The co-benefits of clean air and low-carbon policies on heavy metal emission reductions from coal-fired power plants in China

Guo, Yaqin; Chen, Bin; Li, Yanxian; Zhou, Sili; Zou, Xiaowei; Zhang, Ning; Zhou, Yuran; Chen, Huanxin; Zou, Jun; Zeng, Xianhai; Shan, Yuli; Li, Jiashuo

DOI:
10.1016/j.resconrec.2022.106258

The Co-Benefits of Clean Air and Low-Carbon Policies on Heavy Metal Emission Reductions from Coal-Fired Power Plants in China

Yaqin Guo [1], Bin Chen [2], Yanxian Li [3], Sili Zhou [1], Xiaowei Zou [4], Ning Zhang [4], Yuran Zhou [4], Huanxin Chen [1], Jun Zou [1], Yuli Shan [3 *], Jiashuo Li [4 *]

[1] *School of Energy and Power Engineering, Huazhong University of Science and Technology, Wuhan, 430074, PR China*

[2] *Fudan Tyndall Center, Department of Environmental Science & Engineering, Fudan University, Shanghai, 200438, China*

[3] *Integrated Research on Energy, Environment and Society (IREES), Energy and Sustainability Research Institute Groningen, University of Groningen, Groningen, 9747AG, the Netherlands*

[4] *Institute of Blue and Green Development, Shandong University, Weihai, 264209, PR China*

[*] Corresponding authors: Yuli Shan & Jiashuo Li, Integrated Research on Energy, Environment and Society (IREES), Energy and Sustainability Research Institute Groningen, University of Groningen, Groningen, 9747AG, the Netherlands & Institute of Blue and Green Development, Shandong University, Weihai, 264209, PR China. Email: y.shan@rug.nl & lijiashuo@sdu.edu.cn

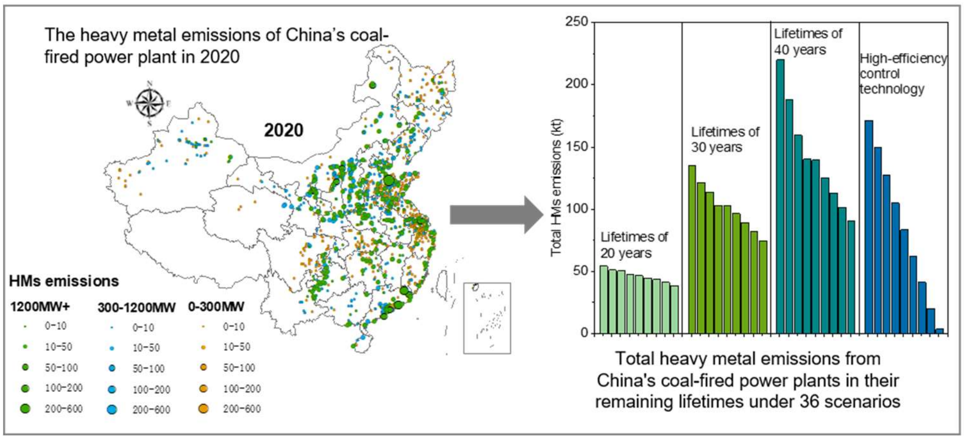
Abstract

China has implemented a series of measures to address air pollutants and carbon emissions from coal-fired power plants, which can mitigate toxic heavy metal emissions simultaneously. By integrating plant-level information and energy activity data, we investigated the co-benefits of clean air and low-carbon policies by compiling a detailed inventory of historical heavy mental emissions (i.e., Hg, Pb, Cd, Cr, Ni, Sb, Mn, Co, Cu, Zn, As, and Se) for China's coal-fired power plants during 2005-2020. Several scenarios were then designed to assess the evolution of heavy metal emissions for each coal-fired power plant with consideration given to the coal washing rate, air pollution control devices, operational hours and lifetime. The total emissions decreased from 12.9 thousand tons in 2005 to 8.8 thousand tons in 2020, which was mainly due to the widely installed upgraded end-of-pipe devices and the decommissioning of small and emission-intensive plants, especially in Sichuan, Jiangsu and Zhejiang. Scenario analysis shows that reducing the operational lifetime to 20 years is the most effective measure to reduce national HM emissions, but the effects differ widely between regions. This study provides insights for the precise co-control of both heavy metals and carbon emissions, which is highly important for meeting the requirements of the Minamata Convention and carbon neutrality.

Keywords

Heavy metal emissions; Coal-fired power plants; Clean air and carbon mitigation policies, Co-benefits; China.

Graphical Abstract



1. Introduction

Owing to their toxicity, long-range transport, and bioaccumulation characteristics, atmospheric heavy metal (HM) emissions can have long-term negative effects on human health and global ecosystems (Bank, 2020; Chen et al., 2018; Dai et al., 2012). Considering the severe threats to human health from HM pollution, the United Nations Environment Programme (UNEP) has listed HMs as a great threat to environmental security (Giang and Selin, 2015; Li et al., 2020). HM pollution has been an urgent concern for China for a long time (Kwon et al., 2018; Tian et al., 2015). The mean atmospheric concentrations of As, Cd, Ni, and Mn are reported to be higher than the WHO standards (Duan and Tan, 2013), and 7,360 deaths were caused by excessive intake of Hg in 2010 in China alone (Chen et al., 2019).

Coal-fired power plants (CFPPs) are one of the main sources of HM emissions (Tian et al., 2014). Approximately 14% of the national emissions of the 12 HMs that we considered in this study (including Hg, Pb, Cd, Cr, Ni, Sb, Mn, Co, Cu, Zn, As, and Se) stemmed from China's CFPPs in 2010 (Tian et al., 2015; Zhu et al., 2016). During 2005-2010, the atmospheric emissions of As, Se, Pb, Cd and Cr stemming from China's CFPPs reached 17.2 kt, approximately three times the total emissions of Europe in 2000 (Pacyna et al., 2007; Tian et al., 2014). In addition, the capacity of CFPPs in China nearly tripled from 2005 to 2020 (National Bureau of Statistics of China, 2006, 2021), resulting in coal consumption increase in the coming decades (Cui et al., 2019).

A series of policies have been issued to comprehensively prevent and control HM pollution in the 12th (2011-2015) and 13th (2016-2020) Five Year Plans (FYP) in China (Environmental Protection Agency of China, 2015a; The State Council of China, 2016). However, few measures have been designed specifically for HM emissions from CFPPs, with the exception of Hg (Environmental Protection Agency of China, 2011). Existing mitigation measures of CFPPs focus on traditional air pollutants (i.e., NOx, SO_2 and PM) and carbon emissions (Tong et al., 2018), such as the “Substitution of Smaller Units with Larger Ones” policy (The State Council of China, 2007), the “Air Pollution Prevention and Control Action Plan” (The State Council of China, 2013), and the “Ultra-Low Emission and Energy Saving of Coal-fired Power Plant Plan” (Environmental Protection Agency of China, 2015b). Thus, current mitigation of HM emissions has mainly resulted from the co-benefits of policies aimed at traditional air pollution and carbon emissions. For example the above policies have driven the successful installation of advanced air pollution control devices (APCDs), which have high removal efficiencies (40%-99%) for HM emissions in China’s CFPPs (Environmental Protection Agency of China, 2015b). The shutdown campaign of small units (usually much more HM-intensive units) also reduced HM emissions (Liu, K.Y. et al., 2018). Efforts have been made to estimate historical inventories of CFPPs’ HM emissions from global, national, regional and individual plant-level perspectives (Liu, K.Y. et al., 2018; Liu, Y. et al., 2018; Pacyna et al., 2007; Pacyna and Pacyna, 2001; Pacyna et al., 2016; Tian et al., 2010; Tian et al., 2015; Zhou et al., 2019). However, these studies fail to identify the changes in plant-level activity data and installed capacity of CFPPs in China in recent years. For instance, a large amount of new coal-fired capacity has been installed in recent years (i.e., approximately 40 GW in 2019), and China has finished ultra-low emission reforms in 2020. Consequently, the accuracy of the historical emission inventory needs to be improved and updated.

