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Synergistic Mercury Removal by Conventional Pollutant Control Strategies for Coal-Fired Power Plants in China

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ABSTRACT
China's 11th 5-yr plan has regulated total sulfur dioxide (SO_2) emissions by installing flue gas desulfurization (FGD) devices and shutting down small thermal power units. These control measures will not only significantly reduce the emission of conventional pollutants but also benefit the reduction of mercury emissions from coal-fired power plants. This paper uses the emission factor method to estimate the efficiencies of these measures on mercury emission abatement. From 2005 to 2010, coal consumption in power plants will increase by 59%; however, the mercury emission will only rise from 141 to 155 t, with an increase of 10%. The average emission rate of mercury from coal burning will decrease from 126 mg Hg/t of coal to 87 mg Hg/t of coal. The effects of the three desulfurization measures were assessed and show that wet FGD will play an important role in mercury removal. Mercury emissions in 2015 and 2020 are also projected under different policy scenarios. Under the most probable scenario, the total mercury emission in coal-fired power plants in China will decrease to 130 t by 2020, which will benefit from the rapid installation of fabric filters and selective catalytic reduction.

IMPLICATIONS
Coal-fired power plants are important sources of atmospheric mercury pollution in China. Conventional air pollution control devices for SO_2, nitrogen oxides, and particulate matter may capture certain amount of mercury in flue gas. This paper assesses the co-benefits of SO_2 control measures on mercury reduction during in China's 11th 5-yr plan. This information provides scientific support for the policy-making on mercury pollution control in China.

INTRODUCTION
Mercury (Hg) has aroused global concern because of its toxic effects on human health, persistence in the environment, and long-range transport. China contributed approximately 28% to the global anthropogenic Hg emissions in 2000.[1] Coal combustion and nonferrous metal smelting were the two dominant sources in China and accounted for over 80% of the Hg emissions of China in 1999.[2] Wu et al.[3] found that China's total Hg emissions reached 696 ± 307 t by 2003. Emission from coal combustion increased from 202 to 257 t during from 1995 to 2003, with an annual growth rate of 3%. Although industrial coal combustion was the largest contributor because of its lack of control, coal-fired power plants drew much attention because of their high annual growth rate of 5.9%.

Emission control technologies for conventional pollutants (e.g., particulate matter [PM], sulfur dioxide [SO_2], and nitrogen oxides [NO_x]) will not only significantly reduce the emission of conventional pollutants, but will also synergistically benefit the removal of Hg emissions from coal-fired power plants.[4] The U.S. Environmental Protection Agency (EPA) reported the average Hg capture rates of different air pollution control devices (APCDs),[5,6] especially PM and SO_2 control devices. In some studies, NO_x control devices were also mentioned as having certain Hg capturing abilities.

China's 11th 5-yr plan has regulated total SO_2 emissions by installing flue gas desulfurization (FGD) devices and shutting down small thermal power units.[7] According to the regulation, three control strategies are going to be executed before 2010: (1) all of the newly built units should install FGD devices, (2) all of the existing units that exceed the emission standard should install FGD devices, and (3) small power units serving for more than 20 yr or having a capacity of less than 100 MW should be closed. Control of NO_x emissions will also be initiated during the period of 2005–2020.

To estimate the co-benefit of Hg removal by conventional pollutant control strategies, especially the SO_2 abatement strategies in the 11th 5-yr plan, this paper will establish three scenarios for 2010 on the basis of the benchmark emission inventory in 2005. A further scenario analysis was conducted for 2015 and 2020 on the basis of the future installation of PM, SO_2, and NO_x control devices.

METHODOLOGY

Method

Emission of Hg from coal-fired power plants at the province level in China was estimated through a detailed emission factor approach for the years 2005 and 2010. Coal consumption, Hg content, and emission control technology were taken into account using eq 1.

$$TME = \sum_{\mathrm{i}} \left[A_{\mathrm{i}} \cdot M_{\mathrm{i}} \cdot (1 - P \cdot w) \cdot R \cdot \left(1 - \sum_{\mathrm{n}} (C_{\mathrm{i,n}} \cdot \eta_{\mathrm{n}}) \right) \right] \tag{1}$$

where subscripts i and n stand for provinces and combinations of emission control devices, respectively; TME is the total Hg emission; A is the coal consumption; M is the Hg content of coal; P is the percentage of coal prewash in power plants; w is the Hg removal efficiency of coal prewash; R is the release factor of Hg from boiler; C is the application rate of a certain combination of emission control devices; and η is the removal efficiency of a combination of emission control devices.

For 2015 and 2020, emissions from coal-fired power plants were calculated at the national level.

Only PM and SO_2 control strategies were considered for 2005 and 2010 under the assumption that the application of NO_x control strategies will remain quite limited nationally before 2010, whereas all conventional pollutant control strategies were taken into consideration for 2015 and 2020.

