) | 34 | Li, 2007 |
| Imperial Smelting Process (ISP) | 122 | Li, 2007 |

## 3.3 Speciation of Mercury Compounds from Different Industrial Sources in China

Primary emissions are classified according to gaseous elemental mercury ($Hg^0$), divalent gaseous mercury ($Hg^{(II)}$), and particulate mercury (Hg(p)). Generally no field measurement data on the speciation of mercury from the industrial sources is available from open literature. Streets et al. (2005) and Wu et al. (2006) used measurements from Pacyna and Pacyna (2002), and Friedli et al. (2001, 2003a, b) for different industrial sources as shown in Table 3.3.

## 3.4 Emissions from Different Industrial Sources in China in 1999

Streets et al.(2005) estimated anthropogenic mercury emissions in China for the year 1999 and the total emission from industrial sources other than coal combustion reached 327.95 Mg. We recalculated the total mercury emissions from industrial sources according to the emission factors listed in Table 3.2 and the total mercury emission was 253.07 Mg as shown in Table 3.4. The difference between these data sets is from the estimate of mercury emission from zinc smelting. We used the new mercury emission factors to estimate mercury emissions from zinc smelting factories using different processing techniques according the studies by Li (2007) and Feng et al. (2004).

It can be seen that non-ferrous metal smelting (including zinc, lead, copper and gold smelting) is the largest industrial mercury emission source in China and the total mercury emissions reached 167.8 Mg, which constituted 66% of the total

**Table 3.3** Speciation of total mercury for each major source type (as fraction of the total)

| Source category | $Hg^0$ | $Hg^{(II)}$ | Hg(p) |
|---|---|---|---|
| 1. Fuel oil for stationary sources (e.g., power plants, industrial use)[a] | 0.50 | 0.40 | 0.10 |
| 2. Gasoline, diesel, and kerosene combustion[a,b] | 0.50 | 0.40 | 0.10 |
| 3. Biofuel combustion[b] | 0.96 | 0.00 | 0.04 |
| 4. Grassland/savanna burning[b] | 0.96 | 0.00 | 0.04 |
| 5. Waste and residue burning[b] | 0.96 | 0.00 | 0.04 |
| 6. Cement production[a] | 0.80 | 0.15 | 0.05 |
| 7. Iron and steel production[a] | 0.80 | 0.15 | 0.05 |
| 8. Caustic soda production[a] | 0.70 | 0.30 | 0.00 |
| 9. Non-ferrous metal smelting[a] | 0.80 | 0.15 | 0.05 |
| 10. Mercury mining[c] | 0.80 | 0.15 | 0.05 |
| 11. Battery and fluorescent lamp production[c] | 0.80 | 0.15 | 0.05 |

[a.] From Pacyna and Pacyna (2002).
[b.] From Friedli et al. (2001, 2003a, 2003b).
[c.] Assumed to be the same profile as other non-combustion sources.

**Table 3.4** Summary of Hg emission estimates (*Mg*) for industrial sources associated with fuel consumption and materials production and use in 1999

