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Evaluation of costs associated with atmospheric mercury emission reductions from coal combustion in China in 2010 and projections for 2020☆

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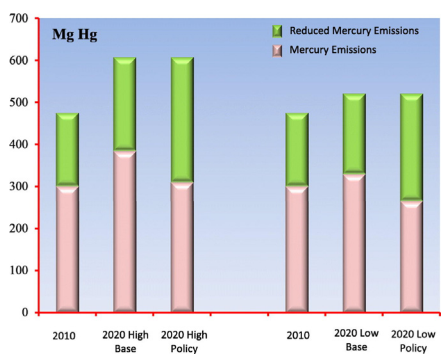
HIGHLIGHTS

- Mercury abatement costs for coal combustion in China for 2010 were estimated.
- Four scenarios were used to project mercury abatement costs for 2020.
- Decrease in unit abatement costs in 2020 suggests viable of various scenarios.

GRAPHICAL ABSTRACT



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ABSTRACT

Coal combustion is the most significant anthropogenic mercury emission source in China. In 2013, China signed the *Minamata Convention* affirming that mercury emissions should be controlled more strictly. Therefore, an evaluation of the costs associated with atmospheric mercury emission reductions from China's coal combustion is essential. In this study, we estimated mercury abatement costs for coal combustion in China for 2010, based on a provincial technology-based mercury emission inventory. In addition, four scenarios were used to project abatement costs for 2020. Our results indicate that actual mercury emission related to coal combustion in 2010 was 300.8 Mg, indicating a reduction amount of 174.7 Mg. Under a policy-controlled scenario for 2020, approximately 49% of this mercury could be removed using air pollution control devices, making mercury emissions in 2020 equal to or lower than in 2010. The total abatement cost associated with mercury emissions in 2010 was 50.2×10^9 RMB. In contrast, the total abatement costs for 2020 under baseline versus policy-controlled scenarios, having high-energy and low-energy consumption, would be 32.0×10^9 versus 51.2×10^9, and 27.4×10^9 versus 43.9×10^9 RMB, respectively. The main expense is associated with flue gas desulfurization. The unit abatement cost of mercury emissions in 2010 was 288×10^3 RMB/(kg Hg). The unit abatement costs projected for 2020 under a baseline, a policy-controlled, and an United Nations Environmental Programme scenario would be 143×10^3, 172×10^3 and 1066×10^3 RMB/(kg Hg), respectively. These results are much lower than other international ones. However, the relative costs to China in terms of GPD are

☆ Capsule: Mercury abatement costs for coal combustion in China was estimated and projected, which accounted for 0.14% of GDP in 2010, and 0.03%–0.06% in 2020.

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higher than in most developed countries. We calculated that abatement costs related to mercury emissions accounted for about 0.14% of the GDP of China in 2010, but would be between 0.03% and 0.06% in 2020. This decrease in abatement costs in terms of GDP suggests that various policy-controlled scenarios would be viable.

1. Introduction

Mercury is a persistent environmental pollutant. It causes global concern because of its long-range transport and high toxicity (Schroeder and Munthe, 1998; Streets et al., 2005; Pirrone and Mason, 2009). Previous studies showed that China's mercury emissions from anthropogenic sources have reached 600 Mg/y (Streets et al., 2005; Pirrone and Mason, 2009; A. Wu et al., 2006; Y. Wu et al., 2006; E.G. Pacyna et al., 2010a; J.M. Pacyna et al., 2010b), accounting for approximately 28–40% of global emissions (Pacyna et al., 2006; Pirrone and Mason, 2009). E.G. Pacyna et al. (2010a) and J.M. Pacyna et al. (2010b) estimated that the 635 Mg emissions in China in 2005 would decrease to between 380 Mg (under the Extended Emissions Control scenario) and 290 Mg (under the Maximum Feasible Technology Reduction scenario) in 2020. This last estimate assumes that all Chinese power plants will be equipped with improved emission control installations by 2020. If improvement is 50% lower than this estimate, then under various scenarios, China's emissions will increase rather than decrease by 2020, related to its ongoing economic development.