Hg Content of Coal in China

The dominating element of the Hg emission factor is the Hg content of coal. In the early studies in China, Hg content data were quite limited. Wang et al.[8,9] and Zhang et al.[10] used 0.22 mg/kg as a national mean value, which was derived from coal analysis of 14 provinces. The values varied from approximately 0.02 to 1.92 mg/kg. Other research yielded estimated values of 0.15 mg/kg[11] and 0.16 mg/kg.[12] All of these results came from very limited raw coal samples from coal mines. To develop a more convincing inventory, the U.S. Geological Survey (USGS) did further studies after analyzing 276 samples from all provinces in China[13] and obtained an average Hg content of 0.15 ± 0.14 mg/kg. On the basis of data from USGS and other research, Streets et al.[2] presented a complete Hg content database by province in China. They also developed a coal transport matrix for China, combining coal production with coal consumption, and obtained a value of 0.19 mg/kg for the Hg content of coal burned in the power sector in China. Zheng et al.[14,15] analyzed 62 samples, summarized 1699 samples from previous studies, and reported the national average to be 0.19 mg/kg. Ren et al.[16] conducted a more detailed data investigation and summarized previous results of 619 samples in their book. Wu et al. (unpublished) ran a stochastic simulation for the Hg content of raw coal by province. The lognormal distribution was found to fit the input dataset for Guizhou and Shanxi provinces. The values with a probability of 50% for these two provinces were more applicable than the mean values. The study presented here integrated the independent data from Ren et al.,[16] USGS,[13] Zheng et al.,[15] and other research and updated the database of Hg content of coal by province (see Table 1).

The Hg content of coal used in power plants by province, as shown in Figure 1, was calculated via the coal transport matrix.[17] For the major coal-supplying provinces such as Shanxi, Shaanxi, Inner Mongolia, and Guizhou, there is little difference between the Hg content as produced and as burned in power plants because within-province supply can meet the demand. The highest Hg content of coal as burned in the power sector (0.32 mg/kg) was found in southwestern China, followed by northwestern and eastern China. Center and southern, northern, and northeastern China have relatively lower Hg content. These values are all of the weighted means on the basis of the coal consumption by province in 2005. The national average was calculated to be 0.18 mg/kg. Chongqing, Guizhou, Anhui, Yunnan, and Guangxi are the top five provinces that consume high-Hg coal.

Hg Removal Efficiency

Hg can be removed by coal washing or it can be captured by APCDs. Washed coal only accounted for 1.5% of the electricity coal consumption in 2005 (National Bureau of Standards and National Development and Reform Commission [NDRC], 2006). The U.N. Environment Programme (2005) reported that the Hg removal efficiency by coal washing was approximately 10–50%.[18] The median value of 30% was used in this study. During coal combustion, approximately 99% of the Hg in coal is emitted into the flue gas, with less than 1% retention rate in bottom ash.[19,20] When the flue gas goes through the control devices designed for conventional air pollutants including PM, SO_2, and NO_x, some Hg in the flue gas is captured into fly ash or gypsum. Although EPA has reported Hg removal efficiencies for different APCDs, this might not be the case in China because of different coal compositions. Therefore, this study summarizes the Hg removal efficiency studies in China as shown in Table 2. The coal used in all test plants was bituminous coal.

Two arithmetic means of 20 and 54% were found for electrostatic precipitation (ESP) and ESP+FGD, respectively, on the basis of the test results from 15 coal-fired power plants in China. However, the efficiency results differ a great deal. One possible reason is the Hg content of coal. Setting 0.10 mg/kg as a critical value, the plants that consumed coals below this value had only 13% Hg removal in ESP devices, whereas the ones using coals above this value achieved 23% removal. The discrepancy was even more significant during ESP+FGD (20% vs. 76%). As shown in Figure 1, in China more than 90% of the coal burned in power plants has a Hg content of more than 0.10 mg/kg. The weighted average Hg content of coal used in tested plants was 0.181 mg/kg, similar to the

Table 1. Hg content of raw coal in China by province (mg/kg).

