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Neglected environmental health impacts of China's supply-side structural reform

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ABSTRACT

"Supply-side structural reform" (SSSR) has been the most important ongoing economic reform in China since 2015, but its important environmental health effects have not been properly assessed. The present study addresses that gap by focusing on reduction of overcapacity in the coal, steel, and iron sectors, combined with reduction of emissions of sulfur dioxide (SO_2), nitrogen oxide (NO_x), and fine particulate matter ($PM_{2.5}$), and projecting resultant effects on air quality and public health across cities and regions in China. Modeling results indicate that effects on air quality and public health are visible and distributed unevenly across the country. This assessment provides quantitative evidence supporting projections of the transregional distribution of such effects. Such uneven transregional distribution complicates management of air quality and health risks in China. The results challenge approaches that rely solely on cities to improve air quality. The article concludes with suggestions on how to integrate SSSR measures with cities' air quality improvement attainment planning and management performance evaluation.

1. Introduction

While President Trump's announced withdrawal from the Paris Agreement garnered much international attention in June 2017, few outside China noticed an announcement by the Chinese National Reform and Development Commission (NRDC) concerning China's progress toward reduction of overcapacity in coal, steel, and iron – one of the four tasks of the ongoing "supply-side structural reform" (SSSR) in that country. SSSR was first proposed by the Chinese government in 2015, as a major response to the slowing of economic growth in the country since 2007 (marked as the "new normal") and an opportunity to optimize the supply side of the economy. Major tasks identified included cutting overcapacity in coal, steel, and iron; reducing housing inventory; lowering leveraged corporate and industrial financing; cutting corporate costs; and improving weak economic links between increasing new demands and short supply. Different from the deindustrialization (mainly expressed with the declines of industrial outputs and employment) that took place in many industrialized countries since 1980s and 1990s (Feinstein, 1999; Koistinen, 2013), SSSR in China is a government-initiated, organized, and enforced reform package that was considered revolutionary for the country's socialist market economy practices and theories.

In 2015, the State Council set targets for 2020 to reduce production capacity by 150 million tons of crude steel and 800 million tons of coal. Provincial and local governments responded immediately to this call with detailed implementation plans. According to NRDC, the annual targeted reductions in production capacity set for 2016, 45 million tons of steel and 250 million tons of coal, were already overachieved by October of that year. By May 2017, another 32 million tons of steel, and 69 million tons of coal production capacities were reduced, to a cumulative 51% and 40%, respectively, of the targets by 2020, a positive sign for full achievement of the overall targets.

In China until now, the social, economic, and sectoral impacts of this countrywide reform have attracted most of the attention. The reform's environmental and health effects have not yet been properly studied, however. Where environmental protection has been brought into discussions, the focus has been on how more stringent environmental regulations could aid the realization of the SSSR targets; nor has

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there yet been systematic quantitative assessment of environmental and health impacts (MEP, 2016). This was mainly due to SSSR being primarily an economic policy that did not involve the Ministry of Environmental Protection directly in its formulation and implementation. Although such a major policy should be subject to strategic environmental assessment, the latter is only encouraged, not enforced, according to the new Environmental Law (Zhang et al., 2015). The present study aimed to begin to address that gap by focusing on the reduction of overcapacity in the coal, steel, and iron sectors; combined with reduction of emissions of sulfur dioxide (SO_2), nitrogen oxide (NO_x), and fine particulate matter ($PM_{2.5}$); and projecting resultant effects on air quality and public health across cities and regions in China. While efforts and costs to reduce overproduction capacities were local, local air quality also can be affected by atmospheric transport of pollution from distant sources (Zhang et al., 2017). Such quantitative assessments are important sources of evaluating the effectiveness of SSSR efforts. They contribute, as well, to a more balanced understanding of impacts of such structural reforms on air quality and public health across the country.

2. Material and methods

This study uses an integrated framework to trace the policies-to-impacts path to assess the environmental and health benefits of China's SSSR. To the best of our knowledge, this is the first study to evaluate the national environmental and health benefits of this top priority policy of China. In this study, we apply a four-step approach: 1) inventorying the projected reduction in overproduction capacity and 2) air pollutant emissions, respectively; and projecting expected resultant 3) concentrations of air pollutants and 4) health impacts by 2020.