Moreover, following the ratification of the Minamata Convention on mercury in 2017, controlling HM emissions in line with the Convention’s stipulations has been an environmental challenge for China (Bank, 2020; Ministry of Environmental Protection, 2017). However, the current environmental policies still focus on the mitigation of carbon emissions and the control of air pollution caused by PM, NOx and ozone (The Central Committee of the Communist Party of China, 2021). For example, China has committed to peaking carbon emissions in 2030 and achieving “carbon neutrality” in 2060 (Liu et al., 2015). To fulfill these commitments, more stringent policies to control CFPPs, the major emitters of carbon emissions and traditional air pollutants in China, will be implemented (Cui et al., 2019; Jewell et al., 2019; Xing et al., 2020). This is bound to synergistically affect the future HM emission trajectory. Understanding the influence of future environmental policies accurately can enlighten policymakers to implement appropriate mitigation policies (Chao et al., 2021; Ding et al., 2021; Tong et al., 2021). However, the future emission reduction pathways of HMs are mainly designed on the basis of the outdated national and provincial emission inventories of a few HMs (Kwon et al., 2018; Navratil et al., 2016; Sunderland and Selin, 2013; Sung et al., 2018). They ignore the integration of the latest policies and the differences in the coal washing rate, APCDs and combustion technologies among current CFPPs.

To fill the above gaps, this study assessed the co-benefits of China's clean air and low-carbon policies on HM emission reduction by integrating plant-level information. We first compiled a time-series emission inventory of 12 HMs (including Hg, Pb, Cd, Cr, Ni, Sb, Mn, Co, Cu, Zn, As and Se) of CFPPs in China during 2005-2020 that contains the latest plant-specific data and APCD optimization. Then, we projected and evaluated different reduction pathways of future HM emissions for each power plant with consideration of APCD upgrades, coal washing rate, operational hours and lifetime.

2. Methodology and Data Sources

2.1 Plant-Level HM Inventories during 2005-2020

The atmospheric emissions of the 12 HMs from China's CFPPs were calculated via a refined activity database and emission factor of each plant (Liu, K.Y. et al., 2018; Zhou et al., 2019). The activity database housed by the China Electricity Council and updated based on the environmental assessment reports of these CFPPs or the report released by the Ministry of Ecology and Environmental of China (Ministry of Ecology and Environmental of the People's Republic of China, 2014a, b). The plant-specific emission factors and the calculation equation of the emissions can be expressed as follows:

$$\mathrm{EF_{ij}} = \mathrm{A_i} * (1 - \mathrm{Q_i} * \omega_\mathrm{k}) * \mathrm{R} * \prod (1 - \eta_{\mathrm{ij}}^{\mathrm{m}}) \quad (1)$$

$$\mathrm{E_{ij}} = \mathrm{C_{ij}} * \mathrm{EF_{ij}} \quad (2)$$

where EF_{ij} is the emission factor of CFPP j in province i; E_{ij} is the HM emissions of CFPP j in province i; C_{ij} is the amount of coal consumption of CFPP j in a specific year (for CFPPs without actual data, the calculation process is shown in the Supplementary methods); A_i is the average content of HM in consumed coal in province i; Q_i is the coal washing rate of province i; ω_k is the removal efficiency of coal washing technology for heavy metal k (Table S1); R represents the release ratio of the coal-fired boiler; η represents the removal efficiencies of end-of-pipe devices for each HM in CFPP j in province i; and m is the type of end-of-pipe control device for NO_x, SO_2 and PM (Table S2).

2.2 Scenario Analysis

A scenario analysis of the plant-based emission pathway was developed in this study to estimate the co-benefits to HM reduction under different clean air and low-carbon policies. We tracked the future HM emission reduction potential from changes from clean air (APCD upgrades and coal washing rate) and carbon reduction policies (annual operational hours and lifetime of CFPPs). Based on the combinations of the four factors (we designed three levels of each dimension, namely, business-as-usual, weak, and strong), we designed 36 reduction scenarios for future HM emissions. The detailed reasons and assumptions of the four factors in the scenario analysis can be seen in section 2 in the supplementary methods and Table S3, and a detailed description of these scenarios is shown in Table S4.

2.3 Data Sources

Based on the reports issued by the China Electricity Council and relevant government agencies of each region, the specific variables of each CFPP are collected (China Electricity Council, 2006, 2011, 2015, 2019). These variables include location, capacity, annual power generation, vintage year, ACPDs, operational hours and coal consumption per unit power supply. This study covers more than 90% of the national coal-fired power capacity each year (National Bureau of Statistics, 2011). CFPPs built in 2019 and 2020 were extracted from the latest public and open sources (Beijixing, 2020; Global Energy Monitor, 2021).

HM Content in Consumed Coal. The average contents of 12 HMs in raw coal from 30 provinces have been elaborately reviewed and assessed by Tian et al. (Tian et al., 2015), as shown in Table S5. Due to the spatial mismatch of coal production and consumption centers in China, the HM contents in consumed coal are different from those in produced coal in the same province. By using a coal transmission matrix of 2005, 2010 and 2014, we summarized the HM contents in the consumed coal of each province (Liu, K.Y. et al., 2018; Wu et al., 2020). Based on the official statistical data obtained from the China Energy Statistical Yearbook (2019) and China Coal Industry Yearbook (2018), a coal transmission matrix of 2018 was established among 30 provinces and other foreign countries (Table S6). Combined with the provincial and international average content of HM in produced coal, the provincial weighted-average contents of 12 HM in consumed coal in 2018 were determined, as illustrated in Table S7. The HM contents of consumed coal in 2020 were assumed to be the same as those in 2018, owing to data limitations.

Removal Efficiencies of APCDs to HM Emissions. The coal washing rates used in this study were derived from the China Energy Statistical Yearbook (2006, 2011, 2015, 2019), and the removal efficiencies for HM of coal washing are summarized in Tables S1-S2 (Tian et al., 2015). The removal efficiencies of different APCDs to HM, ranging from 32.4% to 99.9%, were derived from Tian et al. (Tian et al., 2015), as summarized in Table S7. Notably, there are few end-of-pipe devices dedicated to removing HM from flue gas in CFPPs in China. To further quantitatively assess HM emissions, a unit-based inventory of APCDs was established in accordance with the Ministry of Ecology and Environmental of China and the Environmental Assessment Report for certain CFPPs.

2.4 Uncertainty and Sensitivity Test

Monte Carlo simulation is applied to quantitatively estimate the uncertainties in the emission inventories of each HM. The overall uncertainty is calculated under the 95% confidence interval around the arithmetic mean. Based on previous studies, we assume that the coal consumption and average concentration of 12 HMs comply with a normal distribution with a coefficient of variation (the standard deviation divided by the mean) of 5%. The distribution characteristics of the APCD removal efficiencies are shown in Table S8. Moreover, we performed a sensitivity test for coal washing and the removal efficiency of end-of-pipe devices, and the detailed settings are shown in SI.

3. Results

3.1 Historical HM Emission Inventories from 2005 to 2020

3.1.1 Overall HM Emissions and Emission Intensity from 2005-2020.

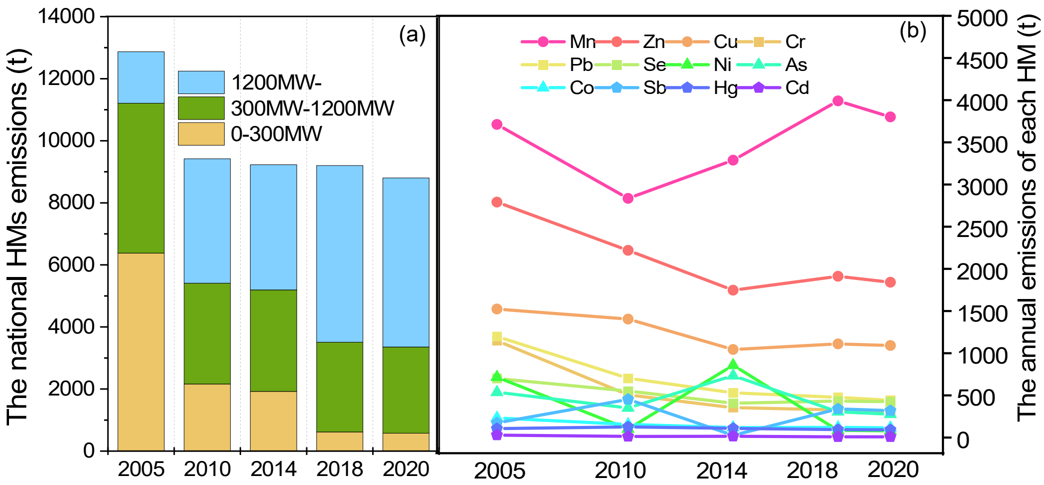


Figure 1. The total HM emissions from 2005 to 2020 classified by (a) the capacity sizes of CFPPs and (b) the types of HM.