Province	This study	Ren et al.[16]	USGS[13]	Zheng et al.[15]	Wu et al. (unpublished)	Streets et al.[2]	Zheng et al.[14]
Anhui	0.36[a]	0.46 (50)[i]	0.19 (11)	0.26 (29)		0.26	0.21
Beijing	0.33[b]	0.10 (1)	0.55 (1)			0.44	0.34
Chongqing	0.46[b]	0.64 (12)	0.15 (7)				
Fujian	0.10[c]		0.10 (2)			0.08	
Gansu	0.26[b]	1.35 (1)	0.05 (5)			0.05	
Guangdong	0.10[b]	0.10 (1)	0.09 (1)			0.15	
Guangxi	0.35[c]		0.35 (5)			0.30	
Guizhou	0.36[e]	0.70 (133)	0.21 (15)		0.357	0.52	1.14
Hainan	0.10[f]					0.15	
Hebei	0.16[b]	0.16 (33)	0.15 (14)			0.14	0.46
Heilongjiang	0.10[b]	0.12 (14)	0.06 (10)			0.09	0.13
Henan	0.16[a]	0.14 (115)	0.21 (27)	0.57(1)		0.25	0.17
Hong Kong[h]	–						
Hubei	0.18[b]	0.23 (1)	0.16 (3)			0.16	
Hunan	0.11[b]	0.08 (14)	0.15 (9)			0.10	0.07
Inner Mongolia	0.17[a]	0.17 (14)	0.17 (15)	0.19 (4)		0.22	0.16
Jiangsu	0.24[b]	0.18 (10)	0.35 (6)			0.16	0.09
Jiangxi	0.22[b]	0.13 (4)	0.27 (7)			0.22	0.16
Jilin	0.15[b]	0.34 (2)	0.07 (5)			0.20	0.34
Liaoning	0.16[a]	0.14 (16)	0.19 (9)	0.23 (1)		0.17	0.17
Macao[h]	–						
Ningxia	0.28[b]	0.28 (19)	0.27 (3)			0.20	
Qinghai	0.25[b]	0.31 (4)	0.04 (1)			0.04	
Shaanxi	0.23[b]	0.30 (3)	0.21 (7)			0.11	0.64
Shandong	0.24[a]	0.18 (11)	0.13 (19)	0.37 (22)		0.18	0.28
Shanghai[h]	–						
Shanxi	0.09[e]	0.17 (79)	0.17 (77)	0.17 (4)	0.091	0.16	0.08
Sichuan	0.26[b]	0.35 (14)	0.12 (8)			0.14	0.18
Taiwan	0.06[d]	0.06 (4)					
Tianjin[h]	–						
Xinjiang[h]	0.08[b]	0.09 (6)	0.04 (2)			0.02	0.03
Xizang	–						
Yunnan	0.30[a]	0.32 (56)	0.14 (7)	0.3 (1)		0.29	0.30
Zhejiang	0.35[g]	0.75 (2)				0.35	

Notes: [a]The data from Ren et al.,[16] USGS,[13] and Zheng et al.[15] are merged for these provinces; [b]The data from Ren et al.[16] and USGS[13] are merged for these provinces; [c]The mean values from USGS[13] are used; [d]The mean value from Ren et al.[16] is used for Taiwan; [e]The values with a probability of 50% from Wu et al. (unpublished) are used for Guizhou and Shanxi; [f]The value for Hainan is assumed to be equal to that for Guangdong because of a lack of samples; [g]The value for Zhejiang is derived from Streets et al.[14]; [h]Hong Kong, Macao, Shanghai, Tianjin, and Xizang do not produce raw coal; [i]The expression of 0.46(50) means 50 samples with an average value of 0.46 mg/kg.

national average value. The weighted average efficiencies of ESP and ESP+FGD turned out to be 22 and 70% respectively, which were more convincing.

In UNEP's toolkit,[18] 36% for ESP and 74% for ESP+FGD were quoted from the EPA report. The efficiencies of these two combinations were 52 and 74% in Japanese power plants[21] and 59 and 67% in tests conducted in South Korea,[22] respectively. The differences for the efficiency of ESP+FGD were acceptable, but the situation for ESP was more complex. Coal composition is one of the important factors influencing the Hg removal efficiencies of ESP and FGD.

More than 99% of particulate Hg (TPM) is removed in ESP.[23,24] There are two processes for removal of gaseous Hg in ESP: (1) oxidation of gaseous elemental Hg (GEM), and (2) adsorption of reactive gaseous Hg (RGM) (Figure 2). The chlorine content of coal was an important factor influencing the Hg removal in ESP. Figure 3 shows the influence of chlorine on the distribution of Hg in flue gas before ESP, which might contribute to the Hg removal inside ESP. Chlorine species are the primary reagents for oxidizing GEM into RGM. The average chlorine content of coal used in tested plants was 468 mg/kg. Ren et al.[16] reported that the mean chlorine content of 721 Chinese coal samples was 274 mg/kg, which was much lower than the arithmetic mean for 4906 U.S. coal samples—628 mg/kg. This could be an important reason for why ESP devices in U.S. power plants have a higher removal efficiency for total Hg. However, no significant correlation was observed between the GEM/RGM/TPM percentage in flue gas and the ESP efficiency, indicating that chlorine content was not the only dominant aspect of coal quality affecting Hg removal efficiency by ESP. Loss on ignition (LOI) could be another important factor. The unburned carbon (UBC) component of LOI catalyzes GEM oxidation and retains TPM as the most effective inherent sorbent for Hg in fly ash.[25]

In contrast with ESP, the Hg removal efficiency of FGD was highly dependent on the proportion of RGM in flue gas before FGD (Figure 4). RGM can be sufficiently removed by FGD because of its high solubility in water.

