Chapter 2
Mercury Emissions from Coal Combustion in China

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Summary This chapter reviews the magnitude and spatial distribution of mercury emissions from coal combustion in China. Due to the large quantities of coal burned and the relatively low level of technology, particularly in industry, emissions are high. Emissions were stable at about 200-210 Mg during the period 1995-2000, but because of rapid economic growth starting in 2001, mercury emissions grew quickly to a value of 334 Mg in 2005. The annual average growth rate for the period 1995-2005 was 5.1%. The uncertainty in emission estimates is about ±35% (95% confidence intervals). Emissions are concentrated in those provinces with high concentrations of mercury in coal (like Guizhou Province) and provinces in which a lot of coal is burned (like Shanxi Province). Because significant amounts of coal are burned in homes and small industrial facilities, without any kind of emission control at all, emissions of particulate mercury are higher in China than in the developed world; the speciation profile nationwide is: 64% $Hg^{(II)}$, 19% Hg(p), and 17% Hg^0. In the future, growth in mercury emissions is expected to be limited by the application of FGD for SO_2 control and other advanced technologies. Estimates of emissions are hampered by the lack of comprehensive and reliable emissions testing programs in China.

2.1 Introduction

Mercury pollution has been recognized by Chinese researchers and government officials for some time. However, it is only relatively recently that researchers have begun to quantify the releases of mercury and measure the concentrations of mercury in the air, water and land. The serious nature of the pollution levels in China has now begun to raise issues that could lead to regulation of mercury emissions in the future. Feng (2005), Jiang et al. (2006), and Zhang and Wong (2007) have summarized the state of knowledge about mercury pollution in China. In addition, concern has been raised about transport of mercury away from the Asian continent and its contribution to regional and hemispheric background levels (see, e.g., Friedli et al., 2004; Jaffe et al., 2005; Pan et al., 2006). Coal combustion and nonferrous metals smelting are

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roughly equally responsible for mercury releases in China, supplemented by other industrial operations (Streets et al., 2005; Wu et al., 2006a). This chapter only addresses mercury releases from coal combustion. We discuss the contextual background for estimating emissions of mercury from coal combustion in China and present estimates for the period 1995 to 2005 with a forward glance to 2020.

2.2 Results and Discussion

2.2.1 Coal use Trends, 1995-2005

The major determinant of mercury emissions from coal combustion is the amount of coal burned. Coal consumption data for China are available by sector, coal type, and province from the China Energy Statistical Yearbooks (NBS, 1998-2005). It is important to distinguish between coal that is combusted directly and coal that is diverted to other uses, because this has major implications for mercury release rates. Trends in total raw coal consumption are shown in Figure 2.1. In 1995, total raw coal consumption was 1460 Tg, of which the industrial sector consumed 482 Tg (33%) for direct combustion, slightly more than the power sector, 446 Tg (31%). The residential sector consumed 138 Tg (9%). The remaining 27% of coal was used for coal washing, coking, industrial feedstocks, briquettes, and miscellaneous types of combustion. Coal use declined during the period 1996-1999 due to a variety of economic and other reasons (see, e.g., Sinton and Fridley, 2000), but subsequently began to increase quickly, as the economy of China underwent rapid expansion. By 2005, total raw coal consumption had risen to 2650 Tg, of which the major contributing sectors were: power plants 1050 Tg (40% of total), industrial combustion 718 Tg (27%), and residential use 138 Tg (5%).

Among the major coal-consuming sectors, the power sector was the leading sector in total coal growth, increasing by an average of 8.9% annually during the period 1995-2005. The industrial coal-combustion sector showed a moderate increase in coal use, 4.1% annually. Coal use in the residential sector was the same in 2005 as in 1995 in absolute terms, meaning that its share had been slowly decreasing (-0.1% per yr) mainly due to fuel transitions to cleaner gaseous and liquid fuels. Other uses of coal have grown as well, notably a tremendous annual-average growth rate of 17.8% in the use of coal as industrial feedstock, mostly achieved during the past five years. Figure 2.2 shows how the various uses of raw coal in the industrial sector have changed during the period 1995-2005.

2.2.2 Mercury in Coal

A reliable determination of the average or typical concentration of mercury in Chinese coals by province or nationwide is hampered by the innate heterogeneity of mercury in coal, as well as the relative paucity and unrepresentativeness of