Coal combustion is believed to be the largest anthropogenic mercury emission source, producing approximately 24–66% of global emissions, with coal being consumed mainly in power plants and industrial boilers (Pacyna et al., 2006; United Nations Environment Programme (UNEP), 2002, 2013a, b). Pacyna and Pacyna (2002) estimated that China's mercury emissions from coal burning contributed to more than 25% of the total global emissions. Furthermore, A. Wu et al. (2006) and Y. Wu et al. (2006) showed that mercury emissions from coal combustion in China increased from 202 Mg in 1995 to 257 Mg in 2003, with an annual growth rate of 3.0%. Zhang et al. (2015) estimated that mercury emissions from coal combustion in China reached 254 Mg in 2010, accounting for approximately 47% of the national total emissions.

As one of the major mercury emitters, China signed the *Minamata Convention* in 2013, affirming that mercury emissions from coal-fired power plants and industrial boilers should be strictly controlled. Therefore, an evaluation of costs associated with atmospheric mercury emission reductions from China's coal combustion is essential. It would also support development of mercury-related environmental policies.

Several studies on abatement costs of mercury emissions, conducted in Europe and North America, showed a marked decrease in emissions since 2000 (Pirrone et al., 2001; USEPA, 2002; Visschedijk et al., 2006; E.G. Pacyna et al., 2010a; J.M. Pacyna et al., 2010b). However, few studies have been carried out in China. Wu et al. (2011) calculated the mercury abatement costs of air pollution control devices (APCDs) in China's power plants based on other pollutants' reduction costs. The cost data used in a number of studies originated from developed countries. Because of the differences in economic development levels among countries, the costs for China may differ significantly from other developed countries. Furthermore, large uncertainties are associated with previous estimations for China, because they did not consider detailed mercury emission inventories.

In this study, mercury abatement costs for coal combustion in China in 2010 were estimated, based on a provincial mercury emission inventories. Different types of coal, industries, and APCDs are considered in this estimation. Using updated installation and operation costs for APCDs, both total and unit abatement costs were calculated. In addition, two scenarios, namely a baseline and a policy-controlled scenario were proposed to describe mercury control policies for 2020. Two other scenarios, namely a high-energy and a low-energy consumption scenario were developed to describe energy consumption in 2020. Based on these scenarios, abatement costs for 2020 were estimated. Results of these estimations were compared with other reported costs (United Nations Environment Programme (UNEP), 2013a, b; E.G. Pacyna et al., 2010a; J.M. Pacyna et al., 2010b; Wu et al., 2011).

2. Data sources and methodology

2.1. Installation rate and removal efficiency of air pollution control devices

As environmental regulations in China becoming increasingly stringent, the installation rate of APCDs in power plants and other industries has grown rapidly, especially in the last few years. Currently, widely used APCDs in China include electrostatic precipitators (ESPs), fabric filters (FFs), and flue gas desulfurization (FGD). Only a few sites have selective catalytic reduction (SCR), while none use activated carbon injection (USEPA, 1997; UNEP, 2002; E.G. Pacyna et al., 2010a; J.M. Pacyna et al., 2010b; United Nations Environment Programme (UNEP), 2013a, b).

In some previous studies, the installation rate of APCDs in coal-fired industries was recognized to be equivalent to that in coal-fired power plants (United Nations Environment Programme (UNEP), 2013a, b). However, given the high installation and operation costs for large numbers of industrial boilers, the actual installation rate of APCDs in coal-fired industries is much lower than for power plants (NBSDE, 2011). Zhang et al. (2015) showed that only some large-capacity boilers in China have adopted a combination of FFs & FGD. Clearly, mercury emissions from coal-fired industries have been underestimated. The overall installation rate and removal efficiency of the whole coal-fired industry is lower than that in the power plants. In this study, we used real data for APCDs installation rates and estimated future development to carry out our scenario analysis.