Another factor that might affect Hg removal is the type of coal. According to the EPA report, APCDs have a

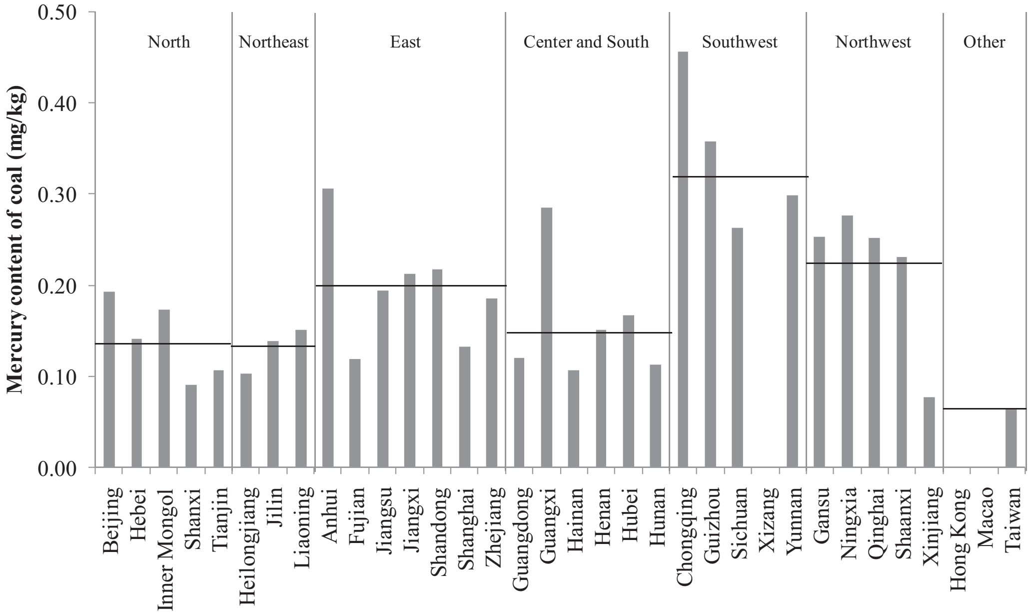

Figure 1. Hg content of coal used in power plants in China by province (mg/kg).

higher Hg reduction efficiency for bituminous coal than other coals. In China, approximately 85% of the coal used the in power sector is bituminous coal, 10% is anthracite, and 5% is lignite. Considering the dominance of bituminous coal use and the lack of Hg removal efficiency data for anthracite and lignite, all of the removal rate values were based on results for bituminous coal.

The efficiencies for combinations other than ESP and ESP+FGD were obtained from values in the literature because of lack of data for Chinese power plants. Table 3 lists all of the Hg removal efficiencies that were used in the calculation of the inventory.

SYNERGISTIC HG REMOVAL 2005–2010

Hg Emission Scenarios

Coal consumption and the application of different control devices are the major reasons for the differences between different scenarios. Because ESP is used in over 90%

Table 2. Hg removal efficiency results from studies in China.

No.	Capacity (MW)	Combinations	Hg Content (mg/kg)	Chlorine Content (mg/kg)	ESP Efficiency (%)	ESP+FGD Efficiency (%)	Reference
1	220	PC+ESP	0.011	152	6		Chen et al.[23]
2	600	PC+ESP	0.210	202	21		Chen et al.[23]
3	300	PC+ESP	0.170	600	18		Zhou et al.[27]
4	600	PC+ESP	0.160	700	16		Zhou et al.[28]
5	300	PC+ESP	0.245	541	13		Zhou et al.[24]
6	200	PC+ESP	0.200	362	15		Tang[19]
7	600	PC+ESP	0.204	180	39		Wang et al.[29]
8	220	PC+ESP	0.010	267	36		Yang et al.[30]
9	100	PC+ESP	0.120	1006	25		Duan et al.[31]
10	300	PC+ESP	0.328		13		Guo et al.[32]
11	600	PC+ESP+FGD	0.035	630	4	27	This study
12	600	PC+ESP+FGD	0.091	510	4	13	Chen et al.[23]
13	200	PC+ESP+FGD	0.233		35	71	This study
14	600	PC+ESP+FGD	0.142		43	74	This study
15	300	PC+ESP+FGD	0.174		18	81	This study
Arithmetic mean			0.155	468	20	54	
Hg content $<$ 0.10 mg/kg			0.037		13	20	
Hg content $>$ 0.10 mg/kg			0.199		23	76	
Weighted mean			0.181		22	70	
References		United States			36	74	EPA[6]
		Japan			52	74	Ito et al.[21]
		S. Korea			59	67	Lee et al.[22]

Notes: PC = pulverized coal boiler.