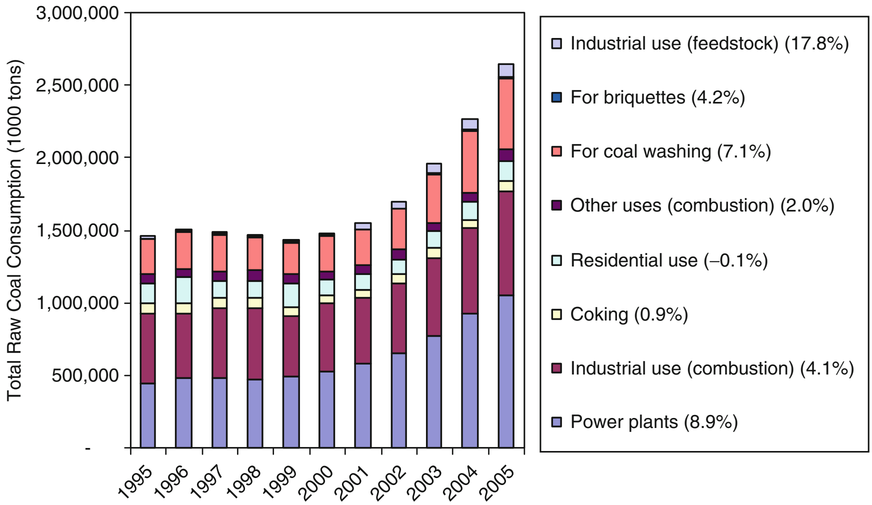

Figure 2.1 Trends in total raw coal consumption in China, 1995-2005; annual-average growth rates for the entire period are shown in the caption

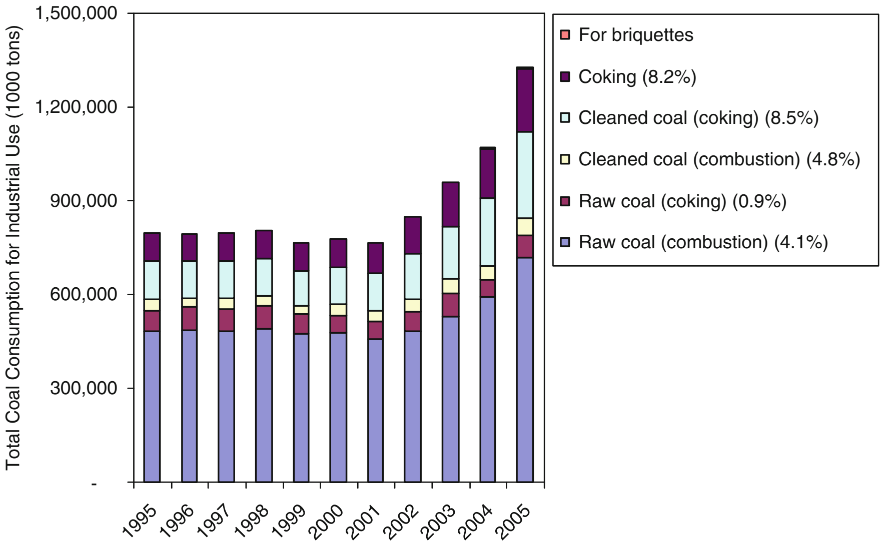

Figure 2.2 Trends in industrial raw coal consumption in China, 1995-2005; annual-average growth rates for the entire period are shown in the caption

measurements. Early Chinese studies on mercury emissions relied on limited sampling data. Wang et al. (1999, 2000) and Zhang et al. (2002) used an average value for the mercury concentration of Chinese coals of 0.22 g Mg^{-1}, with a wide range of 0.02-1.92 g Mg^{-1}, based on samples from fourteen provinces. Other estimates from the Chinese literature are 0.15 g Mg^{-1} (Huang and Yang, 2002) and 0.16 g Mg^{-1}

(Zhang et al., 1999). All these estimates were based on sampling of raw coal in coal-producing areas. An advancement in our understanding of mercury in Chinese coals occurred through an initiative by the U.S. Geological Survey, as part of the World Coal Quality Inventory, to measure about 300 coal samples from around China in collaboration with the Institute of Geochemistry in Guiyang. They obtained an average value of 0.15 g Mg^{-1} with a 1∑ standard deviation of 0.14, within a range of <0.2–0.69 g Mg^{-1}. Finally, Zheng et al. (2007) summarized previous studies of mercury in Chinese coals and reported new measurements of 1699 coal samples, having an average concentration of 0.19 g Mg^{-1}. The highest values of mercury content in raw coal are found in Guizhou Province (~0.52 g Mg^{-1}) (Zheng et al., 2007). Figure 2.3 presents the average mercury content of raw coal, as mined, for coal-producing provinces.