Atmospheric HM emissions experienced a rapid reduction from 12,869.8 tons in 2005 to 8,801.0 tons in 2020 (Figure 1). However, nearly 90% of this reduction occurred during 2005-2010, after which the speed of emission reduction slowed down quite markedly, as further updates for APCDs, such as the wide application of denitration devices, have limited contributions to HM emission reductions with the exception of Hg. Correspondingly, the national emission intensity, defined as the amount of HM emissions per GW*h of electricity generation, showed a remarkable reduction from 6.4 t/GW*h in 2005 to 1.9 t/GW*h in 2020 (Figure 2). In addition, the uncertainty ranges of the 12 types of HMs in different years are presented in Figure S1, and the detailed settings results of the sensitivity test are shown in Figures S2-S4.

From the perspective of installed capacity, the total emissions from large-scale CFPPs (with a capacity larger than or equal to 1200 MW) nearly tripled (from 1,659.5 tons to 5,446.3 tons) and became the largest emitters in 2020, due to the increasing amount of installed capacity during 2005-2020. However, the total HM emissions of small CFPPs (with a capacity less than 300 MW) showed a sharp decrease from 6,377.9 tons to 577.5 tons, totally offsetting the emission increase from large-scale CFPPs. As most small CFPPs are industrial self-use plants equipped with less efficient boilers and either unadvanced APCDs or even no APCDs before 2010, the shutdown campaign of small units led to substantial emission reductions. In addition, the emissions from medium-scale CFPPs (with a capacity ranging from 300 MW to 1,200 MW) decreased by more than 40% from 4,832.4 tons to 2,777.2 tons, accounting for 50.3% of the total reductions. Among HM emission types, Mn, Zn and Cu were ranked as the top three, together accounting for nearly 70% of the total HM emissions. Almost all HM types showed downward trends from 2005 to 2020; the exceptions were Mn and Sb. Sb underwent a significant increase of 84.0%, the result of high-consumption provinces switching their coal sources from regions with a low content of Sb in raw coal (i.e., Shanxi and Inner Mongolia) to regions with a high content (i.e., Shaanxi and Guizhou).

3.1.2 Provincial HM Emissions and Emission Intensity

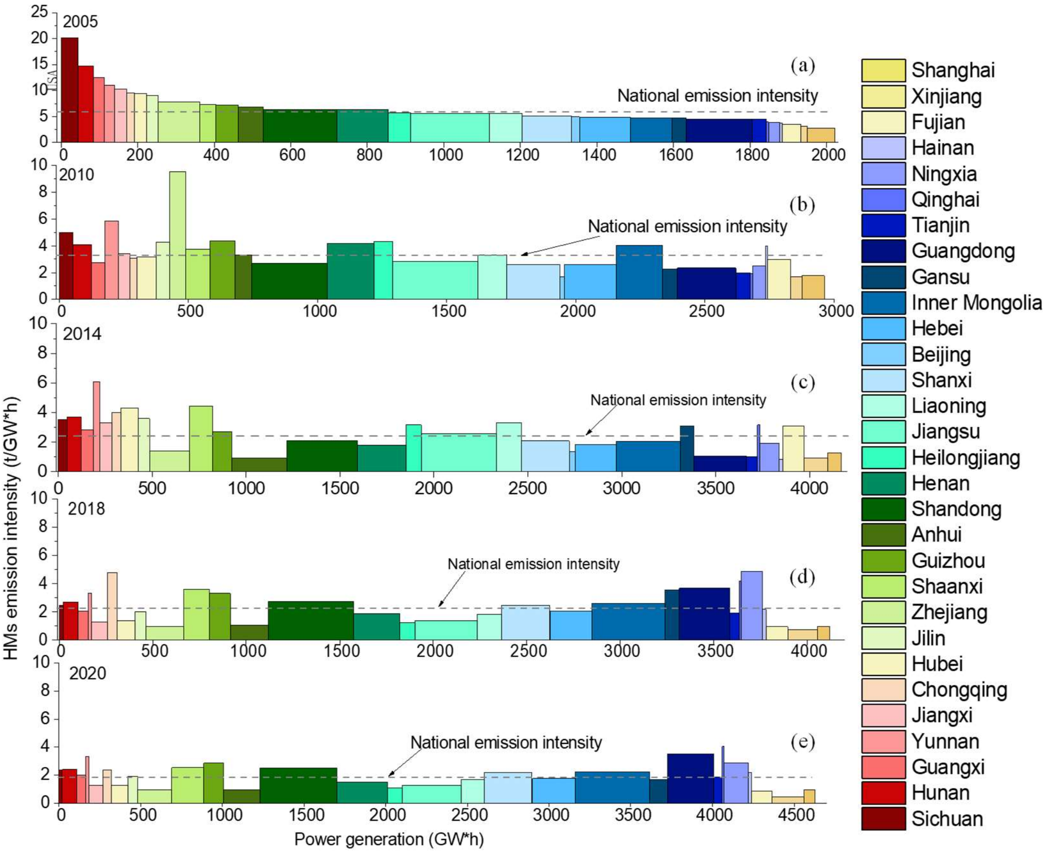


Figure 2. Provincial HM emissions and emission intensity from CFPPs in China in (a) 2005, (b) 2010, (c) 2014, (d) 2018 and (f) 2020. The provincial emission intensities are shown (y-axis). The widths of the bars represent the amount of power generation (x-axis), and the area indicates the HM

emissions of the specific province. The gray line represents the national emission intensity.

Enormous differences existed between provinces, as revealed in Figure 2. Specifically, the top five provincial emitters together contributed to approximately 40% of the total emissions in these years, while the bottom five were responsible for less than 2%. In addition, the larger provincial emitters have shifted from China's traditional industry-based, economically intensive regions, or populous economic centers, such as Jiangsu, Guangdong and Sichuan, respectively, to regions with abundant coal reserves, such as Inner Mongolia, Shanxi and Shaanxi, as several large energy-based construction projects have been constructed in these provinces with the implementation of the "Western Development" and "West-East Electricity Transmission Project" strategies. However, HM emissions remain at a low level in Northwest China due to its relatively small installed capacity. Notably, 22 provinces experienced emission reductions from 2005 to 2020, among which Sichuan, Jiangsu and Zhejiang ranked as the top three. In contrast, the HM emissions of Ningxia, Qinghai and Inner Mongolia nearly doubled, mainly due to the large-scale construction of coal-fired power projects and changes in consumed coal sources in these regions.

The emission intensities of nearly all provinces were reduced from 2005 to 2020, apart from those in Qinghai. Specifically, Sichuan, Hunan, and Guangxi were responsible for the largest decline in emission intensity during 2005-2020, with reductions of -17.8 t/GW*h, -12.3 t/GW*h, and -10.5 t/GW*h, respectively. Regions with high emission intensities usually consume coal from Gansu or Shaanxi, where the production of coal is HM-intensive. In addition, the major contributor to the decline in emission intensity changed from the reduction in HM emissions in 2005 to the increase in power generation in 2020.