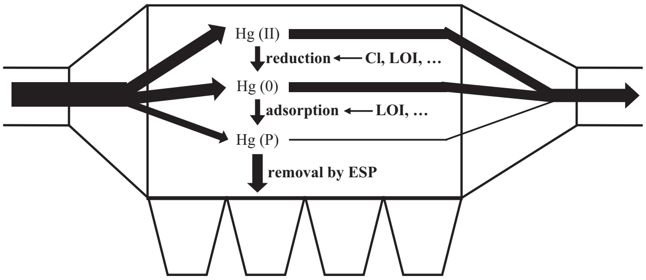

Figure 2. Hg removal processes in ESP.

of the power plants, the differences in its application between provinces were ignored. In other words, only the provincial differences in the application of FGD were considered. To evaluate the effect of the SO_2 control strategies in the 11th 5-yr plan on Hg removal, the inventories for 2005 and 2010 were calculated based on the 11th 5-yr plan. Three SO_2 control measures were evaluated.

(1) Strategy A: All of the newly built units install FGD devices.
(2) Strategy B: All of the existing units that exceed the emission standard should install FGD devices.
(3) Strategy C: Small power units serving for more than 20 yr or having a capacity of less than 100 MW are shut down.

Coal consumption and the application of FGD by province in China are summarized in Figure 5. All of the statistic data were unit based and originated from the database for Chinese coal-fired power units.[26] This database also includes the units under construction or planned to be built before 2010. The prediction of coal consumption for 2010 was in accordance with the power capacity of each unit. FGD devices are going to be installed for all of the newly built units according to the 11th 5-yr plan. Table 4 lists the proportion of different types of boilers and other emission control devices.

Synergistic Hg Removal by SO_2 Control Measures

A provincial Hg emission inventory for coal-fired power plants in China for 2005 and 2010 was developed using an emission factor method (Table 5). The top five emitters

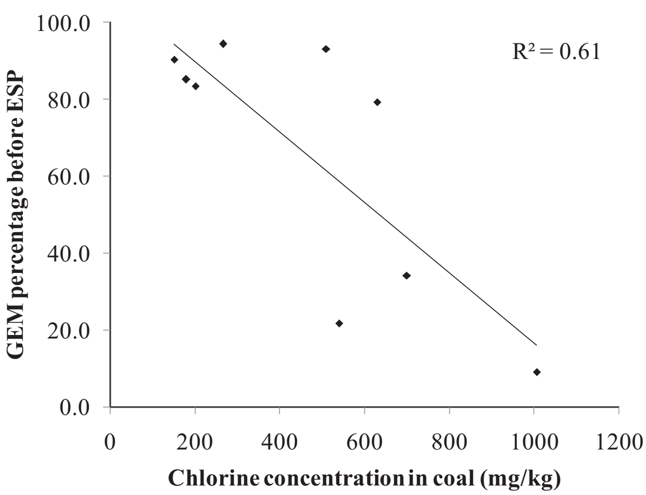

Figure 3. The relationship between chlorine in coal and GEM before ESP.

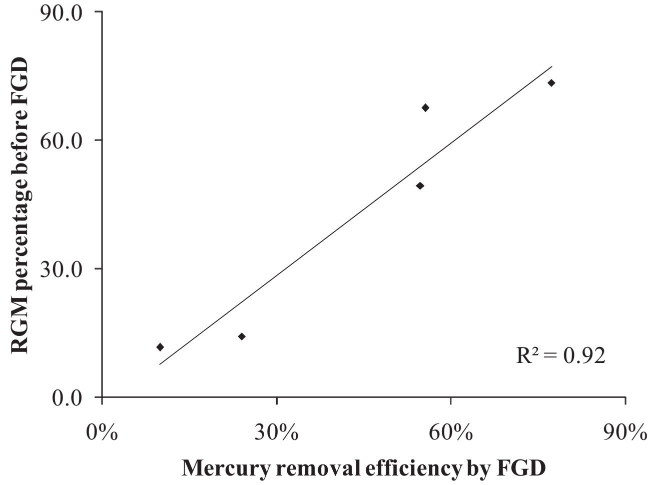

Figure 4. The relationship between efficiency of FGD and RGM before FGD.

in 2005 were Shandong, Jiangsu, Henan, Anhui, and Inner Mongolia. The top 10 emitters contributed over 60% of the total Hg emission from the power sector in China.

From 2005 to 2010, coal consumption will increase by 59%, which means the total national Hg emission will rise by 73% if no specific control strategy is implemented. In fact, the total emission will rise from 141 to 155 t because the three SO_2 control measures will counteract most of the influence brought about by the rapid growth of coal consumption. The control strategies are more effective to remove RGM and TPM. From 2005–2010, emissions of RGM and TPM will decrease 19 and 18%, respectively; however, emissions of GEM will increase 27%. Hg emission per ton of coal will drop from 126 to 87 mg with a reduction of 31%. The effects of SO_2 control strategies found in each province were quite different. Qinghai, Chongqing, Inner Mongolia, Guangxi, and Zhejiang will have a rapid increase in Hg emission from 2005 to 2010, whereas Beijing, Shanghai, Hainan, Sichuan, and Hebei (the top five provinces) will realize an obvious Hg reduction during this period.

