Effect of mercury emissions in China on North America

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Abstract

In general, it has been shown that the economic benefits of industrialization have deleterious effects on the quality of the environment. Political regulation, however, often ameliorates the most serious problems, but at a substantial cost. In a study of this type, the effect of mercury emissions by China on North America (NA) and, in particular, the emissions from coal-fired power plants was investigated. A Hg inventory for China indicated that China emits ten times more total Hg than the US and as much as four times more Hg per ton of coal burned. The intercontinental transport of this mercury was shown to increase the flux of Hg over the US up to four times. Pollution control technologies were reviewed and their costs were estimated. In response to recent legislation that would cap US power plant mercury emissions, two scenarios were proposed: one where the US enhances its existing fluegas cleanup technology and one where the US invests in first-generation pollution control in China. It is concluded that the first scenario will cost $28 – 37 billion and could reduce Hg fluxes over the US by 25 tons / yr. The second scenario will cost $41 – 73 billion and could reduce Hg fluxes over NA by as many as 323 tons / yr.

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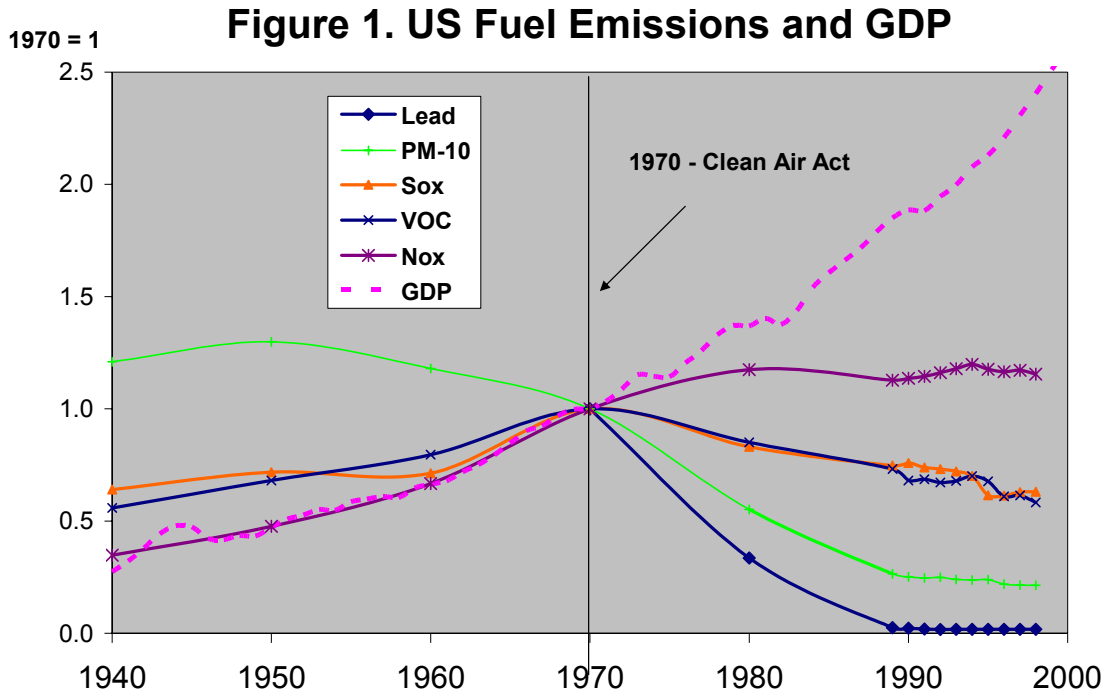
0. Introduction

Under the Clean Air Mercury Rule issued on March 15, 2005, in 1990, the EPA committed to forming a cap-and-trade system for Hg that would limit emissions from 48 tons per year to 15 tons per year by 2018.[1] Dr Nicholas Themelis calculated, this would cost the energy industry over $22 billion dollars.[2] As the United States is tightening its environmental regulations by fine-tuning its existing infrastructure, much of the rest of the world is seeking to hastily expand its energy production. Because economic necessities often trump environmental considerations, hazardous emissions levels in developing countries can be high. This paper investigates the relationship of global emissions to the air quality of the developed world. Perhaps, the billions of dollars being considered for tweaking US power plants could be more efficiently spent in reducing China's emissions.