Mercury removal efficiencies for different APCDs vary significantly (Wu et al., 2010; Wang et al., 2010; Zhang et al., 2012; United Nations Environment Programme (UNEP), 2013a, b). Detailed studies have been carried out on the combustion efficiencies of different devices, such as the capture of mercury in particulate control devices by unburned carbon (Hower et al., 2010; Chen et al., 2007). In our study, the information of removal efficiencies came from previous studies, and the removal efficiencies of the individual techniques have been considered in their calculations.

The detailed installation rate and mercury removal efficiency of each combination of APCDs in 2010 are listed in Table S1. We calculated the weighted equivalent (average) removal efficiencies for each industry, which are about 60.22% and 4.30% for coal-fired power plants and industry boilers, respectively. Since it is difficult to obtain detailed emission and control information of different coal types, and the production volume of some coal types such as lignite is not high, we assumed that the type of coal used had no significant impact on mercury removal (Zhang et al., 2012).

2.2. Emission and reduction factors

Typically, the mercury concentration of raw coal is used as the primary emission factor (EF) in calculating mercury emissions from coal combustion. In our study, mercury concentrations of raw coal were obtained at a provincial level from Streets et al. (2005) (Table S2). According to previous research, not all the mercury in the fired-coal is released into the atmosphere. Usually, release rates are related to the type of industrial boiler. In this study, the release rates for power plants, coking industry, and other industries were taken as 99%, 63% and 83%, respectively (Streets et al., 2005; Jiang et al., 2005). Thus, the equivalent EFs for raw coal for different industries and provinces were obtained by multiplying the provincial mercury concentrations in 2010, with their release rates, and emission efficiencies (Table S3).

In addition to raw coal, fired-coal in China includes washed coal and briquette coal (NBSDE, 2011). The mercury concentration of washed coal can be calculated using the quantity of washed coal, the output of washed coal, and the removal efficiency of the washing process (30%; Streets et al., 2005; Pacyna et al., 2010b; Wu et al., 2010). In contrast, because briquette coal is machined directly from raw coal, its mercury concentration is unchanged. Therefore, based on their equivalent EFs in terms of raw coal, we also obtained EFs for washed and briquette coal for different industries and provinces in 2010.

Similarly, by replacing the emission efficiency with removal efficiency in our EF-calculation, we calculated the equivalent reduction factors (RFs) for raw coal, washed coal, and briquette coal for different industries and provinces in 2010.

2.3. Emission and reduction calculations

For bottom-up atmospheric mercury emission inventories, the EF method is widely used. In this study, the provincial-level mercury emissions and reductions related to coal combustion in China were calculated by multiplying the corresponding amounts of consumed coal and the equivalent EFs/RFs for different industries for each province, using Eq. (1) and Eq. (2).

$$E = \sum\sum(EF_{i,j,m}A_{i,j,m}) \tag{1}$$

$$R = \sum\sum(RF_{i,j,m}A_{i,j,m}) \tag{2}$$

where E is mercury emissions from coal combustion, R is mercury reductions from coal combustion, EF is the equivalent emission factor, RF is the equivalent reduction factor, A is the amount of consumed coal, while the parameters of i, j and m represent province, industry and coal type, respectively. Here, the amounts of consumed coal in 2010 were obtained from the China Energy Statistical Yearbook (NBSDE, 2011), which provides information for various provinces, industries and coals.

2.4. Abatement cost calculation

Various APCDs are primarily used to control major air pollutants such as SO_2, NO_x and particles, and the removal of mercury is identified as a “co-benefit”. However, it is difficult to differentiate the contributions of a control technique in different aspects. In this study, we assume the future installations are made for the purpose of mercury removal (Wu et al., 2011).

Here, the total abatement costs of mercury reductions from coal combustion were obtained by multiplying the annual operation cost and the installed power of any APCDs. The unit abatement costs were calculated by dividing the total cost of mercury reduction by the number of tons of mercury. We used annual exchange rate data from the National Bureau of Statistics; we also used 1% as the discount rate E.G. Pacyna et al., 2010a; J.M. Pacyna et al., 2010b). The cost calculation of APCDs in thermal power industries was based on 300 MW-units, with an annual operation time of 5000 h. For other coal-fired industries, we assumed that the abatement cost was equal to that of the thermal power industry (Zhang et al., 2015; Yu, 2012). The power values were obtained from coal consumption associated with these industries, or else the power value of the power plant was used. Consistent with previous studies (Chen et al., 2007; Li, 2011; Hao et al., 2005; Yan et al., 2008), including the *Thermal Power Project Cost Control Index of China Power Investment Corporation* (2011), we selected 30 years as the economic life span of both ESPs and FFs, but 15 years as the economic life span of FGD and SCR. Our calculation of the annual operating costs of APCDs in China for 2010 is given in Table S4; they are clearly lower than previous results.