If no control strategies were carried out, the total Hg emission would increase to 243 t according to the unit-based thermal power coal consumption prediction. Therefore, the total effect of the three strategies can be described as cutting 86% off the potential 103 t of Hg emission growth. Strategies A, B, and C contribute 59, 35,

Table 3. Hg removal efficiencies for different emission control device combinations.

Combinations	Removal Rate (%)	Reference
ESP	22	This study
ESP+FGD	70	This study
SCR+ESP+FGD	85	Unpublished study
FF	90	EPA[6]
FF+FGD	98	EPA[6]
WSCRB	6.5	Jiang[17]
CYC	0.1	Jiang[17]

Notes: WSCRB = wet scrubber, CYC = cyclone.

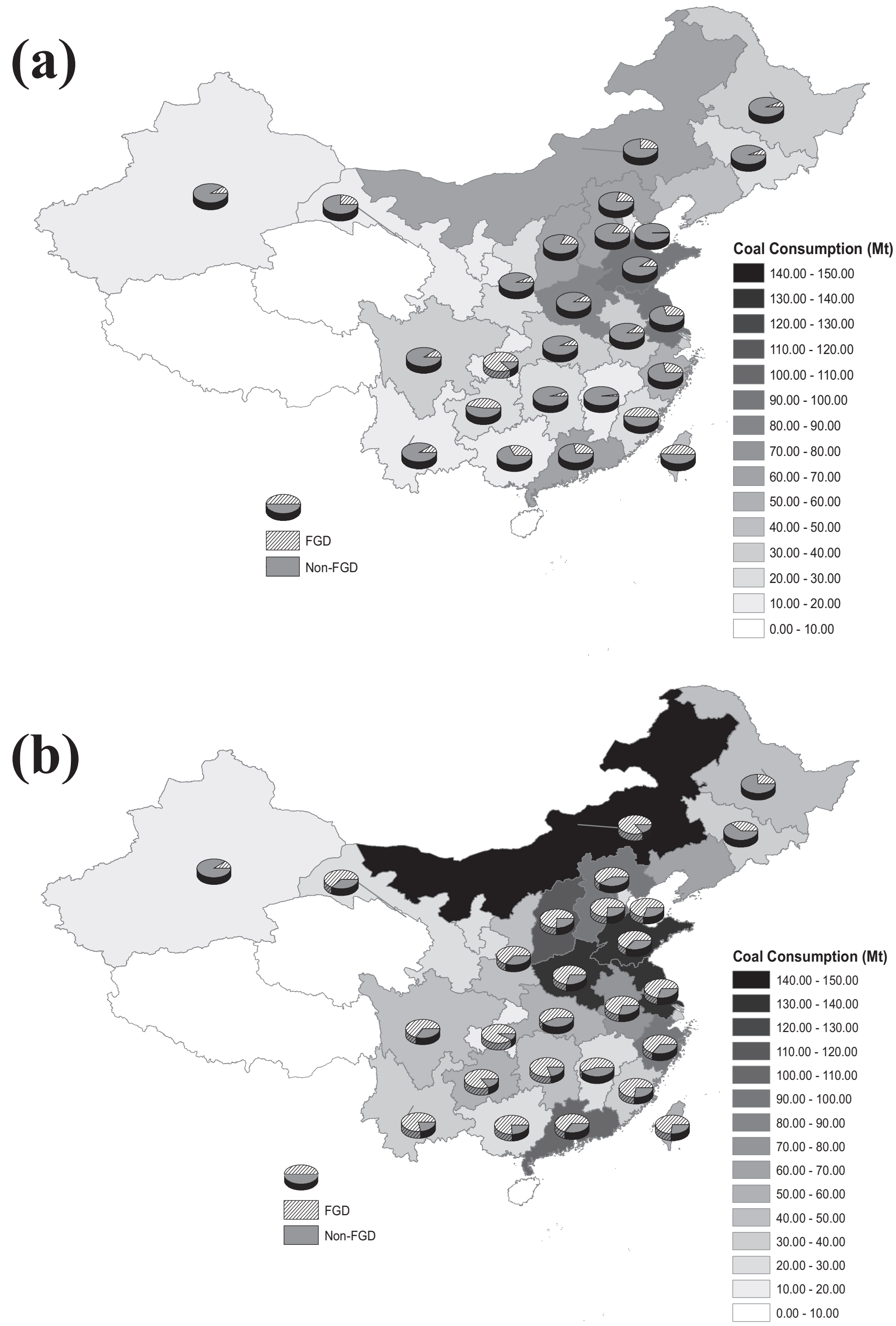

Figure 5. Coal consumptions and FGD device installation ratios in China by province for (a) 2005 and (b) 2010.