1. Background

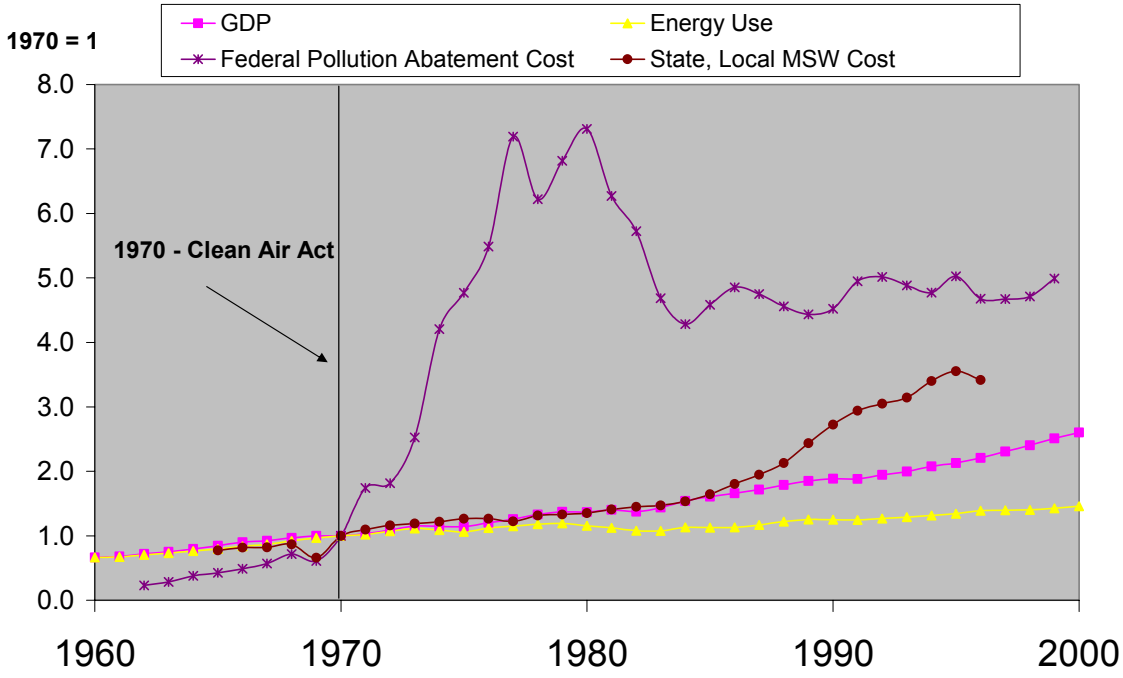
1.1 Environmental economics and political regulation

Intensified industrial production and energy consumption almost inevitably lead to higher pollution levels. The United States exhibited such a trend before the 1970s as can be seen by eye in Figure 1.[3,4,5] The data, normalized to 1970 output, shows that before 1970 most atmospheric emissions from fossil fuel burning steadily increased with GDP. Then, the country witnessed the passage of several effective laws at controlling air emissions including the Clean Air Act (1970) and the Resource Conservation and Recovery Act (RCRA, 1976). At this point, emissions of the most toxic elements and compounds substantially decreased. Emissions from particulate matter (PM-10) did not show an increase with GDP before 1970. However, their levels significantly decreased with regulation.



Since the first wave of environmental litigation in the US, net emissions have leveled off as better technology has been implemented according to the Maximum Achievable Technology (MACT) standard. Figure 2 plots in parallel the growth in GDP and energy use with the total government expenditures for both pollution abatement and solid waste management with the data normalized to 1970.[3,4,5] After regulation, a sharp jump in expenditures is witnessed in response to new environmental standards. This example supports the general theory that before government litigation, pollution generally increases with industrial activity, but after litigation it can be held roughly constant if the cost can be afforded.

Figure 2. US Pollution Cost and GDP



This tell-tale story is becoming more prominent globally as other countries seek the lifestyle that enhanced industrial activity allows. This is especially the case in China, a nation preparing to raise the basic standard of living for its 1.3 billion people. Absolute economic growth rates for the 1990s show that China's economy grew at a 7.5% annual rate compared with only 3.8% in US.[6,7] Thus, China seems to be undergoing a rapid expansion of its productive capacities. Consumption data shows that China is Asia's leading coal consumer, burning 58% of the continent's total.[8]

Already, there is a growing awareness of China's output and its environmental quality. A report by The Economist sites several measures of poor environmental quality in China including 300,000 premature deaths per year from respiratory ailments, the world's highest SO_2 emissions, and 2/3 of its cities failing to meet WHO criteria.[9] Several researchers cite that coal power plants are the largest single group responsible for poor air quality.[10] The Economist claims that 70% of China's energy needs are supplied by coal and many of the burners are decentralized or unregulated.[11]

While researchers like He have expressed concern for air quality within China, others have begun to investigate the effect of China's emissions on the rest of the world.