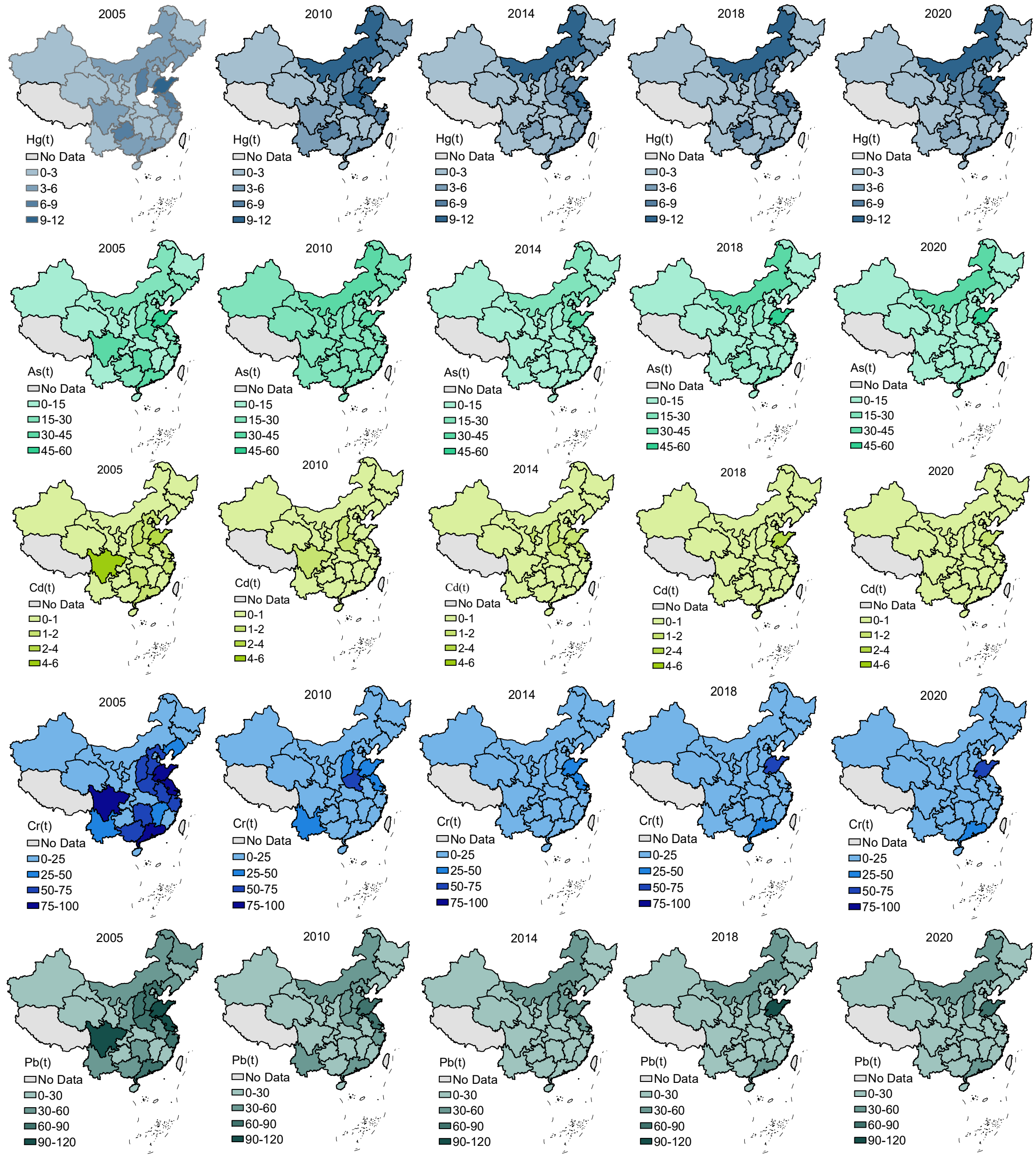


Figure 3. Provincial emission evolutions of Hg, Cd, As, Cr and Pb in 2005, 2010, 2014, 2018 and 2020 (more detailed information about the emissions of other HMs can be found in the supplementary results and Figures S5-S11 in SI).

Figure 3 portrays the provincial emission evolution of five HM emissions (Hg, Cd, As, Cr and Pb) that have attracted the most attention in China and have the greatest impact on China's human health. The total emissions of Hg, Cd, As, Cr and Pb declined by 11.7%, 71.2%, 47.4%, 73.2% and 62.1%, respectively. The provincial emissions of the five HMs varied greatly in space, and the difference gradually expanded. Specifically, Shandong was the largest provincial emitter of As, Cr and Pb and the second largest emitter of Hg and Cd in 2005, accounting for 10%, 9%, 10%, 8.5% and 9.3%. of the national, respectively. In 2020, the above HM emissions of Shandong occupied a larger proportion, more than 20%, except for Hg (12.4%). Due to the high content of Cd in consumed coal (1.72 g/t), the emissions of Cd were as high as 5.8 tons in 2005, and it experienced a sharp decline in 2010 (1.3 tons). In addition, the emissions of Cr and Pb in all provinces declined, especially in Beijing, Sichuan and Yunnan. Notably, although the national As emissions have decreased greatly, the As emissions in Shandong and Inner Mongolia increased by 17.5% and 32.2%, respectively. In addition, the Hg emissions of Inner Mongolia doubled from 2005 (5.4 tons) to 2020 (11.3 tons), making it the second largest emitter in 2020, second only to Shandong (11.3 tons). Conversely, due to the decline in coal consumption, Henan, the largest emitter in 2005, reduced Hg emissions by nearly 60%.

3.1.3 Plant-Level HM Emissions

China's CFPPs are unevenly distributed and mainly located in populated areas, as shown in Figure 4. In addition, from 2005 to 2020, the major plant-level emitters changed from small-scale CFPPs to large-scale CFPPs. There were 6 CFPPs with emissions of more than 100 tons in 2005, together accounting for 7% of the total, although they constituted only 0.4% of the national installed capacity. Specifically, the Laiwu power plant (located in Laiwu, Shandong) was the largest individual emitter (205.13 tons) in 2005 due to its large capacity (405 MW) and lack of end-of-pipe devices. From 2005 to 2014, mainly due to the wide application of dust removal equipment capable of eliminating 33%-99% of the HM emissions in China's CFPPs, emissions decreased suddenly for most CFPPs. Unlike CFPPs with high emissions in 2005, CFPPs with high emissions in 2020 were generally equipped with desulfurization and dust removal devices, however, they also possessed large capacities and were responsible for a large amount of coal consumption. For instance, the CFPPs of the Binzhouweiqiao Cotton Textile Group (located in Shandong) ranked first with 445.7 tons of HM emissions due to their enormous capacity (22,050 MW) and inefficient APCDs (SNCR+ desulfurization tower + ESP) with a low removal efficiency of HM emissions.

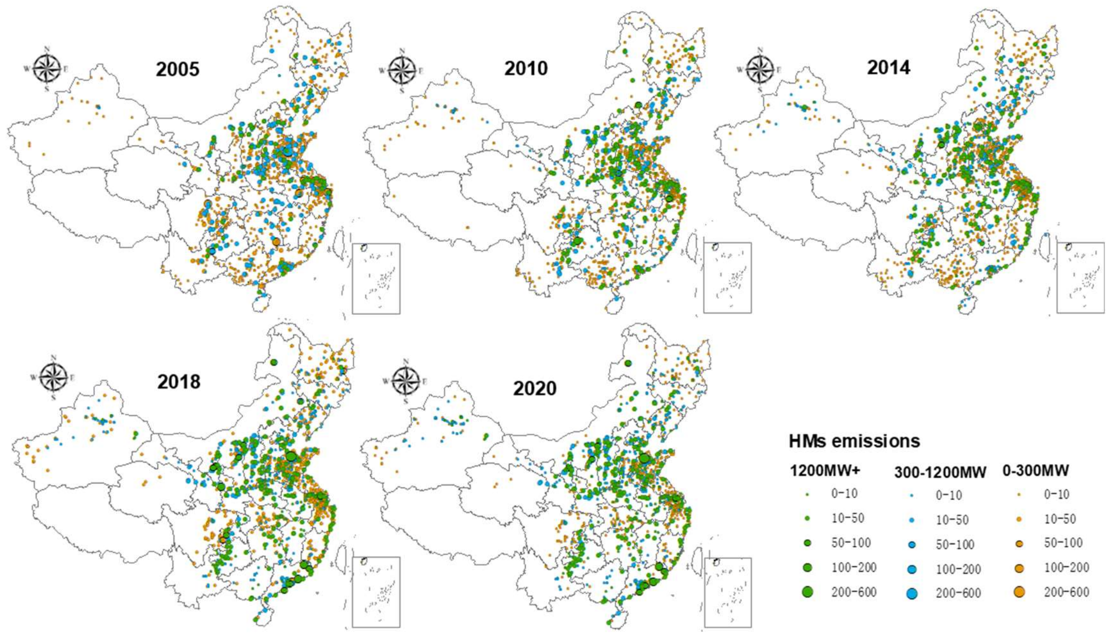


Figure 4. Plant-level HM emissions of large-scale CFPPs (in green), medium-scale CFPPs (in blue), and small-scale CFPPs (in yellow) during 2005-2020.