| Source category | Fuel consumption or material yield *(Mg)* | Hg | $Hg^0$ | $Hg^{(II)}$ | Hg(p) |
|---|---|---|---|---|---|
| Fuel oil for stationary sources | $33.8 \times 10^6$ [a] | 0.47 | 0.24 | 0.19 | 0.05 |
| Gasoline, diesel, and kerosene | $96.8 \times 10^6$ [a] | 5.61 | 2.81 | 2.25 | 0.56 |
| Biofuel combustion | $413.0 \times 10^6$ [b] | 8.26 | 7.93 | 0.00 | 0.33 |
| Grassland/savanna burning | $52.1 \times 10^6$ [c] | 4.17 | 4.00 | 0.00 | 0.17 |
| Waste and residue burning | | 5.94 | 5.71 | 0.00 | 0.25 |
| Agricultural residue | $105.3 \times 10^6$ [c] | 3.90 | 3.74 | 0.00 | 0.17 |
| Household waste | $0.7 \times 10^6$ [d] | 2.05 | 1.96 | 0.00 | 0.08 |
| Cement production | $566.9 \times 10^6$ [e] | 22.68 | 18.14 | 3.40 | 1.13 |
| Iron and steel production | $123.0 \times 10^6$ [e] | 4.92 | 3.94 | 0.74 | 0.25 |
| Caustic soda production | $9.3 \times 10^3$ [d] | 0.19 | 0.13 | 0.06 | 0.00 |
| Non-ferrous metal smelting | | 167.8 | 134.23 | 25.17 | 8.39 |
| Zinc (Zn) | $1.7 \times 10^6$ [f] | 73 | 58.4 | 10.95 | 3.65 |
| Copper (Cu) | $1.1 \times 10^6$ [f] | 10.12 | 8.09 | 1.52 | 0.51 |
| Lead (Pb) | $0.9 \times 10^6$ [f] | 40.08 | 32.06 | 6.01 | 2.00 |
| Gold (Au): large scale | 20.4 [d] | 6.10 | 12.88 | 2.41 | 0.80 |
| Gold (Au): artisanal | 1.9 [g] | 28.50 | 22.80 | 4.28 | 1.43 |
| Mercury mining | 195.0 [f] | 8.78 | 7.02 | 1.32 | 0.44 |
| Battery/fluorescent lamp production | 485.0 [h] | 24.25 | 19.40 | 3.64 | 1.21 |
| Total | | 253.07 | 203.55 | 36.77 | 12.78 |

[a] From NBS (2001).
[b] From ECCCEY (2000).
[c] From Streets et al. (2003b).
[d] From Jiang (2004).
[e] From NBS (2000).
[f] From ECCNMI (2000).
[g] Artisanal gold smelting activities were officially banned in September 1996, but some mines continue to operate surreptitiously. In our study, we assume artisanal gold production in 1999 is 1.9 Mg, one-third of 1995 artisanal gold production (Feng, 2005).
[h] This is the amount of Hg used in battery and fluorescent lamp production (Jiang, 2004; Yang et al., 2003).

mercury emissions from all industrial sources excluding coal combustion. Zinc production in China is increasing significantly, reaching 1.7 million Mg in 1999. We estimated that total mercury emissions in 1999 were 73 Mg which is less than the value of 147.6 Mg estimated by Streets et al. (2005). Copper production in China was about 1.1 million Mg in 1999. Total mercury emissions from copper smelting were 10.12 Mg. Hg emissions from copper smelting are much lower than those of zinc smelting due to the use of a lower emission factor for copper smelting (9.6 g $Mg^{-1}$ of copper produced) in the estimation, which mainly results from much lower mercury contents in copper concentrate ore than those in zinc concentrate ore. Lead production in China was about 0.9 million Mg in 1999. It is estimated that total mercury emissions from lead smelting were 40.08 Mg.

In 1999, the 15 largest lead smelting plants contributed 57% of total lead yield. Most of these large plants are located in Hunan, Yunnan, Henan, and Guangdong Provinces (Streets et al., 2005). Because mercury emissions from gold smelting using amalgamation technology are strongly affected by the size of the smelting plant, the gold smelting process is separated into two parts: large-scale gold smelting in industrial plants and small-scale artisanal gold smelting.

Amalgamation technology is gradually being phased out in the large-scale gold smelting plants. In 1999, only about 20 Mg of gold were produced from large-scale plants using amalgamation technology, which resulted in 16.10 Mg of total mercury emissions. Although artisanal gold production was small in 1999, mercury emissions were still large due to the high emission factor for this process (Feng, 2005). It is estimated that total mercury emissions from small artisanal gold smelting were 28.50 Mg in 1999. Artisanal gold smelting was officially banned in September 1996, though it persists in remote areas. It is difficult to get precise gold production estimates from these small activities, and the mercury emission estimates from this activity are subject to large uncertainties. Battery/fluorescent lamp production and cement production emitted 24.3 and 22.7 Mg of mercury, which constituted 10% and 9% of the total mercury emissions from all industrial sources except coal combustion, respectively. Other sources contributed about 15% of the of the total mercury emissions from all industrial sources excluding coal combustion.

As shown in Table 3.4, among the total emission of 253.1 Mg from various industrial sources, 203.55 Mg, 36.77 Mg and 12.8 Mg were emitted as $Hg^0$, $Hg^{(II)}$ and Hg(p), respectively. Gaseous $Hg^0$ is the major form of mercury emitted from industrial sources other than coal combustion, and it constituted 80% of total mercury emissions. Divalent gaseous mercury ($Hg^{(II)}$) and particulate mercury (Hg(p)) constituted 15% and 5% of total mercury emissions, respectively.