2.1. Data source

For the SSSR policy, the base and target years are 2014 and 2020, respectively. The baseline emission data for each province in 2014 are cited from the *China Statistical Yearbook on Environment 2016* (http://www.stats.gov.cn). Data on the reduction of production capacity of coal, steel and iron by 2020 were collected from the government website of each province, and summarized in Table S1 in the *Supporting Information*. Spatial distribution of the planned production capacity reduction is illustrated in Fig. S1 in the Supporting Information.

Today China is the world's biggest producer and consumer of coal, and relies on coal power for approximately 70% of its energy, with 45% used for the industrial sector and the remainder used to generate electricity (NEA, 2011). The burning of coal is the main source of air pollution. Most coal in China is mined in the less populated western and northeastern regions, and the coal consumption is concentrated in the populated eastern and central China regions. Therefore the inter-provincial redistribution of coal in consumption is important to calculate the reduction of air emissions from SSSR on coal. In this study, the reduction of coal production at the supply side was transformed into the reduction of coal consumption using the inter-province coal trading matrix in China (Tian et al., 2014), and the result was summarized in Table S2 in the Supporting Information. Based on reduction of coal, steel, and iron capacity, the abatement of air pollutant (NO_x, SO_2, $PM_{2.5}$) emissions were calculated and is summarized in Table S3 in the Supporting Information.

2.2. Inventory of air pollutant emissions

To avoid repeated calculation of emissions from industrial coal combustion, the air emissions from steel and iron production only focused on the iron-making and steel-making processes, and the emissions from coking and other coal burning processes in steel and iron production were calculated under the coal combustions. Details on the air emissions calculation are described below.

(1) Sulfur dioxide (SO_2)

In China, coal combustion is the largest atmospheric pollution source. It was estimated that in all kinds of air emission sources in China, about 87% of SO_2 and 67% of NO_x were emitted from coal combustion (Xu et al., 2000). Emission of SO_2 from coal combustion depends on the sulfur content, boiler type, conversion rate of sulfur, and desulfurization efficiency of air emission control devices. The coefficient of SO_2 emission was calculated as:

$$E_{SO2} = 2 \times 1000 \times C_s \times P \times (1 - \rho_{so2}) \tag{1}$$

where,E_{SO2}: emission coefficient of SO_2 (kg/ton)C_s: sulfur content in coal (%)P: sulfur conversion rate (%)ρ_{SO2}: desulfurization efficiency (%)

Based on a large amount of measured and statistical data in China, Wang and Zhang (2012) report that the conversion rate of sulfur in coal combustion ranges from 80% to 85%. In this study, the average between 80 and 85% was used to calculate the SO_2 emission, and 80% and 85% was used as the low and high bound estimation respectively in the uncertainty analysis. In China, the development of specific desulfurization devices was still in the stage of exploration and demonstration, and there was a considerable gap to meet the requirements of practical application (Wang and Zhang, 2012). The wet dust scrubbers that are currently widely used for industrial boilers can make full use of its own emissions of alkaline substances (alkaline substances in the boiler wastewater and boiler ash), and achieve a general desulfurization efficiency of 20%–30% (Wang and Zhang, 2012). The desulfurization efficiency can be further improved by adding additional alkaline substances. Therefore in inventory studies of SO_2 emission, the desulfurization efficiency is usually set between 10 and 50% (Wang and Zhang, 2012). In this study, 30% is used to calculate the average SO_2 emissions, and 10% and 50% are used as the lower and upper bound in uncertainty analysis.

The SO_2 emission factor for coal combustion with different sulfur content was calculated utilizing Eq. (1). In this study, because the coal reduction goals were broken down into provinces, the sulfur content in coal in each province was used, and the SO_2 emission factors of typical sulfur content were summarized in Table S4 in the Supporting Information. The provincial sulfur contents in coal were investigated by China Coal Processing & Utilization Association in 2012 (Sun and Ye, 2012), and their published data are cited and summarized in Table S2.