3.2 Co-Benefits of Clean Air and Low-Carbon Policies to HM Reduction

3.2.1 National Evolution of HM Emissions

The total HM emissions vary among the 36 scenarios, as presented in Figure 5e. Reducing the lifetime of all CFPPs to 20 years will substantially reduce the future potential HM emissions in comparison with scenarios with longer CFPP lifetimes of 30 or 40 years. Specifically, the scenarios with the shortest lifetime (20 years) account for 7 out of 10 scenarios with the lowest total emissions (light green columns in Figure 5e). In contrast, nine of the top ten scenarios with the largest total HM emissions represent scenarios with CFPP lifetimes of 40 years, and their total HM emissions range from 125.1 kt to 220.1 kt (dark green columns in Figure 4e). The scenario with the strictest policy (“5%-1.5-20”) contributes only 38.72 kt of HM emissions, which is larger than that under “a2021” (3.5 kt) and “a2022” (20.3 kt). However, the above scenarios represent extreme situations with the most flexible or strictest policy combinations in the future, which are infeasible in practice. Additionally, if the HECTs are widely applied in all CFPPs before 2030, the total HM emissions are equal to those under the “2%-1.5-20” scenario. If the national application of HECTs is completed after 2052, then the total HM emissions exceeds those of the “0-0-30” scenario. Thus, widely applying HECTs in CFPPs with remaining lifetimes over 10 years can relieve the pressure of HM emission reductions and ensure the long-term operation of CFPPs in China in the future.

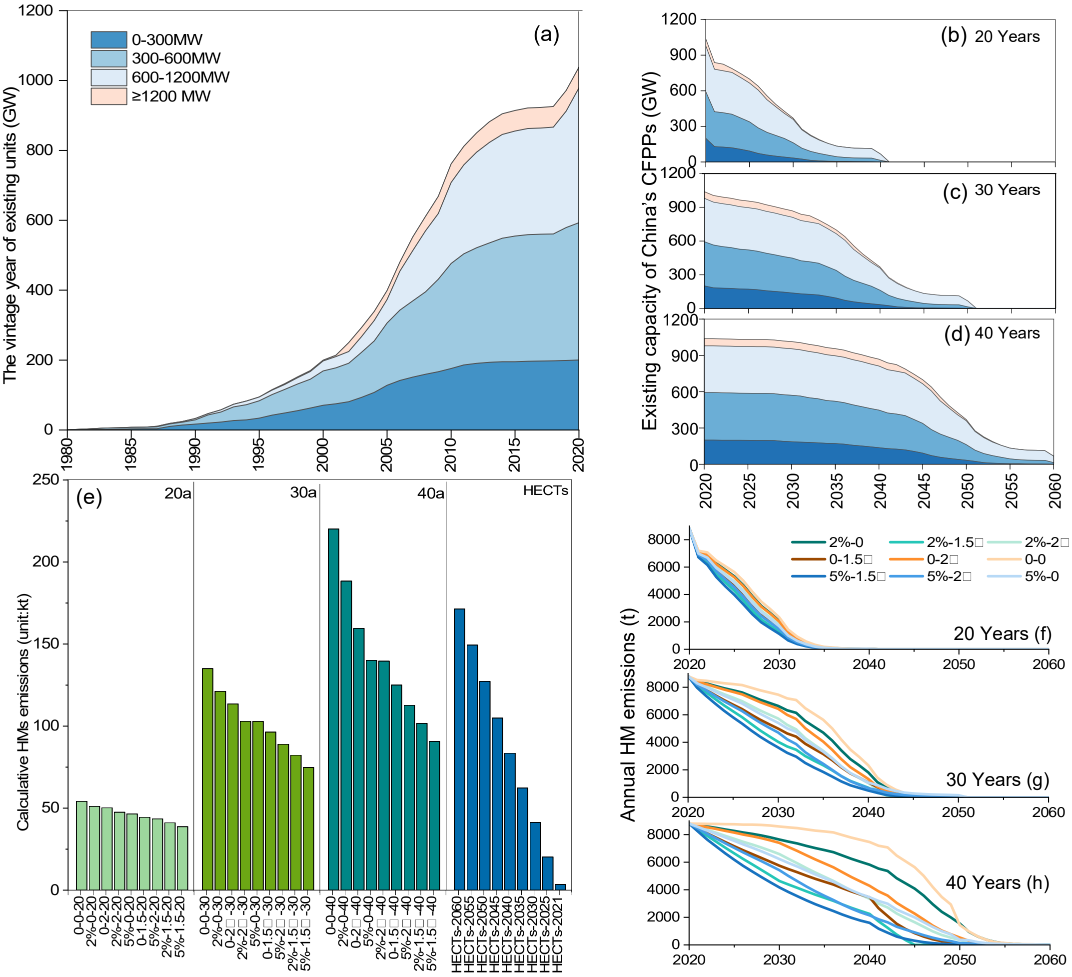


Figure 5. (a) The current state and that of previous years of China's existing units until 2020 and the annual installed capacity in the future with lifetimes of (b) 20 years, (c) 30 years and (d) 40 years; (e) the total HM emissions from CFPPs under 36 scenarios (the first number is the annual increasing rate of the coal washing rate, the second one is the change in the annual operational hours, and the third number is the assumed lifetime). These results are ordered by their lifetimes; for example, "0-0-20" means no change in the coal washing rate and annual operational hours, and the lifetimes of all the CFPPs are set as 20 years. The dark blue columns represent the total HM emissions under the HECTs scenarios); the trajectories of HM emissions under different scenarios with lifetimes of (f) 20 years, (g) 30 years and (h) 40 years and the trajectories of HM emissions under the HECTs scenarios are shown in Figure S12 in SI.

The annual emissions under the major scenarios are illustrated in Figure 5 (f)-(h). The HM emissions under the scenario "5%-1.5-20" experience a sharp reduction since 2021, as more than

17 GW of CFPPs were built before 2000 and then would be retired in 2021, leading to a 2,072.5 t HM reduction. In addition, when the base lifetimes are 40 years, the annual emissions for three of the scenarios, namely, “5%-0-40”, “0-1.5-40” and “0-2-40”, exceed those under the “0-0-30” scenario in the 2030-2040 period, although their total emissions are similar. This indicates that increasing the coal washing rate could be a substitute for reducing the operational hours or lifetime of CFPPs.