## 3.5 Mercury Emission Trends from 1995 to 2003

Wu and co-workers (2006) developed multiple-year inventories of anthropogenic mercury emissions in China from 1995 through 2003. After updating mercury emissions estimates from zinc smelting using the new emission factors (Li, 2007 and Feng et al. 2004), we also analysed mercury emission trends for the industrial sources from 1995 to 2003 as shown in Table 3.5.

The total mercury emissions from industrial sources in China in 1995 was 296.4 Mg, increasing to 360.5 Mg in 2003, at an average annual growth rate of 2.90%. Non-ferrous metals smelting operations are known to be one of the largest sources of mercury in China (Streets et al., 2005; Wu et al., 2006). Due to fast economic development in China, the demand for non-ferrous metals is increasing significantly. As a result, total mercury emissions from the non-ferrous metals smelting category (zinc, lead, copper and gold) increased rapidly at an annual average rate of 4.30%, from 182.5 Mg in 1995 to 248.0 Mg in 2003.

**Table 3.5** Summary of total mercury emission estimates (*Mg*) from industrial sources from 1995 to 2003

| Source category | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | 2001 | 2002 | 2003 | AAGRa (%) |
|---|---|---|---|---|---|---|---|---|---|---|
| Non-ferrous metals smelting | 182.5 | 158.9 | 152.8 | 155 | 167.8 | 186 | 213.8 | 221.1 | 248 | 4.3 |
| 1) Zinc (Zn) | 50 | 49 | 66 | 69 | 73 | 85 | 105 | 105 | 115 | 11.6 |
| 2) Copper (Cu) | 10.4 | 10.7 | 11.3 | 8.4 | 10.1 | 12.7 | 13.7 | 14.8 | 17.6 | 6.9 |
| 3) Lead (Pb) | 26.5 | 30.8 | 30.9 | 33 | 40.1 | 48 | 54.3 | 57.8 | 70.7 | 13.0 |
| 4) Gold (Au): large scale | 10.1 | 11.4 | 16.1 | 16.1 | 16.1 | 11.8 | 12.3 | 15 | 16.2 | 6.0 |
| 5) Gold (Au): artisanal | 85.5 | 57 | 28.5 | 28.5 | 28.5 | 28.5 | 28.5 | 28.5 | 28.5 | -12.8 |
| Fuel oil for stationary sources | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.6 | 2.3 |
| Gasoline, diesel, and kerosene | 4.3 | 4.6 | 4.6 | 5 | 5.6 | 6.1 | 6.4 | 6.8 | 7.6 | 7.2 |
| Biofuel combustion | 10.1 | 9.1 | 8.7 | 8.7 | 8.3 | 8.6 | 9.5 | 10.6 | 10.7 | 0.7 |
| Grassland/ savanna burning[b] | 4.2 | 4.2 | 4.2 | 4.2 | 4.2 | 4.2 | 4.2 | 4.2 | 4.2 | 0.0 |
| Agricultural residue burning[b] | 3.9 | 3.9 | 3.9 | 3.9 | 3.9 | 3.9 | 3.9 | 3.9 | 3.9 | 0.0 |
| Household waste burning | 0.6 | 0.6 | 0.6 | 2 | 2 | 2.8 | 3.2 | 7.7 | 10.4 | 42.5 |
| Cement production | 19.9 | 20.5 | 21.3 | 21.4 | 22.7 | 23.9 | 27 | 29.4 | 35 | 7.4 |
| Iron and steel production | 3.8 | 4.1 | 4.4 | 4.6 | 4.9 | 5.1 | 6.1 | 7.3 | 8.9 | 11.2 |
| Caustic soda production | 2.4 | 2.4 | 2.5 | 1.3 | 0.2 | 0.2 | 0.2 | 0.2 | 0 | N/A |
| Mercury mining | 35.1 | 22.9 | 37.6 | 10.1 | 8.8 | 9.1 | 8.7 | 22.3 | 27.5 | -3.0 |
| Battery/fluorescent lamp production | 29.1 | 34.1 | 49.7 | 37.6 | 24.5 | 16.2 | 8.7 | 6.2 | 3.7 | -22.7 |
| Total | 296.4 | 265.8 | 290.8 | 254.3 | 253.4 | 266.6 | 292.2 | 320.2 | 360.5 | 2.9 |
| a) $Hg^0$ | 238.4 | 213.7 | 233.6 | 204.7 | 203.8 | 214.4 | 235.0 | 258.2 | 290.6 | 2.9 |
| b) $Hg^{(II)}$ | 43.2 | 38.8 | 42.7 | 36.9 | 36.8 | 38.7 | 42.5 | 45.9 | 51.7 | 2.7 |
| c) Hg(p) | 14.8 | 13.2 | 14.5 | 12.7 | 12.8 | 13.5 | 14.7 | 16.1 | 18.1 | 3.0 |