In order to obtain reliable estimates of the magnitude and spatial distribution of mercury emissions, it is essential to know the mercury content of the coal as burned, not just as mined. Therefore, it is necessary to relate the coal produced (mined) in particular provinces to its consumption in each province. Streets et al. (2005) and Wu et al. (2006a) developed a coal transportation matrix to link coal production to coal consumption. Using a merged data set from the USGS data and the Chinese literature data, they determined that the average mercury content of coal as burned was 0.18-0.19 g Mg^{-1}, varying very slightly in the range of 0.180 to 0.189 g Mg^{-1} during the period 1995 to 2003, due to fluctuations in provincial coal production. Streets et al. (2005) also calculated the mercury content of cleaned coal, coal briquettes, and coke, as produced. They assumed an average Hg removal

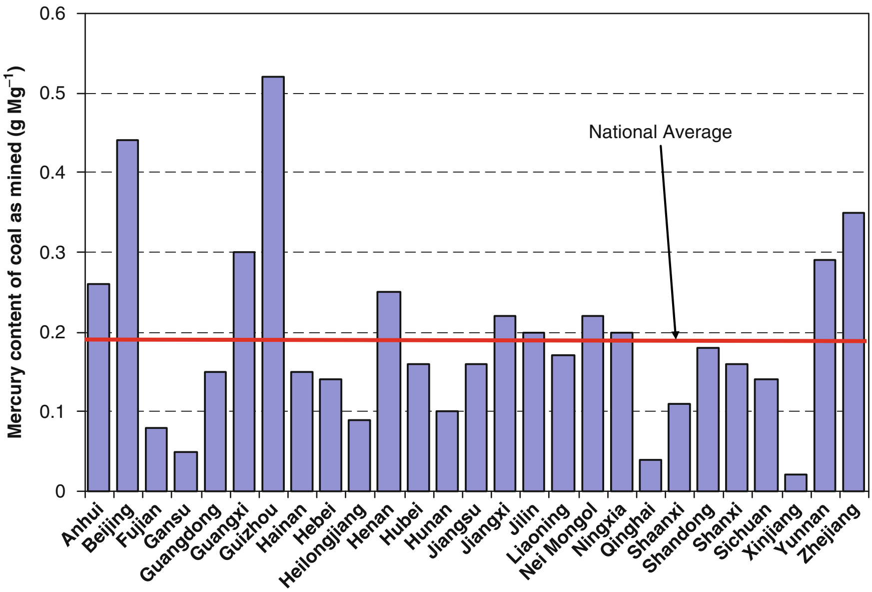

Figure 2.3 Mercury content of raw coal, as mined

efficiency for coal cleaning of 30% that is independent of the mercury content; during this period about 16% of total coal was cleaned in China. It was further assumed that 10% of the mercury contained in a given coal remains in coke after the coking process. Because there is no evidence of mercury removal during the briquette production process, it was assumed that 100% of the mercury in the raw coal or cleaned coal is transferred to the briquettes. Further tests on these and other coal-derived products are clearly called for.

2.3 Mercury Released to the Atmosphere

Because mercury release rates and the speciation profiles depend greatly on combustion technology, combustion conditions, and emission control technology, it is necessary to define Chinese coal utilization practices rather carefully. Streets et al. (2005) and Wu et al. (2006a) developed a model containing 65 individual source types for coal combustion, 22 of which are for coal-fired power plants, 30 for industrial use, nine for residential use, and four for other uses. The partitioning of each combustion technology/control device/fuel type by province and sector over time is built into their model based on a wide literature review.

In the past decade, the installation of particulate matter (PM) control devices in boilers has increased significantly in China, especially in the power sector. Since the mid-1980s, electrostatic precipitators (ESP) increased their share by 4-5% annually, to replace wet particle scrubbers and cyclones in power plants. Now the share of ESP installation in the total coal-fired power capacity is about 95% nationwide. However, in the industrial sector, the penetration of PM control installation lags behind. Although installation of wet particle scrubbers increased during the past decade, the fraction of industrial coal use without any PM control device is still large at present, close to 30%. The reasons are: (a) a large number of small boilers are scattered throughout China, especially in the poorer and more remote provinces such as Guizhou and Yunnan, without PM control; and (b) coke ovens, consuming a large amount of raw coal and clean coal, are generally without PM control. Since the mid 1990's, flue-gas desulfurization (FGD) also began to be installed in power plants to reduce SO_2 emissions. In 1995, FGD installed capacity was only 0.7 GW, rising to 5 GW in 2000; however, by the end of 2005, the FGD capacity had reached 53 GW, mostly in Sichuan (including Chongqing), Beijing, Shandong, Guangdong, Heilongjiang, Jiangsu, and Zhejiang Provinces. It is essential to reflect the rapidly changing mix of technologies in the coal-consuming sectors of China in calculations of mercury emissions over time. Figure 2.4 illustrates how the mix of PM controls changed during the period 1995-2003 in the power and industrial sectors. Residential use is also an important coal-consuming sector in China, representing 7% of raw coal, 5% of cleaned coal, and 90% of briquettes in 1999. Traditional cookstoves and improved cookstoves are the major combustion types for residential cooking and heating, both of which are without any PM control device. In the big cities, however, many residents obtain heat from centralized heating systems that