Akimoto states that by the 1990s, Asia produced more NO_x than North America and Europe. [12] Wilkening reviews the bulk of the current research into Pacific intercontinental transport of pollution focusing mostly on SO_x and ozone. Their preliminary conclusions indicate that significant impacts are present and are likely to increase as Asia expands its economic capacity.[13,14] Few studies, however, have investigated the intercontinental transport of pollution.

1.2 Mechanisms of intercontinental transport

Most measurements and theories to this point have focused on smog forming compounds and transport models. Berntsen finds that CO, O_3, and organic nitrate concentrations in Washington increase by 34 ppbv, 26 pptv, and 4 ppbv, respectively, as a result of springtime emissions in Asia.[15] Jacob finds similar conclusions about ozone enhancement in the Western United States and claims that the increased overseas pollution offsets a 25% reduction in national NO_x emissions.[16] Several investigations track individual pollution and dust events across the Pacific.[17] They conclude that transported gases follow episodic trends rather than enhanced background levels due to differing initial pollutant concentrations and interactions with varying amounts of dust particles.[18] Quantifying the error of these estimates is difficult however due to the complex factors involved in atmospheric transport.

Concerning mechanisms of intercontinental transport, some models have been proposed. Jaffe (2003) claims that Marine Boundary Layer (MBL) transport of ozone is insignificant because the residence time of ozone is shorter than the transport time near the surface. This finding does not exclude the possibility that other pollutants cannot be transported in the MBL. The Nowak study (2004) shows that transport of chemical plumes occurs at initial altitudes of 6-9 km and fall 2-3 km lower when they reach North America. In both the Nowak (2004) and Jaffe (2002) examples, these plumes take 8 days to traverse the ocean.

Though much work has examined ozone forming gases, relatively little has been performed on trace metals and, in particular, mercury which is recognized to be a major anthropogenic pollutant. The major source of atmospheric mercury appears to be from coal burning. As such increased use of coal might pose a significant risk to the air quality of North America and globally.

1.3 Atmospheric fate of mercury

When considering mercury, it is customary to perform separate treatments of the metallic (Hg^0), ionic (Hg^{+2}), and particulate (Hg^p) forms. Researchers estimate that typical dry deposition velocities lie in the range of .01 cm/s.[19] The net ground flux of mercury will be proportional to the atmospheric concentration times the deposition rate divided by the area to be considered. For intercontinental transport, only emissions of Hg^0 are considered because the other two mercury species have considerably short lifetimes during a rain event. A mild precipitation event (10 mm/hr for 4 hours) can reduce the Hg^{+2} and Hg^p by higher than 90%.[20] Although not every air plume is associated with a rain event, for these conclusions it is assumed that dry deposition of Hg^0 is the most prominent form of intercontinental transport.

Measurements and modeling have shown some preliminary results. A 1999 study conducted in Changchun City, revealed that average atmospheric concentration of Hg^p

was 0.145 ng/m^3 in the summer and 0.491 ng/m^3 in the winter. The higher value is attributed to increased coal combustion. Yearly average dry deposition net ground flux were theoretically calculated to be 45 μg/m^2.[21] As a comparison, a recent mercury recording site in Pennsylvania measured an average deposition flux for both wet and dry conditions of 0.021 μg/m^2 s for an annual deposition flux of 7.6 μg/m^2.[22] The residents of Changchun City appear to have a sixfold higher exposure than over the eastern US.

2. Analysis Methods

This study examines the mercury produced by the Chinese coal industry. An intercontinental transport model is used to determine the theoretical flux of Chinese derived mercury on the United States. Next, different pollution control scenarios are analyzed: one where the US upgrades its existing technology to MACT and another where China implements first-generation cleaning technologies.