2.5. Scenario calculations

To estimate mercury emissions and their abatement costs for 2020 for China's coal-fired industries, four scenarios were explored. Two scenarios, namely a baseline and a policy-controlled scenario were proposed to describe mercury control policies for 2020. Two other scenarios, namely a high-energy and a low-energy consumption scenario were used to describe energy consumption for 2020.

In the baseline scenario, we assumed that China's mercury control policies in 2020 would be unchanged. In this case, the installation rates of APCDs in 2020 would be equal to those in 2010 and the same EFs can be used. In the policy-controlled scenario, we assumed that a stricter mercury control policy would be adopted in 2020, resulting in a higher installation rate of APCDs and lower equivalent EFs. Given that applications for FGD has rapidly increased from 67% in 2008 to 82% in 2010, while SCR has gone from 9% in 2009 to 18% in 2011 (CEPYEB, 2009–2012), we forecast that installation rates of ESP/FF + FGD and SCR will be 100% and 50% in 2020 (Pacyna et al., 2010b). Detailed installation rates for APCDs and their equivalent EFs under a policy-controlled scenario are listed in Tables S1 and S5, respectively.

In both high- and low-energy consumption scenarios, we assumed that China's per capita energy consumption will reach 5600 kwh and 4800 kwh in 2020, whereby coal consumption will account for about 60%. We also assumed that coal consumption per unit of electricity generation will decrease from 333 g/kw in 2010 to 310 g/kwh in 2020 (UNEP, 2011). We estimated the coal consumptions of industries under both high- and low-energy consumption scenarios in 2020, based on the actual coal consumption information in 2010 (Tables S6 and S7). Assuming no change in technology, the abatement costs for these scenarios only reflect the discount rate of 8% (*Thermal Power Project Cost Control Index of China Power Investment Corporation*, 2011). The operation costs of APCDs under these 2020 scenarios are given in Table S4.

2.6. Uncertainty analysis

The uncertainties of the inventories were evaluated using Monte Carlo simulations. Variations in mercury content of coal, EFs and RFs, coal consumption, operation costs, and installed power of APCDs were considered in our simulations. Input parameters used in the calculations were randomly selected from the corresponding statistical distributions. The mercury contents of coal consumed in each province were considered to have lognormal distributions (Wu et al., 2011), having a coefficient of variation (CV) of 50%. The activity level of thermal power industry was assumed to be a normal distribution, with a CV of 5% (Zhao et al., 2011), while the activity level of other industries was assumed to be a normal distribution with a CV of 20% (Zheng et al., 2007). In our estimations for 2020 scenarios, the CV values of energy consumption were doubled. The reduction efficiency of APCDs was assumed to be a normal distribution having a CV of 20% (Zhao et al., 2011), while other parameters were assumed to have uniform distributions. By performing 10,000 Monte Carlo simulation iterations, a range of mercury emissions (with a 90% confidence interval) was derived.

3. Results and discussion

3.1. Mercury emissions in 2010 and projections for 2020

The provincial mercury emissions and reductions from China's coal-fired industries for 2010 are shown in Table S8. The original mercury emissions from coal combustion in 2010 were 475.5 Mg. After taking into account APCDs, the atmospheric mercury emissions were estimated at 300.8 Mg, yielding a mercury reduction of 174.7 Mg. Of the total emissions, about 84% are from raw coal combustion, while 15% are from washed coal. Compared with other studies (Zhang et al., 2015), our estimates are slightly higher, reflecting the EFs and activity levels used.