3.2.2 The evolution trajectories of provincial and plant-level HM emissions

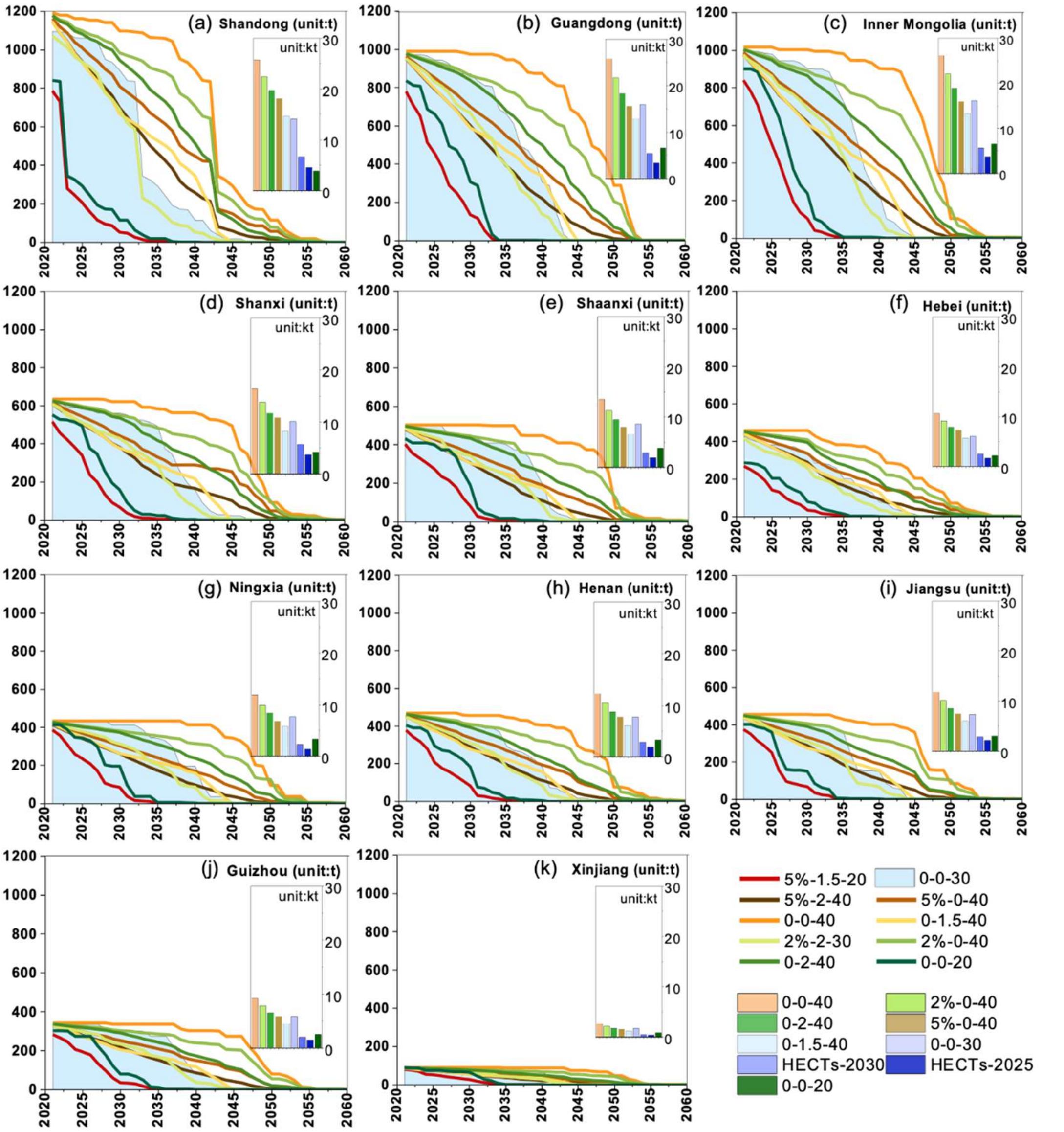


Figure 6. Future HM emissions from CFPPs under the major scenarios in the top 10 provincial regions and the province with the smallest emissions (the lines and areas are consistent with those

in Figure 5f, and the small histograms represent the total HM emissions of the major 7 scenarios).

Coal-fired units are extremely young in many provinces, including Inner Mongolia, Jiangsu, and Henan; thus, they have long-term locked in effects on HM emissions throughout their lifetimes (Figure 6). In Shaanxi, for example, annual HM emissions start to show an obvious reduction in approximately 2050 under the “no policy” scenario, as most units in Shaanxi are built after 2010 (Figure 6e). In other provinces with high HM emissions, such as Inner Mongolia, Shanxi, and Jiangsu, sharp declines occur in approximately 2045 under the “no policy” scenario. In contrast to the provinces mentioned above, the annual emissions of Guangdong, Henan, and Hebei decline at a steady rate due to the gradual development of power plant construction.

In addition, the provincial emission reduction pathways are quite unequal. Provincial regions with small amounts of HM emissions show limited emission reduction potentials under various scenarios. For instance, the total HM emissions in Xinjiang under the 10 scenarios range from 2,664.2 tons under the “no policy” scenario to 568.7 tons under the “5%-1.5-20” scenario with the strictest policy. However, reduction potential will be different for provinces with high HM emissions, especially for the top ten provincial emitters. For example, if “the strictest policy” is applied in Shandong, the total HM emissions will be just 2,862.6 tons, 53.1% of which would occur during the first two years (2021 and 2022), while the total HM emissions could reach 25,689.1 tons under the “no policy” scenario. This unusual decline in Shandong results from the retirement of CFPPs of the Binzhouweiqiao Cotton Textile Group, as its APCD combination, SCR+ seawater desulfurization control system (SWD)+ESP, has a much lower synergistic removal rate of HM emissions (ranging from 41% to 97%) compared with that of HECTs (ranging from 90.5% to 99.9%).

Notably, none of the 30 provinces except Shandong can meet the demand of the HM emission reduction targets under the “HECTs-2025” scenario with only one measure. Specifically, the HM emissions in Shandong are only 3,823.9 tons under the “0-0-20” scenario, and account for 83.5% of those under the “HECTs-2025” scenario. Additionally, the HM emissions of Shanxi and Hebei under the “0-0-20” scenario are less than those under the “HECTs-2030” scenario. As the remaining lifetimes of units in these provinces are limited under the “0-0-20” scenario, a lifetime of 20 years will lead to rapid decommissioning of large numbers of units in a short time.

4. Discussion and Policy Implications

This study verified that China's clean air and low-carbon policies can lead to considerable co-benefits to the HM emission reductions in CFPPs, as more than 30% of the emissions were reduced during 2005-2020, despite the installed coal-fired capacity nearly tripling (from 370 GW to 1080 GW) (Ministry of Ecological Environment of China, 2018). Considering that nearly 90% of the reduction in emission intensity occurred during 2005-2014, the co-benefit to HM reduction of these existing measures weakened.

Currently, some CFPPs still lack efficient HM control APCDs. For example, the CFPP of the Binzhouweiqiao Cotton Textile Group in Shandong used an SNCR+ desulfurization tower + ESP with a low removal efficiency of HMs. Thus, a quick upgrading of APCDs in CFPPs that are not yet equipped with HECTs can indeed contribute to considerable HM emission reductions in the short term. The results of the scenario analysis indicated that more than 98% of HM emissions would have been removed if HECTs were applied in all CFPPs by 2021. However, as China completed ultra-low emission reforms in the coal power sector by 2020, CFPPs with such low removal efficiency now account for less than 10% of the national installed capacity. These old, small CFPPs generally have been in service for almost 20 years and will be eliminated early according to the requirements of ultralow emission reforms (Environmental Protection Agency of China, 2015b). Moreover, upgrading APCDs will place a heavy financial burden on CFPPs that have already been suffering a deficit (Chi et al., 2021). Therefore, there is limited possibility for upgrading APCDs of CFPPs nationwide in a short time. However, coal power will survive and account for 5% of all power by 2060 (Duan et al.; Global energy Internet Development Cooperation Organization, 2021). Hence, further upgrading of APCDs, including the construction of specialized control devices for HM emission reduction, is necessary in the future, especially for newly built CFPPs with long remaining lifetimes in Shaanxi, Inner Mongolia and Henan. In addition, future HM emission reduction in CFPPs can also rely on the co-benefits from China's clean air and low-carbon policies (Global Energy Monitor, 2019).

Reducing the lifetime is more effective than reducing the operational hours and increasing the coal washing rate, as nearly 70% of the existing CFPPs would be retired by 2025 under scenarios with lifetimes of 20 years. Additionally, almost 59 GW of small units with high emission intensity still exist, most of which are self-use CFPPs built before 2005, such as CFPP of Binzhouweiqiao Cotton Textile Group in Shandong and the self-use CFPP of Wu'an Mingfang Iron and Steel Co., Ltd. in Hebei. Quickly decommissioning these CFPPs can contribute nearly 20 Kt in emission reductions compared with extending their lifetimes to 40 years. Thus, shutting down inefficient small units can be set as the priority in emission reduction strategies. However, it is impossible to accelerate early retirements nationwide or reduce the lifetime of all units to 20 years in China, considering rich domestic coal resources, growing energy demand, and compound financial losses (China Electricity Council, 2021; Cui et al., 2019; Peng et al., 2018; State Statistical Bureau, 2021).

Reducing operational hours and lifetimes are both effective ways to mitigate HM emissions. However, how to choose these two measures needs to be carefully considered in different regions with striking heterogeneity in terms of renewable energy availability, potential asset shelving and social impacts (e.g., employment and residential heating). Western and northwestern China, which have abundant wind and solar resources, will experience rapid renewable power expansion, especially in Gansu, Inner Mongolia and Xinjiang (Chi et al., 2021), as China has committed to installing more than 1200 GW of wind and solar power by 2030 (National Energy Administration, 2020). However, the supply of renewable energy is intermittent and fluctuates over time. Consequently, quickly halting the construction of new CFPPs nationwide can not only reduce the risk of wind and solar curtailments, but also promote the process of eliminating coal in China. CFPPs that have high generation efficiency and advanced APCDs can serve as supplementary power sources, such as residential heating in winter or electricity peak shaving. However, regions with limited renewable energy but developed manufacturing industrial chains, such as Jiangsu, Zhejiang and Guangdong, have a high demand for stable and reliable electricity and heating. Power transmission instead of coal transport from western to eastern China can not only meet the power demand in these densely populated regions but also reduce potential HM pollution. Moreover, progressively shutting down pollutant-intensive CFPPs with a reasonable timetable in those regions is necessary. For instance, for CFPPs with the similar pollution levels, priority should be given to the elimination of HM-intensive CFPPs in the eastern regions of China. In addition, the governments of these regions can replace coal with clean energy in advanced CFPPs, such as natural gas, methane and biomass, which can reduce HM emissions without loss of power generation. Tiered pricing for electricity can be applied in these regions to contain high-energy industries and promote the readjustment of industrial structure. Then, they can gradually eliminate small- and then medium-scale CFPPs with low co-benefits to HM emission reductions.