For SO_2 emissions from iron and steel manufacturing processes, Ma et al. (2012) summarize the SO_2 emission factors in operation units in China iron and steel industry. The values of the key units, including sintering (1.374 kg/ton), pelletizing (0.395 kg/ton) and iron making (0.837 kg/ton), were used to calculate SO_2 emissions from iron and steel manufacturing (Ma et al., 2012).

(2) Nitrogen oxide (NO_x)

The NO_x emissions were calculated using the empirical NO_x emission coefficients (equivalent to NO_2). For coal combustions, the empirical NO_x emission coefficients are cited from Wang and Zhang's (2012) nationwide investigations on coal fired boilers. For iron and steel manufacturing, the NO_x emission coefficients are cited from Duan et al. (2013). These data are summarized in Table S5 in the Supporting Information. Because it was difficult to obtain the information on the percentage of each boiler type in coal combustions, the average value was used to estimate the NO_x emission from coal combustion, and the higher and lower bound were used in the uncertainty analysis.

(3) Particulate matter ($PM_{2.5}$)

Fine particulate matter emission ($PM_{2.5}$) was calculated according to the *Guideline for Primary Fine Particulate Matters Emission Inventory* released by the China Ministry of Environmental Protection (MEP, 2014). The primary $PM_{2.5}$ emission from coal combustion was