Table 4. Applications of different type of boilers and other emission control devices.

	2005		2010	
	PC Boiler (92%)	**Stoker (8%)**	**PC Boiler (95%)**	**Stoker (5%)**
ESP	0.93	0.00	0.96	0.00
WSCRB	0.07	0.88	0.04	0.88
CYC	0.00	0.12	0.00	0.12
FF	0.00	0.00	0.00	0.00

and 6% to the total 88 t of Hg removal, respectively. Installation of FGD devices in all newly built units will reduce more than 50 t of Hg, followed by the FGD device installation in existing power plants. Installation of FGD devices in all newly built units will reduce 52 t of Hg, of which 61% is RGM and 37% is GEM. FGD device installation in existing power plants will reduce 16 t of GEM, 14 t of RGM, and 0.9 t of TPM. Closedown of small power units will remove 3 t of GEM and 2 t of RGM.

SCENARIO ANALYSIS FOR 2015 AND 2020

From 2010 to 2020, fabric filters (FFs) will gradually replace a considerable proportion of ESP. The proportion of

Table 5. Hg emissions and shares under different scenarios.

	2005		2010	
Province	**Emission (t)**	**Share (%)**	**Emission (t)**	**Share (%)**
Anhui	8.66	6.16	10.38	6.68
Beijing	1.30	0.92	0.97	0.62
Chongqing	2.06	1.47	3.42	2.20
Fujian	1.66	1.18	2.17	1.40
Gansu	3.20	2.27	3.44	2.21
Guangdong	5.30	3.77	6.58	4.23
Guangxi	2.29	1.63	3.04	1.95
Guizhou	6.44	4.58	8.09	5.21
Hainan	0.16	0.12	0.14	0.09
Hebei	6.56	4.66	6.07	3.90
Heilongjiang	2.79	1.98	3.10	2.00
Henan	9.38	6.66	9.87	6.35
Hong Kong	0.00	0.00	0.00	0.00
Hubei	4.95	3.52	5.03	3.23
Hunan	2.12	1.50	2.23	1.43
Inner Mongol	7.16	5.09	10.56	6.79
Jiangsu	12.93	9.19	12.96	8.34
Jiangxi	2.87	2.04	3.22	2.07
Jilin	2.56	1.82	2.81	1.81
Liaoning	5.57	3.95	5.50	3.54
Macao	0.00	0.00	0.00	0.00
Ningxia	3.10	2.21	3.29	2.11
Qinghai	0.12	0.09	0.35	0.23
Shaanxi	4.77	3.39	4.99	3.21
Shandong	14.90	10.59	15.15	9.75
Shanghai	3.57	2.54	2.92	1.88
Shanxi	4.19	2.98	4.74	3.05
Sichuan	7.12	5.06	6.39	4.11
Taiwan	1.81	1.28	1.79	1.15
Tianjin	1.28	0.91	1.30	0.84
Xinjiang	0.63	0.44	0.82	0.53
Xizang	0.00	0.00	0.00	0.00
Yunnan	4.43	3.15	5.11	3.29
Zhejiang	6.84	4.86	9.02	5.81
National	140.73	100.00	155.44	100.00

selective catalytic reduction (SCR) will grow faster, eventually covering 25% of the power units, and FGD devices will be further increased to 86%.[26] Because of the high Hg removal efficiencies of FFs, SCR, and FGD, there will be a good possibility for Hg reduction in the decade after 2010. Setting three coal consumption scenarios (high, medium, and low) and three control strategy scenarios (base, reference, and strict), Zhao et al.[26] proposed nine thermal power scenarios. The authors' projection for total national Hg emission from coal-fired power plants in 2015 and 2020 were based on these nine scenarios.

Figure 6 shows the nine scenarios from 2005 to 2020. The year 2005 was set to be the benchmark year. According to Figure 6, a useful conclusion can be made that it is control strategies that determine the trend of Hg emission, whereas coal consumption mainly has an impact on the speed of the increase or decrease of Hg emissions. With the base control strategy, Hg emissions will continue to grow, although the L1 scenario seems to be relatively mild. The reference and strict control strategies can result in a declining trend. Under the most probable scenario (M2), total Hg emission will develop a reversed U-shaped curve and eventually be reduced to 130 t by 2020. Sharp decreases will occur with strict control strategies (H3, M3, and L3), with reductions of 13, 19, and 23% by 2015 and 26, 33, and 42% by 2020, respectively.

UNCERTAINTIES

Two crucial factors that determine the uncertainty in the inventory are the Hg content of coal and the efficiency of Hg control devices.