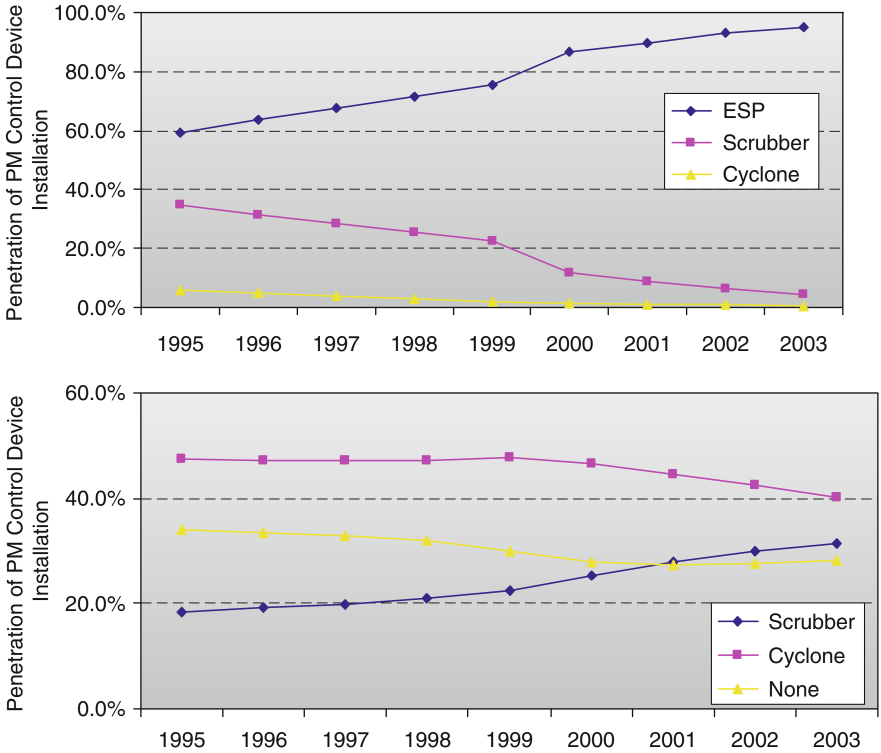

Figure 2.4 Time development of the penetration of PM control devices in China in the power sector (upper) and the industrial sector (lower), 1995-2003

use mid- or large-sized boilers. This part of the coal consumption for residential heating use assumes the use of stoker boilers with cyclone controls. For farming, construction, transportation, and commerce, the coal consumption is combined and assumes the use of small stokers without any PM control.

The typical scheme for calculating mercury emissions from coal combustion is illustrated in Figure 2.5. A fraction of the mercury contained in the fuel is not emitted to the air but is retained in the bottom ash and disposed of as solid waste. The share of Hg remaining in the bottom ash is different for different boiler types. Studies in China (Huang et al., 2003; Zhu et al., 2002) indicate that only 1-2% of Hg remains in the bottom ash for pulverized coal (PC) boilers in power plants; however, the ratio may increase to 7-9% for industrial PC or fluidized-bed boilers and 17-18% for industrial stoker-fired boilers (Wang et al., 2000; Wang and Ma, 1997).

The control technologies used to reduce traditional air pollutant emissions (e.g., PM and SO_2) from coal-combustion boilers also remove some of the mercury from the flue gas; however, the removal efficiencies vary widely. Until very recently, there were few measurements of mercury removal efficiencies for Chinese boilers. Wang et al. (1999, 2000) reported from measurements on two power plants in

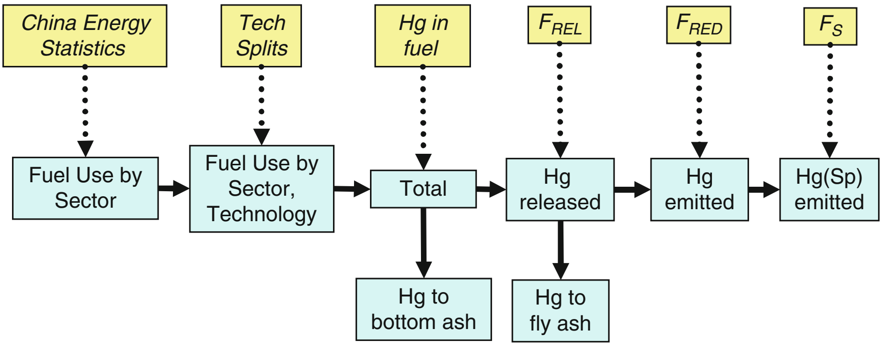

Figure 2.5 Calculation procedure for mercury emissions; F_{REL} = fraction released to the air during combustion; F_{RED} = fraction reduced by emission control devices; F_S = fraction emitted by species type

Changchun that mercury collection efficiency averaged 26% within a range of 7-47%. Zhu et al. (2002) suggested the following mercury removal efficiencies of the three predominant types of PM control devices installed in boilers in China: (a) ESP has a moderate removal efficiency of ~30%; (b) wet PM scrubbers show very little benefit, with mercury removal efficiency of ~4-8%; and (c) cyclones remove essentially no mercury (<0.1%). However, there is very little information about mercury removal efficiencies on devices other than ESPs on PC power plants in China.