The potential emission reduction effect of the “5%-0-0” scenario is similar to that of the “0-0-30” scenario, indicating that there is great potential to further mitigate HM emissions in CFPPs by increasing the coal washing rate. Nevertheless, great gaps exist between China and developed countries in the coal washing rate of CFPPs (National Bureau of Statistics of China, 2021; Pudasainee et al., 2017). Thus, for regions with high contents of HM in coal, such as Shaanxi, Hunan, and Guizhou, the local government can encourage CFPPs to use washed coal with subsidies. Based on knowledge from the successful implementation of PM and SO_2 controls, the implementation of more rigorous emission standards can encourage CFPPs to reduce HM emissions (Tian et al., 2015). Therefore, China can accelerate the formulation of standards for the content of HMs in the consumed coal of CFPPs, which can set an access threshold for coal before combustion. Finally, strengthening the amendment of national ambient air quality standards and further incorporating HM standards into ultra-low emission retrofits in other HM-intensive sectors are conducive to the implementation of HM emission control policies (Chen et al., 2020).

Our study is subject to some limitations. First, this study failed to include all CFPPs (less than 10% of the national capacity), resulting in the underestimation of HM emissions. However, it is thought to be reasonable and acceptable due to data availability. Second, we did not completely eliminate the emissions of units that were shut down in 2019 and 2020, which may overestimate the emissions in 2020. Nevertheless, to obtain a more reliable HM emission inventory of CFPPs in China, many more detailed and comprehensive investigations are still needed.

5. Conclusions

This study investigated the co-benefits of China’s clean air and low-carbon policies by depicting the evolution of 12 types of HMs through the development of a historical inventory with plant-level information of China’s CFPPs during 2005-2020 and then projecting the future reduction pathways, taking into consideration the coal washing rate, lifetimes, operational hours and air pollution control devices (APCDs). The results show that the total HM emissions have been reduced from 12869.82 t in 2005 to 8801.00 t in 2020. However, nearly 90% of the reduction occurred during 2005-2014, indicating that the co-benefit has gradually weakened. The top three ranked HMs, Mn, Zn and Cu, together account for nearly 70% of the total emissions. However, there are very large gaps between provinces and even CFPPs in emissions, and larger emitters have transferred from industry-based or economically intensive regions to regions with abundant coal reserves. Among the 36 scenarios, reducing the operational lifetime to 20 years is the most effective measure for HM emission reduction, especially for the small, old CFPPs in Shandong, Inner Mongolia and Hebei. Additionally, increasing the coal washing rate can be a substitute for reducing the lifetimes or operational hours of CFPPs. Therefore, the old, small units with ineffective technology and unadvanced APCDs should be quickly shut down, especially those in Shandong and southwestern regions. Increasing the coal washing rate gradually can reduce HMs without affecting coal power generation.

Acknowledgments

This work was supported by the National Natural Science Foundation of China (72074137), the National Office for Philosophy and Social Sciences (21ZDA065), the Natural Science Foundation of Shandong Province (ZR2021YQ27), the Shandong Provincial Science Fund for Excellent Youth Scholars (ZR2021YQ27), and the Taishan Scholars Project of Shandong Province.

References

Bank, M.S., 2020. The mercury science-policy interface: History, evolution and progress of the Minamata Convention. Science of the Total Environment 722.

Beijixing, 2020. Beijixing Power News. https://news.bjx.com.cn/, Beijing.

Chao, Z., Gang, H.C., Jjd, E., Lz, F., 2021. Long-term transition of China's power sector under carbon neutrality target and water withdrawal constraint.

Chen, C.Y., Driscoll, C.T., Eagles-Smith, C.A., Eckley, C.S., Gay, D.A., Hsu-Kim, H., Keane, S.E., Kirk, J.L., Mason, R.P., Obrist, D., Selin, H., Selin, N.E., Thompson, M.R., 2018. A Critical Time for Mercury Science to Inform Global Policy. Environ. Sci. Technol. 52(17), 9556-9561.

Chen, L., Liang, S., Liu, M., Yi, Y., Mi, Z., Zhang, Y., Li, Y., Qi, J., Meng, J., Tang, X., 2019. Trans-provincial health impacts of atmospheric mercury emissions in China. Nat. Commun. 10(1).

Chen, L., Liang, S., Zhang, H.R., Cai, X.R., Chen, Y.C., Liu, M.D., Lin, H.M., Li, Y.M., Qi, J.C., Tong, Y.D., Zhang, W., Wang, X.J., Shu, J., 2020. Rapid Increase in Cement-Related Mercury Emissions and Deposition in China during 2005-2015. Environ. Sci. Technol. 54(22), 14204-14214.

Chi, J.R., Wang, B., Zhang, H., Kang, J.J., Lu, T.F., Huang, Y.P., Yang, W., Zhang, H.Y., Sun, L.S., 2021. Regional coal power overcapacity assessment in China from 2020 to 2025. Journal of Cleaner Production 303.

China Electricity Council, 2006. the Inventory of Power Plants in China 2005 (Undocumanted Report). Beijing, China.

China Electricity Council, 2011. the Inventory of Power Plants in China 200 (Undocumanted Report). Beijing,China.

China Electricity Council, 2015. the Inventory of Power Plants in China 2014 (Undocumanted Report). Beijing,China.

China Electricity Council, 2019 the Inventory of Power Plants in China 2018. Beijing, China.

China Electricity Council, 2021. Analysis and forecast report on national power supply and demand situation. Beijing.

Cui, R.Y.Y., Hultman, N., Edwards, M.R., He, L.L., Sen, A., Surana, K., McJeon, H., Iyer, G., Patel, P., Yu, S., Nace, T., Shearer, C., 2019. Quantifying operational lifetimes for coal power plants under the Paris goals. Nat. Commun. 10, 9.

Dai, S., Ren, D., Chou, C.-L., Finkelman, R.B., Seredin, V.V., Zhou, Y., 2012. Geochemistry of trace elements in Chinese coals: A review of abundances, genetic types, impacts on human health, and industrial utilization. International Journal of Coal Geology 94, 3-21.

Ding, D., Xing, J., Wang, S., Dong, Z., Zhang, F., Liu, S., Hao, J., 2021. Optimization of a NOx and VOC Cooperative Control Strategy Based on Clean Air Benefits. Environ. Sci. Technol.

Duan, H., Zhou, S., Jiang, K., Bertram, C., Harmsen, M., Kriegler, E., Vuuren, D.P.V., Wang, S., Fujimori, S., Tavoni, M., Assessing China's efforts to pursue the 1.5°C warming limit. Science 372.

Duan, J., Tan, J., 2013. Atmospheric heavy metals and Arsenic in China: Situation, sources and control policies. Atmospheric Environment 74, 93-101.

Environmental Protection Agency of China, 2011. Emission standard of air pollutants for thermal power plants (GB 13223-2011). Beijing,China.

Environmental Protection Agency of China, 2015a. the 12th Five Year Plan for Comprehensive Prevention and Control of Heavy Metal Pollution. Beijing,China.

Environmental Protection Agency of China, 2015b. Full Implementation of Ultra-Low Emission and Energy Saving Transformation of Coal-Fired Power Plants. Environmental Protection Agency of China, Beijing, China.

Giang, A., Selin, N.E., 2015. Benefits of mercury controls for the United States. Proceedings of the National Academy of Sciences 113(2), 201514395.

Global energy Internet Development Cooperation Organization, 2021. Research on China's energy and power development planning in 2030 and prospect in 2060.

Global Energy Monitor, 2019. Prosperity and decline 2019: tracking global coal-fired power plant development, in: Shearer, C. (Ed.). Global energy monitoring, Greenpeace India and Sierra Club.

Global Energy Monitor, 2021. Boom and Bust 2021-tracking the global coal plant pipeline.

Jewell, J., Vinichenko, V., Nacke, L., Cherp, A., 2019. Prospects for powering past coal. Nat. Clim. Chang. 9(8).