In terms of different industries, mercury emissions from the thermal power industry were 110.7 Mg, with corresponding reductions of 167.6 Mg. Emissions and reductions from general coal-fired industries were 156.8 and 7.0 Mg, respectively. Although coal consumption is largest for the thermal power industry, its higher installation rate of APCDs led to lower mercury emissions. These results show that approximately 97% of mercury reductions were achieved by the thermal power industry. The top five provinces for mercury emissions were Shandong, Henan, Guizhou, Inner Mongolia and Shanxi, accounting for 40% of the China's total emissions. High coal consumption (such as Shandong and Henan) as well as high mercury content of coal (such as Guizhou and Yunnan) are the main reasons for their high mercury emissions.

Mercury emissions and their corresponding reductions for China's coal-fired industries under various 2020-scenarios are shown in Table 1. Under high- and low-energy consumption, mercury emissions would reach 607.8 and 521.5 Mg, reflecting a coal consumption increase of 30% and 10% from 2010 values. Emissions calculated after adopting APCDs under a high-energy consumption baseline versus policy-controlled scenarios, and low-energy consumption baseline versus policy-controlled scenarios would be 384.5 Mg versus 310.1 Mg, and 329.9 Mg versus 266.1 Mg; their corresponding reductions would be 223.3 Mg, 297.7 Mg, 191.6 Mg, and 255.4 Mg, respectively.

Mercury emissions and reductions from China's coal-fired industries in 2010 and under various 2020-scenarios are compared in Fig. 1. Under baseline scenarios, installation rates of APCDs were unchanged, causing mercury emissions under high- and low-energy consumption to reach 384.5 Mg and 329.9 Mg; these values are 83.7 Mg and 29.1 Mg higher than values for 2010. Under policy-controlled scenarios, 49% of the mercury is removed by APCDs, which improves on 2010 levels. Thus, mercury emissions related to coal combustion in 2020 could be equal to or lower than 2010 under policy-controlled scenarios because of the effect of APCDs.

3.2. Abatement costs in 2010 and projections for 2020

The total abatement cost for China's coal-fired industries in 2010 was 50.2×10^9 RMB (Table 2). This cost mainly arises from the investment and operation of FGD (63%). Although the installation rate of ESPs was highest, its abatement costs only accounted for 24% of the total cost, because its annual investment and operation costs are low. Conversely, although the installation rate of SCR was low, its abatement cost comprises a greater proportion of the total cost (10%). The unit abatement cost was 288×10^3 RMB/(kg Hg) in 2010, which is much lower than the reported international cost of 1066 RMB/(kg Hg) (E.G. Pacyna et al., 2010a; J.M. Pacyna et al., 2010b). Thus, the abatement cost comprised about 0.14% of the GPD in 2010.

Table 1
Mercury emissions and the corresponding reductions of coal combustion in China under various 2020-scenarios (t).

	Thermal power plant		General industry (including coking)		Others	Total	
	E[a]	R[b]	E	R	E	E	R
2020-high-base	141.5	214.3	200.5	9.0	42.5	384.5	223.3
2020-high-policy	112.2	243.6	155.4	54.0	42.5	310.1	297.7
2020-low-base	121.4	183.8	172.0	7.7	36.5	329.9	191.6
2020-low-policy	96.3	209.0	133.4	46.4	36.5	266.1	255.4

[a] E = emission.
[b] R = reduction.

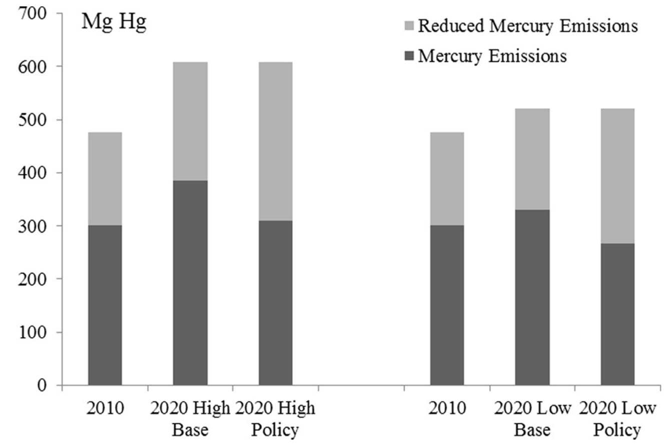

Fig. 1. Comparison of mercury emissions and reductions related to coal combustion in China under different scenarios in 2010 and 2020.