Recently, programs of testing mercury emissions from Chinese sources have begun at Zhejiang University, Tsinghua University, and the Institute of Geochemistry in Guiyang, and studies are beginning to be published in the peer-reviewed literature. Tang et al. (2007), Chen et al. (2007), Yi et al. (2008), and Zhou et al. (2007) have all reported test data, including investigation of the roles of chlorine and ash in mercury release, but further work is needed to digest these results and generalize them to the population of source types in China.

2.4 Emission Trends in China

Wang et al. (1999, 2000) and Zhang et al. (2002) were the first to report mercury emissions from coal combustion in China, citing a value of 213.8 Mg for the year 1995. They further reported an annual average growth rate of ~4.8% a year for emissions in the 17 years prior to 1995, rising from ~95 Mg in 1980 to ~160 Mg in 1990.

Predicted emissions for 2000 were 273 Mg. Streets et al. (2005) conducted a detailed examination of mercury emissions from all sources and reported a value of 202.4 Mg for mercury from coal combustion in 1999, 38% of total emissions of mercury in China (535.8 Mg). Wu et al. (2006a), using the same methodology and data as in Streets et al. (2005), developed an emission trend from 1995 to 2003.

This trend incorporates the coal consumption and technology trends presented earlier. The results of that study are presented in Table 2.1 and Figure 2.6.

Table 2.1 Mercury emissions from coal combustion ($Mg\ yr^{-1}$)

	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	AAGR (%)
Power plants	63.4	68.7	67.2	66.2	67.8	70.1	76.3	84.2	100.1	114.3	124.8	7.0
Industrial use	104.7	106.3	107.8	108.3	103.2	104.2	101.9	109.9	124.3	136.4	169.4	4.9
Residential use	23.1	23.5	22.7	21.5	19.7	19.6	19.9	19.7	21.7	23.7	26.2	1.3
Other uses	11.2	10.8	10.5	11.6	11.5	10.5	10.7	11.8	10.6	11.7	13.6	2.0
Total	202.4	209.3	208.2	207.6	202.2	204.4	208.8	225.6	256.7	286.1	334.0	5.1

Data for 1995-2003 are from Wu et al. (2006a). This trend has been extended to 2004 and 2005 using the same methodology and data sources.
AAGR = Average Annual Growth Rate.

Additionally, for this present report, the trend has been extended from 2003 to 2005.

Wu et al. (2006a) find that mercury emissions from coal combustion increased from 202.4 Mg in 1995 to 334.0 Mg in 2005, an annual-average growth rate of 5.1%. The largest growth in mercury emissions (7.0% per year) has been in the power sector, consistent with the growth in coal combustion in the power sector, from 63.4 Mg in 1995 to 124.8 Mg in 2005.

Emissions from industrial coal combustion have grown by 4.9% per year, from 104.7 Mg in 1995 to 169.4 Mg in 2005. A formal uncertainty analysis has been conducted on these estimates, following the method described in Streets et al. (2003), and is shown in Figure 2.7. The 95% confidence intervals are approximately ± 35%, changing little over the time period.

The mercury emissions have also been speciated as described in Streets et al. (2005) and Wu et al. (2006a) across all emitting source types. The net result for coal combustion – varying over all sectors, source types and technologies – is shown in Figure 2.8 by province. This figure shows that the fraction of mercury emitted in particulate form is particularly high in Guizhou Province and to a lesser extent in Qinghai and Xinjiang Provinces.

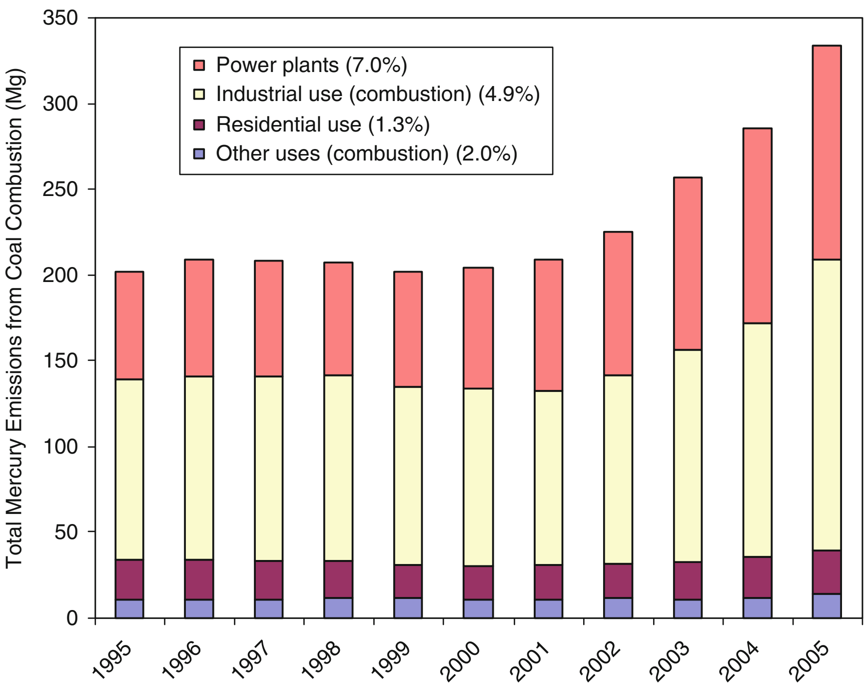

Figure 2.6 Trends in mercury emissions in China, 1995-2005; annual-average growth rates for the entire period are shown in the caption