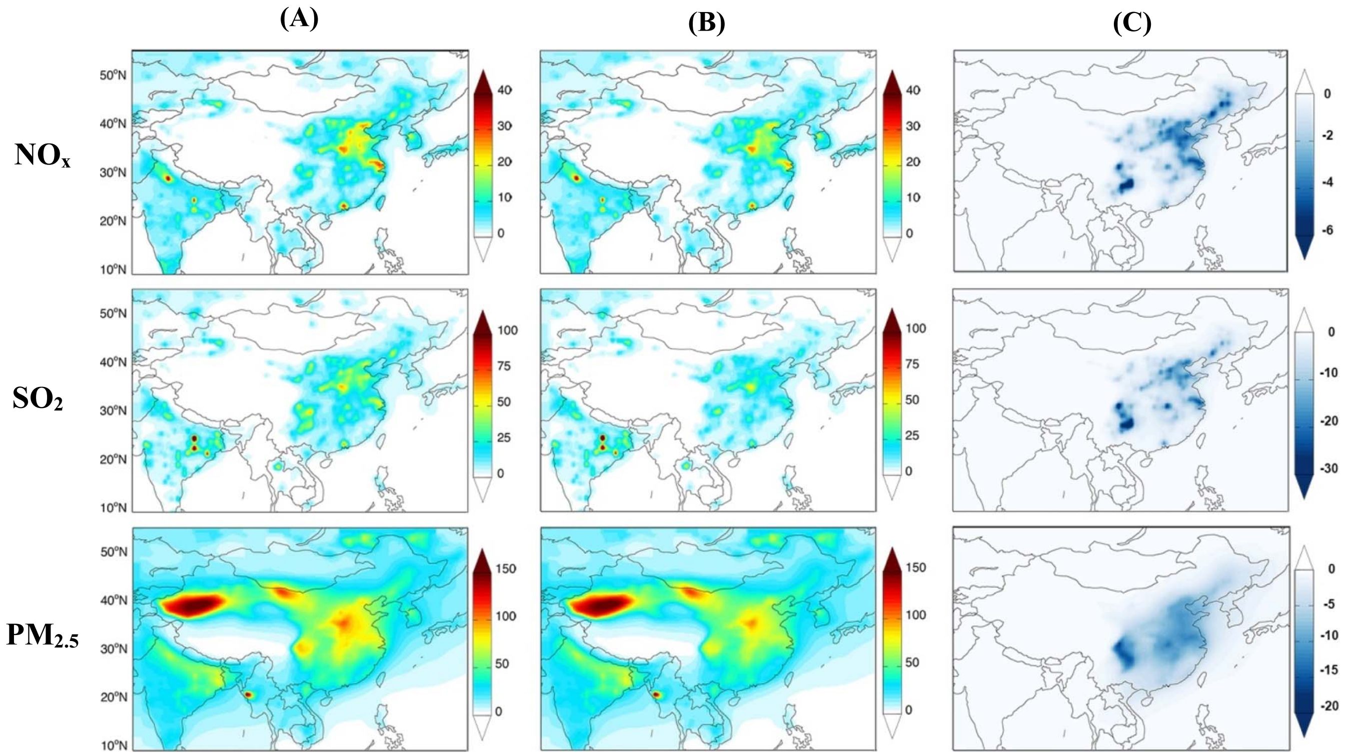

Fig. 1. Air concentrations of NO_x, SO_2 and $PM_{2.5}$ in 2014 (base year) and 2020 (target year), and the reductions by SSSR. (A) presents the concentrations in 2014, which is the base year of this study. (B) presents the concentrations in 2020, which is the target year. (C) is the reductions of air concentrations by SSSR, which are calculated using the concentrations in 2020 minus those in 2014 in the same grid. The unit is μg/m^3.

calculated as:

$$E_{PM2.5} = Aar \times (1 - ar) \times f_{PM2.5} \times (1 - \eta) \times 1000 \tag{2}$$

where,$E_{PM2.5}$: emission coefficient of $PM_{2.5}$ (kg/ton)Aar: ash content as received basis (%)ar: proportion of ash into bottom ash (%)$f_{PM2.5}$: percentage of $PM_{2.5}$ in the total particulate matter produced (%)η: removal efficiency of air pollution control devices (%).

In this study, the values of the proportion of ash into bottom ash (ar) and the percentage of $PM_{2.5}$ in the total particulate matter ($f_{PM2.5}$) recommended in the Guideline (MEP, 2014) were used to calculate $PM_{2.5}$ produced from coal combustions (summarized in Table S6 in the Supporting Information). The air pollutant control devices for particulate matter include fiber filter, electrostatic precipitator, and electrostatic fabric filter. These devices are considered in the calculation, and their removal efficiencies recommended in the Guideline (MEP, 2014) were used in the calculation (Table S6).

The $PM_{2.5}$ emissions from steel and iron manufacturing were calculated as:

$$E_{PM2.5} = EF_{PM2.5} \times (1 - \eta) \tag{3}$$

where:$E_{PM2.5}$: emission coefficient of $PM_{2.5}$ from steel and iron manufacturing process (kg/ton)$EF_{PM2.5}$: emission factor of $PM_{2.5}$ (kg/ton)η: removal efficiency of air pollution control devices (%)

The emission factors of $PM_{2.5}$ from steel and iron manufacturing, as well as the removal efficiency of control devices, were cited from the Guideline (MEP, 2014) and summarized in Table S7 in the Supporting Information.

2.3. Modeling of air pollutant transport and distribution

Transport of air pollutants was simulated using the GEOS-Chem chemical transport model (version 9-02; http://geos-chem.org). The model was driven by the assimilated meteorological fields of GEOS-5 taken from the Global Modeling and Assimilation Office (GMAO) of the National Aeronautics and Space Administration (NASA). It was run with the full O_x–NO_x–CO–VOC–HO_x chemistry and online calculation for various aerosols (sulfate, nitrate, ammonium, BC, OC). Wet scavenging of soluble aerosols and gases in convective updrafts, rainout, and washout follows Liu et al. (2001), with updates for BC by Wang et al. (2011). Dry deposition of gases and aerosols follows Wesely (1989) and L. Zhang et al. (2001), respectively. Mixing in the boundary layer follows a non-local scheme.

We implemented a nested-grid capability with higher resolution for China in the GEOS-Chem model, which has 0.5° × 0.667° horizontal resolution and 47 vertical levels from the surface to 0.01 hPa. We first conducted a global 4° × 4.5° resolution simulation for lateral boundary conditions of the nested model every 3 h and then ran the nested model with an initial spin-up time of three months. The output results were archived daily and monthly to calculate the average concentrations of atmospheric pollutants for analysis.