High abatement costs were associated with Shandong, Inner Mongolia, Jiangsu, Henan and Shanxi (Fig. 2), accounting for about 39% of the China's total costs. These provinces all had high coal consumption. Guizhou and Yunnan had large mercury reductions, but the high mercury content of their coals and low consumption led to low abatement costs. The unit abatement costs were highest in Xinjiang, Qinghai, and Gansu (Fig. 3), although the mercury content of their coal is relatively low. The unit cost in Xinjiang was 2637×10^3 RMB/(kg Hg), which is much higher than the national average of 288×10^3 RMB/(kg Hg). The abatement costs for China's coal combustion in terms of GDP are shown in Fig. S1. Highest percentages were associated with Ningxia, Inner Mongolia, Shanxi, Guizhou, and Gansu.

The total abatement costs of China's coal combustion under various scenarios in 2020 are presented in Table 2. The total abatement costs for 2020 under a high-energy consumption baseline versus policy-controlled scenarios, and low-energy consumption baseline versus policy-controlled scenarios would be 32.0×10^9 versus 51.2×10^9, and 27.4×10^9 versus 43.9×10^9 RMB. The unit abatement costs under baseline and policy-controlled scenarios would be 143×10^3 and 172×10^3 RMB/(kg Hg), respectively. Given that the installation rates of APCDs would be higher under policy-controlled scenarios, the abatement cost for high- and low-energy consumption scenarios would be much higher than equivalent baseline scenarios.

Under a baseline scenario, the abatement costs of ESPs, FFs, FGD, and SCR would account for about 24%, 3%, 63% and 10% of the total costs, respectively. The cost share of FGD would be highest. Although the annual investment and operation costs of SCR would be equal to FGD, the low installation rate of FGD results in a low abatement cost, even lower than for ESPs. However, the shares of abatement costs of ESPs, FFs, FGD, and SCR under a policy-controlled scenario would be 15%, 5%, 57% and 23% of the total cost, respectively. The cost shares of FFs and SCR would increase compared with a baseline scenario. Given that the installation rate of ESPs has been saturated, this would result in its cost percentage decreasing from 24% to 15%, while a slight increase in FF installation rate would lead to its cost share rising from 3% to 5%. The installation rate of FGD in 2020 would be higher than 2010, but

Table 2
Total abatement cost for mercury emissions from coal combustion in China under different scenarios in 2010 and 2020 (10^8 RMB).

	ESP	FF	FGD	SCR	Total
2010	120.4	15.1	316.5	50.5	502.4
2020-high-base	76.6	9.6	201.3	32.1	319.5
2020-high-policy	76.6	27.7	292.6	114.6	511.5
2020-low-base	65.7	8.2	172.7	27.5	274.2
2020-low-policy	65.7	23.7	251.1	98.3	438.8

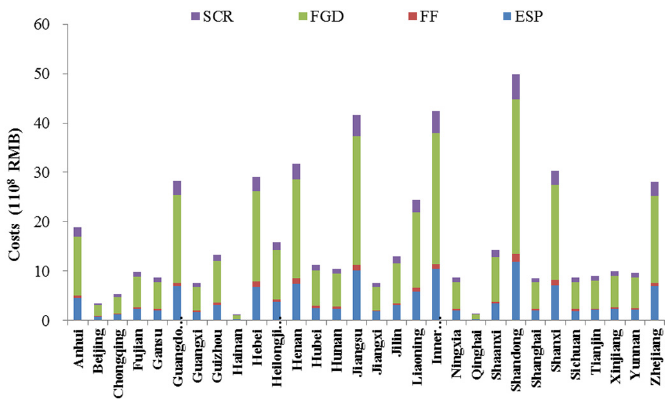

Fig. 2. Total abatement cost for mercury emissions from coal combustion in China by province for 2010.