3. Physical Model

3.1 Coal and mercury production

The amount of coal in use is well documented by several authorities including the Energy Information Administration (EIA) and the International Energy Agency (IEA). In 2002, China and the US consumed 47% of the world's coal, burning 1,290 and 966 million metric tons, respectively.[23] An estimate for the same year by the International Energy Agency (IEA) quoted that China and the US consumed 1,450 and 1050 million metric tons.[24]

Mercury emissions are not as clearly documented but some estimates have been proposed. A study by Pacyna mapped 1995 mercury emissions from every continent. Asia's emissions were determined to be 1,074 tons roughly half the world total. Most of the emissions derived from coal power plants (85%).[25] Seigner et al. estimate similar levels (1,117 tons/yr from Asia) in 1998.[26] The US has much lower emissions levels estimated to be 125.7 tons per year with only 40.9 tons coming from coal power plants.

A Hg concentration in coal is derived by dividing EIA coal consumption information with data from both mercury papers and is compiled in Table 1. The numbers in parentheses show the mercury emissions estimated from coal plants. When no data of coal-derived mercury was available, the total Hg emissions were used providing a possibly inflated concentration. Coals from China appear to possess 2-6 times the mercury content of coals from the US and Europe.

Table 1. Estimates of coal consumption and mercury emissions for 1995 and 1998.

	Hg Emissions*	Coal Consumption**	Hg Conc.	Hg Emissions***	Coal Consumption**	Hg Conc.
Asia	1117.2	2,300.66	485.60	1074.3 (860.4)	2,383.39	361.00
China	---	1,262.83	---	---	1,358.54	---
N. Am.	173.9 (52.1)	1,013.78	51.39	213.5 (104.8)	937.01	111.84
US	125.7 (40.9)	940.86	43.47	---	872.82	---
Europe	326.8	1,042.36	313.52	249.7 (185.5)	1,115.82	166.25
	tons/yr	Mil met tons/yr	ppb	tons/yr	Mil met tons/yr	pbb

Sources: * Seigner C, Karamchandani P, Lohman K, Vijayaraghavan K, Shia RL. "Multi-scale modeling of the atmospheric fate and transport of mercury" Journal of Geophysical Research (106 No. D21) 2001, p. 795.

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Numbers in parentheses indicate Hg derived from stationary combustion (excluding waste incineration). When this information was not available, the total Hg output was used to calculate a concentration

According to survey of Chinese coal composition, the average Hg content of Chinese coal is 303 ppb.[27] This figure is still higher than US coals but, the emissions data suggests that the concentration in Chinese coals may be twice as high. It is interesting to speculate on the cause of the disparity. A geologically-based hypothesis is conceivable. Another proposition may be that the US is more effective at cleaning Hg from its fluegas before it can be emitted into the atmosphere. The latter idea will be explored in more detail in section 4.

3.2 Deposition of Hg on North America

The deposition of Hg on the NA will be estimated by counting the amount of Hg^p and Hg^{+2} it emits and the amount of Hg^0 that China emits. This model assumes that the residence time of Hg^p and Hg^{+2} is short while the residence time of Hg^0 is long. These assumptions can be justified by using the simplification that nearly all non-Hg^0 settles via wet deposition. A quick calculation using a Hg^0 dry deposition velocity is 0.5 cm/s indicates that the lifetime of mercury in the upper troposphere (10 km) would be 11.6 days.[28] This is a similar time-scale as the transport time of material across the Pacific suggesting that Hg^0 should just be reaching the ground when it falls over the US. From information in 3.1, the amount of Hg^0 emitted by China can be estimated at half the total emissions at 537 tons. The amount on Hg^p and Hg^{+2} emitted by NA can also be estimated as half the total Hg emissions at 63 tons. China may increase the US mercury deposition by up to a factor sixteen if it is assumed that all the Hg^0 deposits on NA in the first pass. The magnitude of such an increase is not unreasonable. Bergan et al. (1999) showed that global Hg deposition increased 2-10 times since the start of the industrial revolution.[29]

3.3 Comparison with measurements

Several experimenters have tried to measure increases in Hg fluxes over the US using techniques from more prevalent studies that measure ozone-forming pollutants.