Natural emissions were cited from Lin (2012). Global anthropogenic emissions were sourced from the Emission Database for Global Atmospheric Research (EDGAR database). Anthropogenic emissions over Asia were sourced from the MIX inventory for various species including SO_2, NO_x, and $PM_{2.5}$, which are at 0.25° × 0.25° horizontal resolutions (Li et al., 2017). The MIX inventory is an anthropogenic emission inventory for Asia. It was developed to support the Model Inter-Comparison Study for Asia Phase III (MICS-Asia III) and the Task Force on Hemispheric Transport of Air Pollution (TF HTAP) projects by a mosaic of up-to-date regional and national emission inventories (Li et al., 2017). For the model simulation, the baseline emission data for each province in 2014 were cited from the *China Statistical Yearbook on Environment 2016* (NBS, 2016). The provincial emissions were scaled to high horizontal resolutions based on the distribution information of the MIX inventories.

3. Results and discussion

3.1. Spatial changes of air pollutant distribution between 2014 and 2020

The base year for SSSR is 2014 and the target year is 2020. The reductions of air concentrations by SSSR policy were calculated using the targeted concentrations for 2020 minus those in the baseline year, 2014. Modeling results indicate that effects on air quality are visible and expected to be unevenly distributed across the country (Fig. 1). Changes in air pollutants are expected to mainly benefit the North China Plain, eastern provinces, and Sichuan Basin.

For the three air pollutants, significant reduction of concentrations are noted in the Jing-Jin-Ji region (including Beijing, Tianjin, Hebei, Shandong, and Shanxi provinces), which is well-known for its heavy air pollution in China. In the Jing-Jin-Ji region, the average projected reduction of NO_x under the SSSR program is 2.71 μg/m^3, SO_2 is 10.57 μg/m^3, and $PM_{2.5}$ is 7.77 μg/m^3. The Yangtze River Delta area (including Shanghai, Jiangsu, Zhejiang, Anhui, and Hubei provinces) also are projected to benefit from air quality improvement. In the Yangtze River Delta area, the average reduction of NO_x, SO_2 and $PM_{2.5}$ is projected at 2.22 μg/m^3, 8.94 μg/m^3 and 8.94 μg/m^3, respectively. The projected reduction of $PM_{2.5}$ concentration in the Yangtze River Delta area is larger than that in the Jing-Jin-Ji region. The Jing-Jin-Ji region is the largest contributor to the SSSR capacity reduction targets, with the total reduction of production capacity of 21,563 tons of coal, and 13,109 tons of iron and steel. The total reduction of production capacity in the Yangtze River Delta area is projected at 4783 tons of coal and 3239 tons of iron and steel. Chuan-Gui region (including Sichuan, Chongqing, and Guizhou provinces) is also a major region with reduction of coal production capacity (totally 12,603 tons) under the SSSR program. In this region, the concentration reduction of NO_x, SO_2 and $PM_{2.5}$ is projected at 1.46 μg/m^3, 8.61 μg/m^3 and 7.36 μg/m^3, respectively, lower than the Yangtze River Delta area. These results suggest that areas with projected improved air quality are not consistent with areas with source air emissions reduction, and interregional collaborations are necessary.

3.2. Health impact from reduction of $PM_{2.5}$ exposure

Our quantitative assessment of the public health impacts resulting from the national implementation of SSSR in China focuses on the projected mortality burden associated with exposure to ambient $PM_{2.5}$. While the other air pollutants modeled in this study, including NO_x and SO_2, are also known to have adverse health effects, $PM_{2.5}$ is widely regarded to have the greatest health impacts, mainly due to its effects on cardiopulmonary-related premature deaths. Here we used a log-linear health impact function to combine modeled $PM_{2.5}$ concentrations, the affected population, the baseline incidence rates, and concentration-response relationships from epidemiologic literature, and estimated the changes in cause-specific mortality resulting from the change in air pollution levels as a result of SSSR:

$$\Delta y = y_0 \times (1 - e^{-\beta \cdot \Delta x}) \times \mathrm{Pop} \tag{4}$$

where,Δy: the change in the number of cause-specific deathsβ: concentration-response coefficient from epidemiological studies (increase in the cases of the health endpoint assessed per unit increase in $PM_{2.5}$ concentration)Δx: model predicted change in $PM_{2.5}$ concentration in 2020 relative to the base year 2014y_0: the baseline incidence for each health endpoint assessedPop: the affected population

Eq. (4) is the standard damage function used by the U.S. EPA to relate changes in air pollution exposure to health and productivity, and has been applied to assess burden of disease attributed to ambient air pollution in other international countries and global public health burden (Anenberg et al., 2010).