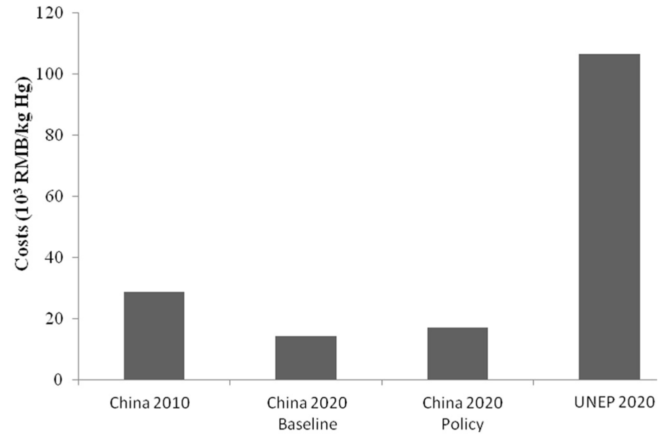

Fig. 4. Abatement costs of per unit of mercury in coal-fired plants under different scenarios in 2010 and 2020.

SCR installation rate would undergo a stronger increase, from 14% to 50%. Therefore, the cost share of FGD would decrease from 63% to 57%, while the cost ratio of SCR would rise. Under a strict policy-controlled scenario, the main investment in mercury reductions for China's coal-fired industries would be in FGD, followed by SCR. The investment in SCR would increase markedly.

3.3. Comparison among countries and different years

The unit abatement costs for 2010 and 2020 calculated in this study were compared with projections for 2020 from the United Nations Environment Programme (UNEP) report in Fig. 4. The unit abatement costs under both baseline and policy-controlled scenarios in 2020 were lower than those in 2010. Compared with the abatement cost under the EXEC scenario in the UNEP report, the unit abatement cost for China in 2010 was about 26% that of the international average. Under baseline and policy-controlled scenarios in 2020, the abatement costs for China would be only 13% and 16% of the international average. This decline in ratios is mainly caused by the differences in discount rates, being 1% in Europe and the US, but 8% in China. Furthermore, the combination of APCDs in the UNEP report was ESP/FF + FGD, while more complex APCD combinations were explored in this study. Using the same technology combination as the UNEP report, the abatement cost of ESP + FGD for China's coal-fired power plants in 2010 was calculated to be 346×10^3 RMB/(kg Hg), about 1/3 of the international average. Similarly, the cost in 2020 would be 160.4 RMB/(kg Hg), only 1/6 of international average. It is clear that China's mercury reduction is more economical than for many other countries.

In contrast, the per capita GDP of China in 2010 was only about 10% of most developed countries. Thus, the relative abatement costs for China would be higher than for most developed countries. Given that the mercury removal efficiencies of APCDs in China were lower than in many developed countries, there is still scope for technological improvement in China.

Using GDP data for China from the World Bank, the proportion of the abatement costs in terms of GDP for coal combustion in 2010 and 2020-scenarios was calculated (assuming an annual growth rate of GDP of 8% over the next decade). Clearly, the abatement costs are about 0.14% of the GDP in 2010, but would be between 0.03% and 0.06% in 2020 (Fig. 5). The decrease of abatement costs in terms of GDP shows that all policy-controlled scenarios are viable.

3.4. Uncertainty analysis

Uncertainties of our results were estimated using Monte Carlo simulations. The uncertainty ranges for mercury emissions and reductions from coal combustion were large; their lower bounds varied between −44% and 47%, while their upper bounds varied between 80% and 92%. This shows that mercury content in raw coal contributes to large uncertainty. In contrast, the uncertainty ranges for total abatement costs of mercury emissions were small; their lower bounds varied between −35% and 43%, while their upper bounds varied between 39% and 47%. Here, annual costs of FGD contribute to uncertainty in the total costs. The uncertainty ranges for the abatement costs of per kilogram mercury were very large; their lower bounds varied between