Steding and Flegal (2002) collected rainwater samples in coastal California. They claim that a fourfold depositional enrichment of an isolated Marine Lab over the ocean's background flux is the result of Asian emissions.[30] Though the result mirrors the prediction above, this work may have some flaws however as noted by published comments.[31]

For this work, a study using Hg flux data from the Mercury Deposition Network was performed.[32] According to the 2003 maps, Hg deposits most heavily in the southeastern United States where wet deposition rates vary from 15 – 28 $\mu g/m^2$ and rainwater concentrations range from 10 – 16 ppb. The Northeast sees more modest fluxes of 6 – 10 $\mu g/m^2$ and concentrations of 5 – 8 ppb. Data for the Western United States was not as extensive, however, the average for a site in California and one in Washington fall at the low end of the range.[33]

Fang (2001) measured values in a Chinese industrial city that were significantly higher with a year-average rainwater concentration of 303 ppb.[34] Of note, the Hg rainwater concentrations in Asia are ~30 times higher than in NA, while the total emissions rate was estimated to be only 10 times higher. The difference might be accountable by considering that the measured depositions only involve non-inert Hg. Since fluegas cleanup systems can better remove the interactive Hg, the there will be proportionally less Hg available for local wet deposition in NA where more cleanup systems are in place.

4. Economics and Efficiency

4.1 Types of mercury removal systems

The EPA lists several options for removing Hg from emissions.[35] Mercury can be removed from the process both before coal burning and after. Conventional (US) coal cleaning (CC) methods such as more efficient removal of waste rock can reduce Hg by up to 55%. Using MACT for CC can reduce Hg by up to 70%. On the back-end, some Hg can be controlled with conventional technologies (ESPs: up to 50%) because some active Hg can adsorb to particulate matter. Wet scrubbers (WS) can reduce Hg^p and Hg^{+2} by up to 90%. Fabric filters (FF) can also remove Hg adsorbed to particles. Catalytic NO_x reformers and residual HCl can enhance the oxidation of Hg^0, but the presence of SO_x will reduce Hg^{+2} to Hg^0. Activated carbon injection (CI) and selenium filters, both novel and expensive treatments, can also significantly eliminate Hg. A MACT pollution control system can remove 98% of the Hg from air emissions. Table 2 summarizes the effect of the major components.[36]

Table 2. Percent of Hg removed by each component.

Process	CC	ESP	WS	FF	CI	EPA
Effect	50%	50%	75%	*	90%	98%

*The FF must be coupled to a sorbent.

4.2 Costs of mercury removal

Two measures were used to determine the cost of new clean-up systems. First, the EPA proposed a system that includes dry scrubbing, CI, and FF.[37] Information on the costs of individual components was taken from the World Bank (WB).[38] The results are listed in Table 3. It should be noted that the EPA assesses a lower cost for the system

than the WB predictions. Nonetheless, the estimate remains within a factor of three so both are acceptable.

Table 3. Cost of fluegas cleanup components. Number on left is for a 975 MW plant; The number on right is for a 100 MW plant.
(Assumes 7% interest over 20 years)

EPA	975 MW plant	100 MW plant	
Capital	$33,700,000	$4,560,000	
OM	$7,490,000	$1,290,000	/yr
NPV	$113,049,167	$18,226,278	
$/kW	$116	$182	

Electrostatic Precipitator (ESP)			
Capital	$39,000,000	$6,000,000	
OM	$11,128,339	$2,282,736	'yr
NPV	$156,893,779	$30,183,339	
$/kW	$161	$302	

Carbon Injection (CI)			
Capital	$68,250,000	$12,000,000	
OM	$16,692,508	$3,994,788	/yr
NPV	$245,090,669	$54,320,844	
$/kW	$251	$543	

Coal Cleaning (CC)			
OM	$3,673,800	$1,884,000	'yr
NPV	$38,920,290	$19,959,123	
$/kW	$40	$200	

Fabric Filter (FF)			
Capital	$48,750,000	$7,000,000	
OM	$19,474,593	$2,568,078	/yr
NPV	$255,064,114	$34,206,257	
$/kW	$262	$342	

Wet Scrubber (WS)			
Capital	$68,250,000	$15,000,000	
OM	$41,731,270	$7,418,893	/yr
NPV	$510,351,673	$93,595,853	
$/kW	$523	$936	

In table 4, the capital cost to implement these technologies across the US and China is estimated. The results assume that every US power plant employs CC and ESP technologies but nothing else, and that no Chinese power plant uses any cleanup technology. These numbers should be a slight overestimate, but should remain within an acceptable range. The EPA's system will cost $28 - 37 billion. A similar system using WB's estimates will cost $97 – 153 billion. The lowest-cost investment is coal cleaning in China. Implementing ESPs there will cost $41 – 74 billion – more than the EPA's second tier system, but less than the analogous estimate by the WB data.