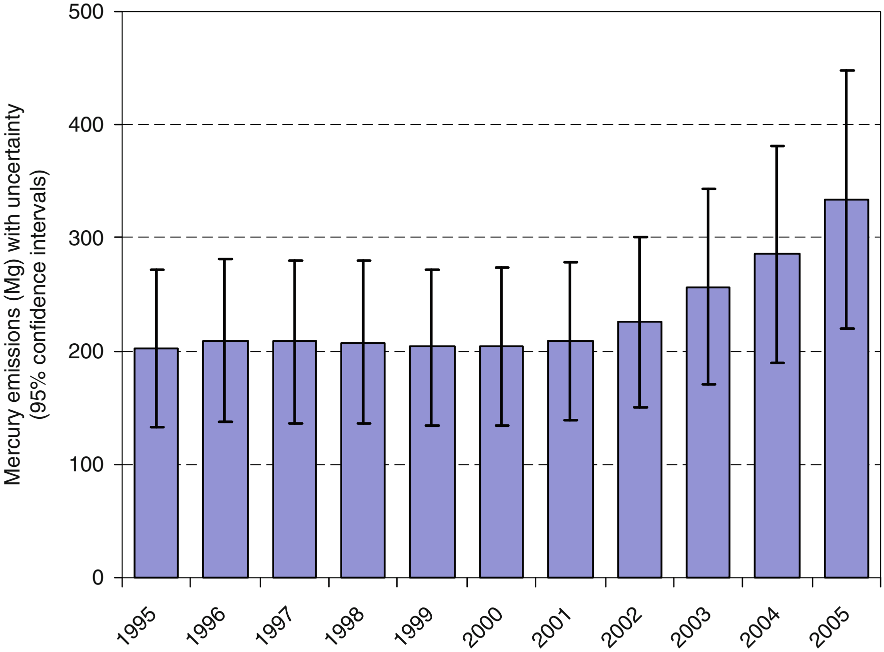

Figure 2.7 Uncertainty in mercury emission estimates for coal combustion, as 95% confidence intervals

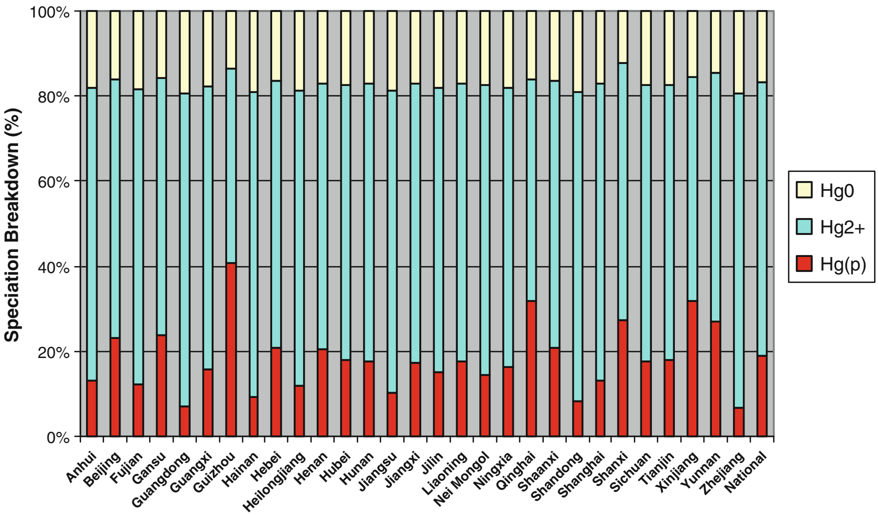

Figure 2.8 Speciation of mercury emitted from coal combustion in 1999, by province