Kwon, S.Y., Selin, N.E., Giang, A., Karplus, V.J., Zhang, D., 2018. Present and Future Mercury Concentrations in Chinese Rice: Insights From Modeling. Global Biogeochemical Cycles 32(3), 437-462.

Li, J., Zhou, S., Wei, W., Qi, J., Liang, S., 2020. China's retrofitting measures in coal-fired power plants bring significant mercury-related health benefits. One Earth(3), 777-787.

Liu, K.Y., Wang, S.X., Wu, Q.R., Wang, L., Ma, Q., Zhang, L., Li, G.L., Tian, H.Z., Duan, L., Hao, J.M., 2018. A Highly Resolved Mercury Emission Inventory of Chinese Coal-Fired Power Plants. Environ. Sci. Technol. 52(4), 2400-2408.

Liu, Y., Xing, J., Wang, S., Fu, X., Zheng, H., 2018. Source-specific speciation profiles of PM2.5 for heavy metals and their anthropogenic emissions in China. Environmental Pollution 239, 544-553.

Liu, Z., Guan, D.B., Moore, S., Lee, H., Su, J., Zhang, Q., 2015. Steps to China's carbon peak. Nature 522(7556), 279-281.

Ministry of Ecological Environment of China, 2018. Technical Specification for Ultra-Low Emission Flue Gas Treatment Engineering of CFPPs (HJ 2053-2018). Ministry of Ecological Environment of China, Beijing.

Ministry of Ecology and Environmental of the People's Republic of China, 2014a. The Inventory of Denitration Facilities of Coal-Fired Power Plants in China (In Chinese). Beijing,China.

Ministry of Ecology and Environmental of the People's Republic of China, 2014b. The inventory of desulfurization facilities of coal-fired power plants in China (In Chinese). Beijing,China.

Ministry of Environmental Protection, 2017. Announcement on the entry into force of the Minamata Convention on Mercury. Ministry of Environmental Protection, Beijing,China.

National Bureau of Statistics, 2011. China energy statistical yearbook 2011. National Bureau of Statistics, Beijing.

National Bureau of Statistics of China, 2006. China Energy Statistical Yearbook 2006. Beijing, China.

National Bureau of Statistics of China, 2021. China Energy Statistical Yearbook 2020, in: Press, C.S. (Ed.). Beijing, China.

National Energy Administration, 2020. Carry forward the past and forge ahead into the future and start a new journey of global response to climate change —Speech at the climate ambition summit, in: Xi，Jinping (Ed.). Beijing.

Navratil, T., Vytopilova, M., Vlckova, S., Mrazova, K., Zakharov, S., Honsova, S., Pelclova, D., 2016. Mercury and its Future in the Czech Republic, 36th International Conference on Modern Electrochemical Methods (MEM). Jetrichovice, CZECH REPUBLIC, pp. 143-147.

Pacyna, E.G., Pacyna, J.M., Fudala, J., Strzelecka-Jastrzab, E., Hlawiczka, S., Panasiuk, D., Nitter, S., Pregger, T., Pfeiffer, H., Friedrich, R., 2007. Current and future emissions of selected heavy metals to the atmosphere from anthropogenic sources in Europe. Atmospheric Environment 41(38), 8557-8566.

Pacyna, J.M., Pacyna, E.G., 2001. An assessment of global and regional emissions of trace metals to the atmosphere from anthropogenic sources worldwide. Environmental Reviews 9(4), 269-298.

Pacyna, J.M., Travnikov, O., De Simone, F., Hedgecock, I.M., Sundseth, K., Pacyna, E.G., Steenhuisen, F., Pirrone, N., Munthe, J., Kindbom, K., 2016. Current and future levels of mercury atmospheric pollution on a global scale. Atmos. Chem. Phys. 16(19), 12495-12511.

Peng, W., Wagner, F., Ramana, M.V., Zhai, H., Sma, M.J., Dalin, C., Zhang, X., Mauzerall, D.L., 2018. Managing China's coal power plants to address multiple environmental objectives. Nat. Sustain. 1(11), 693-701.

Pudasainee, D., Seo, Y.C., Sung, J.H., Jang, H.N., Gupta, R., 2017. Mercury co-beneficial capture in air pollution control devices of coal-fired power plants. International Journal of Coal Geology 170, 48-53.

State Statistical Bureau, 2021. China Statistical Yearbook 2020. Beijing.

Sunderland, E.M., Selin, N.E., 2013. Future trends in environmental mercury concentrations: implications for prevention strategies. Environmental Health 12.

Sung, J.-H., Oh, J.-S., Mojammal, A.H.M., Back, S.-K., Lee, E.-S., Kim, S.-H., Seo, Y.-C., 2018. Estimation and Future Prediction of Mercury Emissions from Anthropogenic Sources in South Korea. Journal of Chemical Engineering of Japan 51(9), 800-808.

The Central Committee of the Communist Party of China, 2021. The 21st Meeting of the Central Committee for Comprehensively Deepening Reform. Beijing,China.

The State Council of China, 2007. Notice on Several Opinions on accelerating the shutdown of small thermal power units. Beijing,China.

The State Council of China, 2013. Air Pollution Prevention and Control Action Plan Beijing,China, p. http://www.chinatoday.com.cn/ctchinese/application/environment/air%20plan.html#.

The State Council of China, 2016. Implementation plan of the 13th Five Year Plan (FYP) for comprehensive prevention and control of HMs pollution. Beijing,China.

Tian, H.Z., Liu, K.Y., Zhou, J.R., Lu, L., Hao, J.M., Qiu, P.P., Gao, J.J., Zhu, C.Y., Wang, K., Hua, S.B., 2014. Atmospheric Emission Inventory of Hazardous Trace Elements from China's Coal-Fired Power Plants-Temporal Trends and Spatial Variation Characteristics. Environ. Sci. Technol. 48(6), 3575-3582.

Tian, H.Z., Wang, Y., Xue, Z.G., Cheng, K., Qu, Y.P., Chai, F.H., Hao, J.M., 2010. Trend and characteristics of atmospheric emissions of Hg, As, and Se from coal combustion in China, 1980-2007. Atmos. Chem. Phys. 10(23), 11905-11919.

Tian, H.Z., Zhu, C.Y., Gao, J.J., Cheng, K., Hao, J.M., Wang, K., Hua, S.B., Wang, Y., Zhou, J.R., 2015. Quantitative assessment of atmospheric emissions of toxic heavy metals from anthropogenic sources in China: historical trend, spatial distribution, uncertainties, and control policies. Atmos. Chem. Phys. 15(17), 10127-10147.

Tong, D., Geng, G., Zhang, Q., Cheng, J., Qin, X., Hong, C., He, K., Davis, S.J., 2021. Health co-benefits of climate change mitigation depend on strategic power plant retirements and pollution controls. Nat. Clim. Chang. 11(12), 1077-+.

Tong, D., Zhang, Q., Liu, F., Geng, G., Zheng, Y., Xue, T., Hong, C., Wu, R., Qin, Y., Zhao, H., 2018. Current Emissions and Future Mitigation Pathways of Coal-Fired Power Plants in China from 2010 to 2030. Environ. Sci. Technol. 52(21), 12905-12914.

Wu, Z.J., Ye, H.F., Shan, Y.L., Chen, B., Li, J.S., 2020. A city-level inventory for atmospheric mercury emissions from coal combustion in China. Atmospheric Environment 223, 10.

Xing, J., Lu, X., Wang, S., Wang, T., Ding, D., Yu, S., Shindell, D., Ou, Y., Morawska, L., Li, S., Ren, L., Zhang, Y., Loughlin, D., Zheng, H., Zhao, B., Liu, S., Smith, K.R., Hao, J., 2020. The quest for improved air quality may push China to continue its CO2 reduction beyond the Paris Commitment. Proc. Natl. Acad. Sci. U. S. A.

Zhou, S., Wei, W., Chen, L., Zhang, Z., Liu, Z., Wang, Y., Kong, J., Li, J., 2019. Impact of a Coal-Fired Power Plant Shutdown Campaign on Heavy Metal Emissions in China. Environ. Sci. Technol. 53(23), 14063-14069.

Zhu, C., Tian, H., Cheng, K., Liu, K., Wang, K., Hua, S., Gao, J., Zhou, J., 2016. Potentials of whole process control of heavy metals emissions from coal-fired power plants in China. Journal of Cleaner Production 114, 343-351.