Considering a local or regional scale, Hg content of coal tends to be the most important factor. The national average Hg content lies in the range of 0.15–0.22 parts per million (ppm).[2,10–16] Therefore, the uncertainty range of this study would be −15% to +24% considering only the uncertainty of Hg content.

Field tests in China and abroad show that the average Hg removal efficiency of ESP would most likely lie in the range of 10–40% and that of ESP+FGD in the range of 60–80%.[27–32] On the basis of these values, the uncertainties in the inventory for the year 2005 and 2010 were calculated. Because FGD was not widely used in 2005, the uncertainty was relatively large when the efficiency of ESP varies and small when that of ESP+FGD varies. The results were the opposite when it came to 2010. Generally speaking, the uncertainty in the inventory is influenced more by the Hg content of coal than the efficiency of control devices.

The uncertainty of Hg emission factor (ΔMEF) can be calculated from eq 2. The results indicate that the uncertainties of the estimation for 2005 and 2010 are 40 and 42%, respectively.

$$\Delta MEF = (1 - P \cdot w) \cdot R \cdot \left(1 - \sum_{n} (\overline{C}_n \cdot \eta_n)\right) \cdot \Delta \overline{M} + \overline{M} \cdot (1 - P \cdot w) \cdot R \cdot \sum_{n} (\overline{C}_n \cdot \eta_n) \quad (2)$$

There still exists a considerable uncertainty in the Hg emission inventory of coal-fired power plants in China.

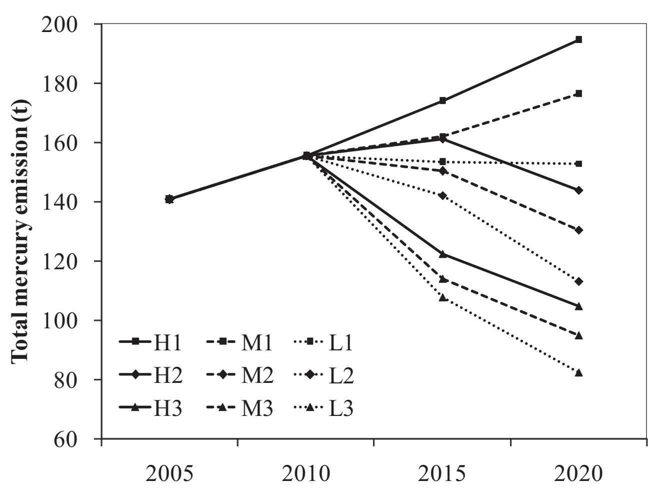

Figure 6. Total national Hg emissions (in t) under different scenarios from 2005–2020.

To further reduce inventory uncertainty, more emission tests and coal sample analyses are needed to understand the Hg content of coal and Hg removal in APCDs.

CONCLUSIONS

This study, for the first time, assessed the synergistic effects of conventional pollutant control devices on Hg removal. In addition, the co-benefit of emission control strategies (especially those applied in the 11th 5-yr plan in China) and the projected scenarios for Chinese coal-fired power plants from 2010–2020 were analyzed. The Hg content database from different studies in China was summarized and that of coal as burned in power plants was calculated based on a coal transport matrix. The national average Hg content of consumed coal was 0.18 mg/kg in 2005. ESP devices in Chinese power plants have lower Hg removal efficiency, which is probably because of the coal quality, especially the chlorine content and the LOI of coal in China. The value of 22% derived from emission tests in a Chinese power plant was used in this study. The efficiency of the device combination ESP+FGD was 70% based on Chinese studies, close to the results from the United States, Japan, and South Korea.

The Hg emission inventories for 2005 and 2010 were calculated. At the national level, the total Hg emission will rise by 10% from 2005–2010 when the coal consumption increases by 59%. Hg emission per ton of coal will drop from 126 to 87 mg. Strategies for FGD device installation in newly built units, existing units, and closing of small units contribute 59, 35, and 6% to the total Hg removal, respectively. FGD device installation in newly built units turns out to be the most effective strategy in the 11th 5-yr plan for Hg removal in coal-fired power plants in China. More efficient Hg capture devices such as FFs and SCR will be widely installed from 2010–2020. Six of the nine scenarios show a descending line from 2010 to 2020. Sharp decreases will be accomplished with strict control strategies (H3, M3, and L3). By 2020, total Hg emission will be reduced to 130 t under the most probable case, which is the reference control scenario at medium activity level (M2).

The uncertainty of the Hg emission inventory in this study is 35–43%, revealing that reliability has been improved with updated testing and data from the literature. However, the diverse Hg content of coal and the various removal efficiencies for control device combinations contribute to the considerable uncertainty in the inventory. More coal samples from different power plants should be analyzed, and in the meantime more tests should be conducted to specify the Hg removal mechanisms of conventional APCDs in coal-fired power plants in China.

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