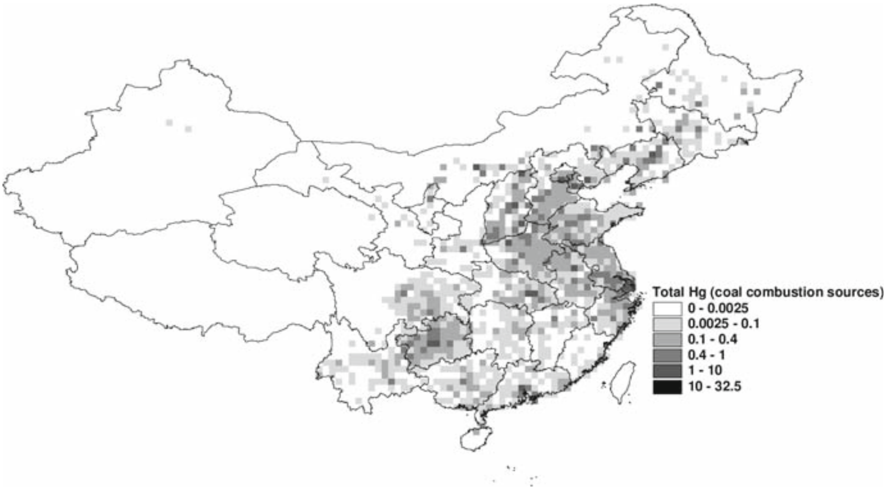

Figure 2.9 Gridded mercury emissions from coal combustion for the year 1999 at 30 min × 30 min spatial resolution (units are *Mg* yr^{-1} per grid cell)

This is a combination of high Hg content of coals and extensive use of coal in small, uncontrolled facilities. In the developed provinces – Guangdong, Shanghai, and Zhejiang, for example – particulate mercury releases are low. For the nation as a whole, the average speciation profile for mercury from coal combustion is: 64% $Hg^{(II)}$, 19% Hg(p), and 17% Hg^0. It should be noted that the mercury speciation profiles used thus far have relied on western data sources and are subject to change when new Chinese test data become available.

Streets et al. (2005) showed that the three provinces emitting the largest amounts of mercury from coal combustion in 1999 were Guizhou (18 Mg), Shanxi (15 Mg), and Henan (14 Mg) areas with heavy coal use and relatively low levels of technology. Figure 2.9 presents the spatial distribution of mercury emissions from coal combustion in 1999 at 30 min × 30 min resolution.

2.5 Future Mercury Emissions from Coal Combustion

Though mercury emissions from coal combustion have grown dramatically since 2001, there is hope for a change in the trend through the expected implementation of FGD on power plants. China announced in its 11th Five-Year Plan a renewed and concerted effort to control SO_2 emissions, intending to achieve a 10% reduction in 2010 emission levels relative to 2005. This goal will mostly be achieved by the installation of FGD units on a large number of power plants.

Reduction targets have been agreed upon with provincial governments and power companies, favorable electricity rate pricing and loans have been granted, and SEPA has been given ministerial status (Ministry of Environmental Protection) with greater enforcement powers. Even by 2006, the installed FGD capacity had doubled relative to 2005, from 53 GW to 104 GW. Figure 2.10 shows that implementation of FGD is expected to reach 58% nationwide by 2010 and 67% nationwide by 2020 starting with the developed coastal provinces and then spreading to the rest of the country. Because FGD also removes some mercury along with the SO_2, there will be a significant co-benefit for mercury reduction.

The key question is whether the implementation of FGD on power plants will be sufficient to offset the expected continued growth in power generation and coal combustion. Figure 2.11(a) shows the expected growth in power generation and coal use in the power sector out to 2020. Fast growth continues, with electricity generation growth outpacing coal growth due to improvements in energy efficiency. Coal use is expected to reach 1290 Tg in 2010 and 1770 Tg in 2020; electricity use rises to 2.62 billion MWh in 2010 and 3.80 billion MWh in 2020. Although more testing is needed to determine typical mercury removal efficiencies in Chinese power plants, we assume 74% reduction from an ESP + FGD configuration.

With this assumption, annual mercury emissions from coal-fired power plants are effectively held to about 2003 levels by 2010 (approximately 105 Mg) (Wu et al., 2006b). About 61 Mg of mercury emissions are avoided through FGD in this scenario, as shown in Figure 2.11(b). Emissions thereafter would begin to rise again, but with a modest additional investment in selective catalytic reduction (SCR) and activated carbon injection (ACI) technologies, 2020 emissions could also be held to the level of 100 Mg or thereabouts (Wu et al., 2006b). So the prospect of stabilizing mercury emissions from power plants is at hand.