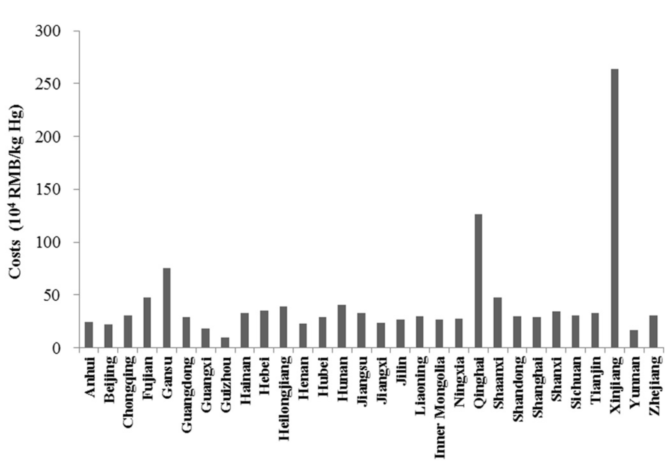

Fig. 3. Unit abatement cost for mercury emissions from coal combustion in China by province for 2010.

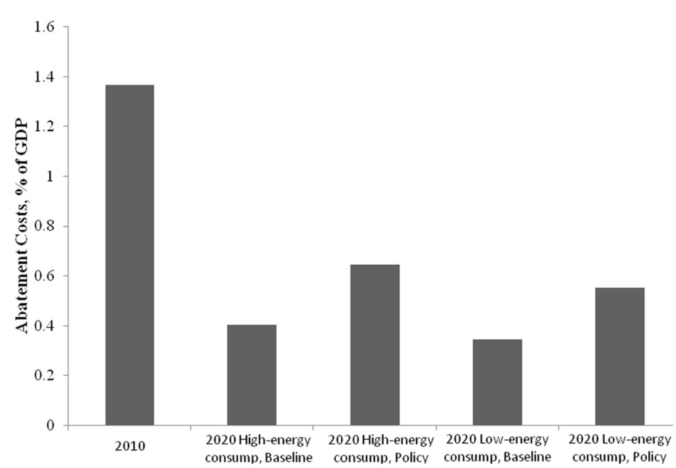

Fig. 5. Abatement costs for mercury emissions as a share of GDP in coal-fired plants under different scenarios in 2010 and 2020.

−52% and 56%, while their upper bounds varied between 99% and 110%. Thus, the mercury content in raw coal as well as the annual costs of FGD cause large uncertainty in estimating the abatement costs of per kilogram mercury.

4. Conclusions

In 2010, mercury emission related to coal combustion in China was 300.8 Mg, and its reduction amount was 174.7 Mg. Approximately 37% of this mercury was removed by APCDs. These data were used project mercury emissions and reductions for 2020. Under a baseline scenario for 2020, the mercury emissions under high and low-energy consumption would increase to 384.5 and 329.9 Mg. In contrast, under a policy-controlled scenario, the corresponding mercury emissions would be 310.1 and 266.1 Mg, respectively. We estimated that about 49% of mercury would be removed by APCDs, resulting in emissions in 2020 being equal to or lower than in 2010.

The total abatement cost of mercury emissions in 2010 was 50.2×10^9 RMB. The total abatement costs for 2020 under high-energy consumption baseline versus policy-controlled scenarios, and low-energy consumption baseline versus policy-controlled scenarios would be 32.0×10^9 versus 51.2×10^9, and 27.4×10^9 versus 43.9×10^9 RMB. The input is spent mainly on FGD, which accounts for about 63% and 57% of the abatement costs under baseline and policy-controlled scenarios. The unit abatement cost of mercury emissions in 2010 was 288×10^3 RMB. The unit abatement costs for 2020 under baseline, policy-controlled and the UNEP scenarios would be 143×10^3, 172×10^3 and 1066×10^3 RMB, respectively. Our results for China are much lower than the international average. However, the relative costs in China were higher than for most developed countries, taking per capita GDP into consideration. The abatement costs are about 0.14% of the GDP of China in 2010, but would be between 0.03% and 0.06% in 2020. The decrease in the GDP share of abatement costs shows that policy-controlled scenarios could be viable.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2017.08.007.

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