Table 4. Cost estimates scaled for installing technologies nationally.

(Billions)	CC*	WS	ESP	FF	CI	EPA
China (min)	$3.5	$65.6	$37.5	$46.9	$65.6	---
(max)	$17.7	$140.6	$56.2	$65.6	$112.5	---
NA (min)	----	$56.7	---	$40.5	$56.7	$28.0
(max)	----	$121.5	---	$56.7	$97.2	$36.9

*Assumes the capital cost include the first year of coal cleaning.

4.3 Scenarios

In light of the types and costs of each several scenarios were made in addition to the EPA's scenario. Scenario 1 (S1) is the WB data's estimate for the EPA system. Scenario 2 (S2) is upgrading China's power plants to present US standards (ESPs and CC) and doing nothing in the United States. The cost and net Hg reduction are summarized in Table 5. The calculation assumes that the EPA system and S1 will reduce NA Hg

emissions by 90% and that S2 will reduce China's Hg emissions by 75%. Model Model S1 assumes that only Hg^{+2} and Hg^{p} (approximately half of the total) deposit on NA. Model S2 uses the results from 3.2 to estimate that every ton of Hg^{0} emitted by China (approximately half of the total) falls on NA. The net change in Hg flux to NA is listed as the 'Effect on US' in Table 5. The results show that the cost of investing in China may be more expensive than the EPA's program; however, it may potentially yield a 13-fold reduction in the Hg concentrations over NA. This translates into an extra 4.5 tons Hg removed per billion dollars spent.

Table 5. The costs and effects of several scenarios.

EPA	Cost Billions	Hg before tons/yr	Hg After tons/yr	Net Hg removed tons/yr	Effect on US tons/yr	Effect on US bill $ / ton Hg
NA	$28 - 37	52	1.04	50.96	-25	$1.12 - 1.48
China	0	860	860	0	0	

Scenario 1	Cost Billions	Hg before tons/yr	Hg After tons/yr	Net Hg removed tons/yr	Effect on US tons/yr	Effect on US bill $ / ton Hg
NA	$97 - 153	52	1.04	50.96	-25	$3.88 - 6.12
China	0	860	860	0	0	

Scenario 2	Cost Billions	Hg before tons/yr	Hg After tons/yr	Net Hg removed tons/yr	Effect on US tons/yr	Effect on US bill $ / ton Hg
NA	0	52	52	0	-323	$0.12 - 0.22
China	$41 - 73	860	215	645	-322	

5. Sources of Error

Due to the number of factors, atmospheric modeling is inherently associated with much error. Though the anthropogenic output of Hg can be confidently obtained, deposition rates and transport mechanisms are more complex. For example, deposition of Hg^{0} could be enhanced if it is oxidized in the atmosphere by sunlight and ozone. If this time scale occurred faster than dry deposition, it is possible that the average lifetime of Hg^{0} would be severely shortened. Thus, most of Asia's emissions would fall over the ocean. A second problem might concern the transport mechanism. Because of its inert behavior, Hg^{0} may not be carried by atmospheric jet streams. Hence, more Hg^{0} would fall over the area of generation than over the intercontinental zones. Nonetheless, as China and the remainder of Asia seek to burn more coal, the sources of error will move to the background.

On the efficiency estimation side, the might be significantly higher costs of implementing the same technologies in China as there would be in NA. First, power plants in China may be more decentralized and smaller in scale. This would tend to raise costs. Secondly, it should be noted that gas clean-up technologies mostly remove Hg^{+2} and Hg^{p}. If the clean-up technologies do not improve Hg^{0} adsorption, the effect on NA of S2 will be lessened. It is recommended that further study be directed in this direction.

6. Conclusion

This study shows that investing in first-generation flue-gas cleanup technologies in China would be a more effective at lowering Hg deposition over NA than upgrading all coal plants in NA to the second-generation. Despite the wide room for error in the

calculation, the magnitude of the figures indicates that there may be potentially more significant gains in cleaning up China's emissions before further enhancing the US emissions. Additionally, other pollutants such as ozone forming gases will be substantially reduced. The major resistance to this program could be investors in the US who may be wary of spending money abroad. However, in light of the evidence here-in, it is recommended that investment in China be strongly considered with further study into the intercontinental effect of mercury pollution.

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