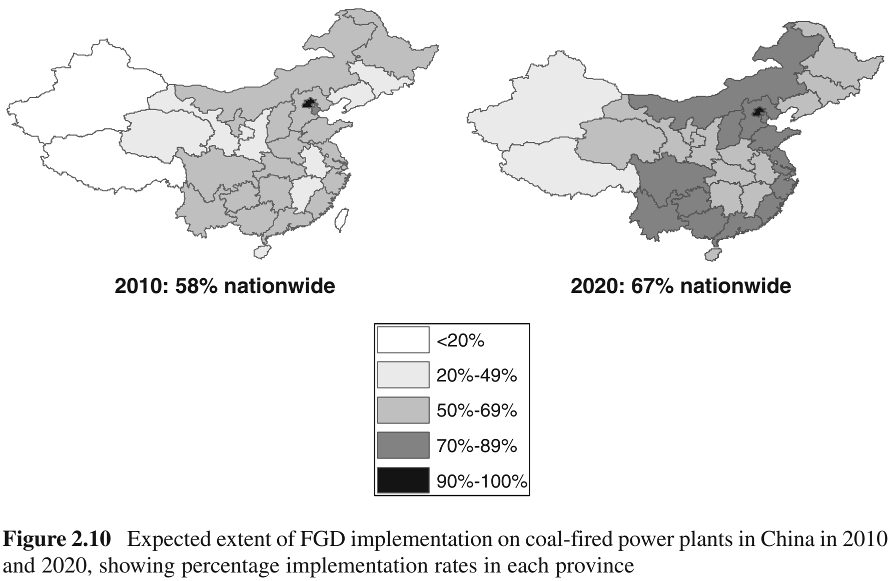

Figure 2.10 Expected extent of FGD implementation on coal-fired power plants in China in 2010 and 2020, showing percentage implementation rates in each province

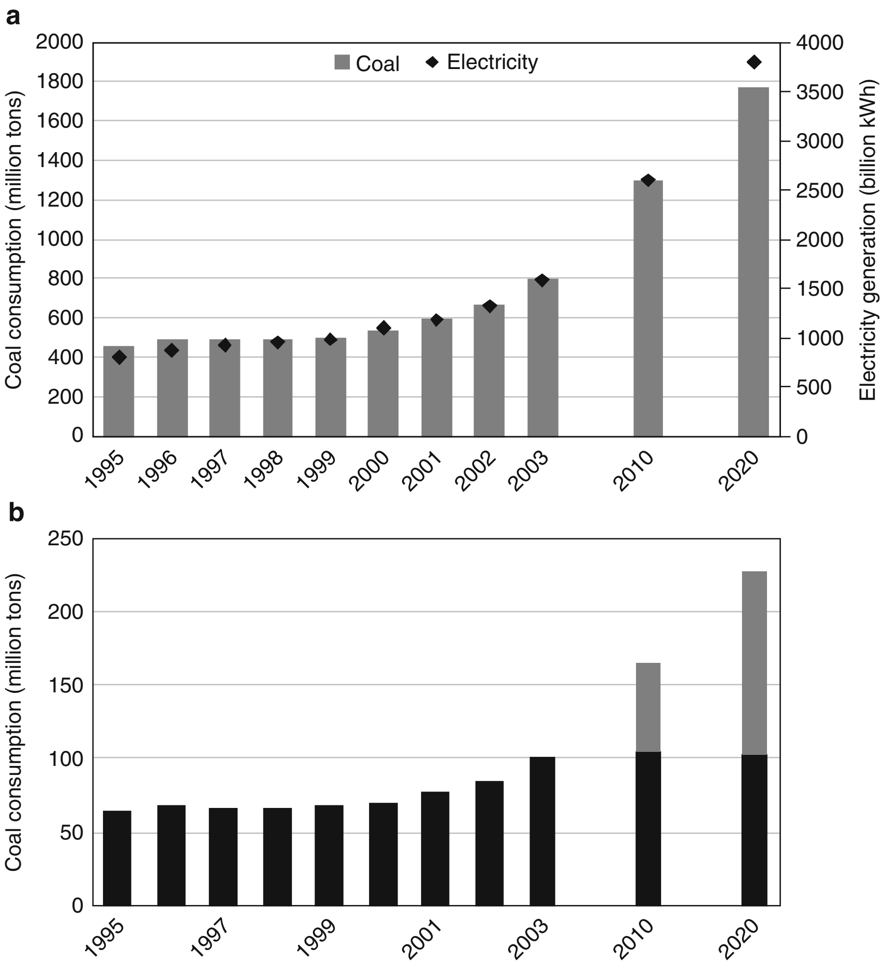

Figure 2.11 (a) Anticipated growth in power generation and coal use in power plants out to 2020; and (b) the effect of FGD and other controls on future mercury emission levels (grey), showing avoided emissions through application of emission control technology (black)

2.6 Future Research and Policy Implications

The existence of a mercury pollution problem has been known in China for several decades, but it is only recently that quantification has been attempted. Estimates of mercury emissions from coal combustion were first made about ten years ago, and they have improved over time. Considerable progress has been made in understanding the mercury content of Chinese coals. However, to a large extent, emissions quantification has had to rely on technology performance data as measured in the West. This is a considerable drawback, as we are not at all sure that facilities in

China achieve the same level of performance as they do in the West-and many of the special Chinese technologies have never been sampled at all. It was only in 2007 that a number of papers were published by Chinese researchers reporting on field testing of mercury emissions and collection in Chinese plants. Over the next few years, these results need to be collated, compared with western data, extrapolated to the whole of China, and supplemented with additional test data. In particular, the mercury collection efficiencies of PM control devices and FGD need to be refined when burning coals with the special chlorine and ash specifications of Chinese coals. When this has been accomplished, we will be able to have greater confidence in the estimates.

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