[a]Annual average growth rate.

[b]Assumed no change over time due to lack of data

In this category, zinc smelting is the largest single sector in total mercury emissions, reaching 115.0 Mg in 2003 at an average annual rate of 11.6%. However, lead smelting was the leading sector in mercury emissions growth: from 26.5 Mg in 1995 to 70.7 Mg in 2003, increasing by 13% annually. Total mercury emissions from copper smelting increased to 17.6 Mg in 2003, increasing at an annual rate of 6.9%. Gold smelting is the only sector with decreasing mercury emissions in this category, attributed to an official ban of artisanal gold production in China since 1996. In 2003, total mercury emissions from gold smelting were 44.7 Mg, compared to 95.6 Mg in 1995. It should be noted that the estimate of mercury emissions from metals smelting is subject to a high uncertainty due to limited test samples, lack of detailed information on metal smelting processes in typical Chinese plants, and lack of precise production estimates from small activities (Wu et al., 2006).

Besides non-ferrous metals smelting, cement production, mercury mining, battery and fluorescent lamp production, household waste burning, and biofuel burning are also major contributors of mercury emissions during the whole period or part of the period (1995-2003). Total mercury emissions from cement production (coal-related emissions are excluded to avoid double-counting with industrial coal use) increased steadily from 19.9 Mg in 1995 to 35.0 Mg in 2003, at an annual rate of 7.4%. In China, domestic mercury mining shrank dramatically in the late 1990s, but has rebounded since 2002. As a result, the total Hg emissions from mercury mining fluctuated significantly, decreasing from 35.1 Mg in 1995 to 8.7 Mg in 2001, then back up to 27.5 Mg in 2003. However, mercury emissions from artisanal mercury mining activities in Guizhou during that period of time were not included because it is very difficult to obtain the precise mercury production from this small scale mercury mining activity. It was estimated that the annual mercury emissions from artisanal mercury mining activity in the Wuchuan mercury mining area in Guizhou reached 3.9 to 9.8 Mg (Li et al., 2006). Mercury containing batteries are being phased out in China due to the release of a stringent standard in December 1997. Therefore, total mercury emissions from this sector increased initially, from 29.1 Mg in 1995 to 49.7 Mg in 1997, then decreased significantly from 1998, to as low as 3.7 Mg in 2003. Biofuels dominate rural energy supply in China. Total mercury emissions from biofuel burning have remained nearly constant, at around 10 Mg annually. Although household waste burning contributed only 0.6 Mg of mercury emissions in 1995, it is the leading sector among all of the mercury source sectors in emission growth, reaching 10.4 Mg in 2003 at an annual growth rate of over 40%. This is simply because the living standards of Chinese people has increased latterly and consequently the amount of waste produced by each family increased. Among other miscellaneous small sources, liquid fuels (gasoline, diesel, and kerosene) and iron and steel production are two sectors with high mercury emission growth, at annual average growth rates of 7.2 and 11.2%, respectively.

It is estimated that 81% of total mercury from industrial sources in 2003 is released as $Hg^0$, 14% as $Hg^{(II)}$, and 5% as Hg(p), compared to 80% as $Hg^0$, 15% as $Hg^{(II)}$, and 5% as Hg(p) in 1995.

## 3.6 Uncertainties

Quantifying Hg emissions is more difficult than quantifying, say, $SO_2$ or $NO_x$, because the emissions come from so many source types, not primarily combustion sources. In this respect Hg emissions are similar to VOC emissions. It is acknowledged that for some types of sources very little is known about actual activity levels and emission factors, and the choices in such cases rely heavily on inferences of activity levels from quite limited and uncertain statistical information. On the other hand, at least for combustion sources and releases from mercury containing ores, total emissions are constrained by the Hg content of the raw material, in a similar way to the sulfur content of fossil fuels, and this acts to reduce the uncertainty. Several factors influence the estimation of emissions, including emission factor and activity level. We estimate the uncertainty for each emitting sector by combining the coefficients of variation (CV, or the standard deviation divided by the mean) of the contributing factors. We then combine these uncertainties to estimate the total uncertainty of Hg emission estimates by quadrature average when the source estimates are uncorrelated. We follow the same detailed methodology for uncertainty analysis that was described in the TRACE-P inventory paper of Streets et al. (2003a). Figure 3.1 shows the results of uncertainty estimation in Hg emissions by source type.

The general findings are that Hg emissions are known least well in the artisanal gold smelting sector (±450%), followed by the mercury mining sector (±340%).

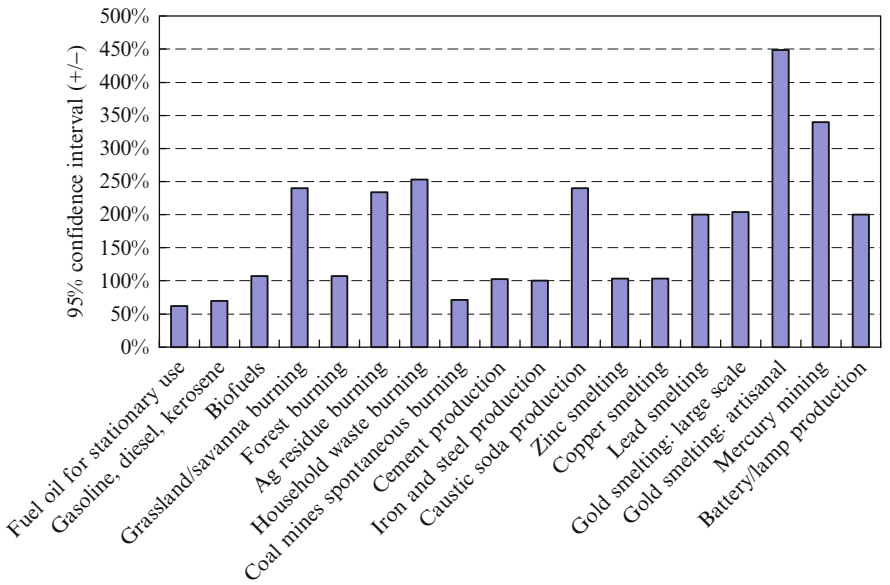


**Figure 3.1** Uncertainty (%) in Hg emission estimates (95% confidence intervals, ±) by sector (modified from Streets et al., 2005)

As the confidence intervals are frequently greater than the mean, the presentation of relative confidence intervals >±100% might suggest that the lower confidence interval is negative. However, the true confidence interval is not symmetric about the mean because some of the underlying variables are log-normally distributed. A better interpretation of "±400%", for example, might be "within a factor of five" so that the confidence interval would be 20-500% of the mean given.

## 3.7 Future Research and Policy Implications

Due to the fact that high uncertainties are associated with the current mercury emission inventory from industrial sources other than the coal combustion sector, a great number of studies need to be undertaken to reduce the uncertainties. First of all, surveys are needed to evaluate the mercury content of the raw materials of different industrial categories. Knowing mercury contents in raw materials, we can easily constrain the upper limit of mercury emissions from industrial sources. However, information regarding the mercury content of raw materials in industrial sectors is extremely scarce in China. Secondly, mercury balance studies are necessary for representative plants of different industrial sources. Until we know the mercury balance in an industrial process, it is impossible for us to determine what percentage of mercury in raw materials is emitted to the atmosphere. This kind of study is rarely reported in open literature in China. Thirdly, the speciation of mercury emissions from different industrial sources are also urgently needed in order to better understand the atmospheric fate of mercury emitted from these sources. Once we have all the above mentioned information, we will have accurate mercury emission inventory from industrial sources in China.

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