Health impact assessment for ambient air pollution generally use risk estimates from cohort studies, which investigate the long-term chronic effects of exposure, to link human exposure to adverse health outcomes. However, epidemiologic cohort studies have been conducted largely in western developed countries such as the United States, and currently not available in China (Liu et al., 2017). The cohort studies conducted in developed countries reported relative risk of all-cause and cause-specific mortality per unit $PM_{2.5}$ exposure for annual ambient average $PM_{2.5}$ concentrations ranging from approximately 5 to 30 μg/m^3, which may not be applicable to Asian developing countries, such as China, where annual average exposures can exceed 100 μg/m^3 (Burnett et al., 2014). For the present study, we selected the risk estimates from a recent study that developed $PM_{2.5}$ mortality relative risk (RR) functions over the entire global exposure range (Burnett et al., 2014). These RR functions have been used in the latest global burden of disease study (Cohen et al., 2017), some recent studies of health impacts of global air pollution (Zhang et al., 2017), and also studies of health impacts in individual developing countries in Asian, such as China (Liu et al., 2017) and India (Chowdhury and Dey, 2016). Burnett et al. (2014) reported risk estimates for China for four causes of mortality in adults: ischemic heart disease (IHD), cerebrovascular disease (stroke), chronic obstructive pulmonary disease (COPD), and lung cancer (LC). We excluded COPD deaths because the mortality data for COPD were not reported in China. Also, we applied these RRs to the entire population instead of adults only, since the age-specific population projection and baseline incidence rates were not available. The bias introduced by this simplification was considered acceptable because most of the deaths due to IHD, stroke and LC occurred in adults (Liu et al., 2017).

The annual incidence rates of IHD, stroke and LC in the base year 2014 were obtained from *China Health Statistics Yearbook 2015*, and assumed unchanged in 2020. We obtained the projected population in China in 2020 on the same grid created for simulating $PM_{2.5}$ concentrations, using the gridded population data from NASA Socioeconomic Data and Applications Center (https://doi.org/10.7927/H4SF2T42). All people living within each grid cell were assumed to be exposed to the same pollutant concentration. The change in the number of cause-specific deaths was calculated in each grid using Eq. (4), and then aggregated across all the grid cells to estimate to total mortality burden by province and in the entire nation. The total $PM_{2.5}$-related mortality was calculated as the summary of deaths due to IHD, stroke and LC.

We estimate that SSSR will avoid approximately 38,365 (95% CI: 14996, 64,918) $PM_{2.5}$-related premature deaths in China in 2020 relative to business-as-usual. These deaths include 14,486 (95% CI: 2518, 28,318), 22,047 (95% CI: 3629, 44,479), and 1833 (95% CI: 793, 2914) excess deaths due to LC, IHD and stroke, respectively. The avoided $PM_{2.5}$-related premature deaths from the national implementation of SSSR by 2020 (at provincial level) are presented in Fig. 2, and the data in high spatial resolution is illustrated in Fig. S2 in the Supporting Information. Avoided mortality is projected to be greatest in the Yangtze River Delta, Pearl River Delta, Beijing–Tianjin Metropolitan Region, Sichuan Basin, and Wuhan Metropolitan Region, which had the highest $PM_{2.5}$ air pollution levels in the base year 2014, high population density, or both. At the provincial level, our estimates suggest that four provinces (Shandong, Sichuan, Jiangsu and Henan) will have annual avoided total deaths over 3000, and another three provinces (Anhui, Hebei, and Guangdong) will have annual avoided total deans between 2000 and 3000 (Fig. S3 in the Supporting Information). Together, these seven provinces account for more than 50% of total projected avoided $PM_{2.5}$-related deaths by 2020 (Table 1).

3.3. Uncertainty analysis and limitations

The impact of the parameter uncertainty along the policies-to-impacts pathway on the model output can be investigated through an uncertainty analysis. Uncertainty analysis based on numerical techniques is usually used for complex models. The widely used numerical techniques include: 1) sensitivity analysis; 2) mean value first-order

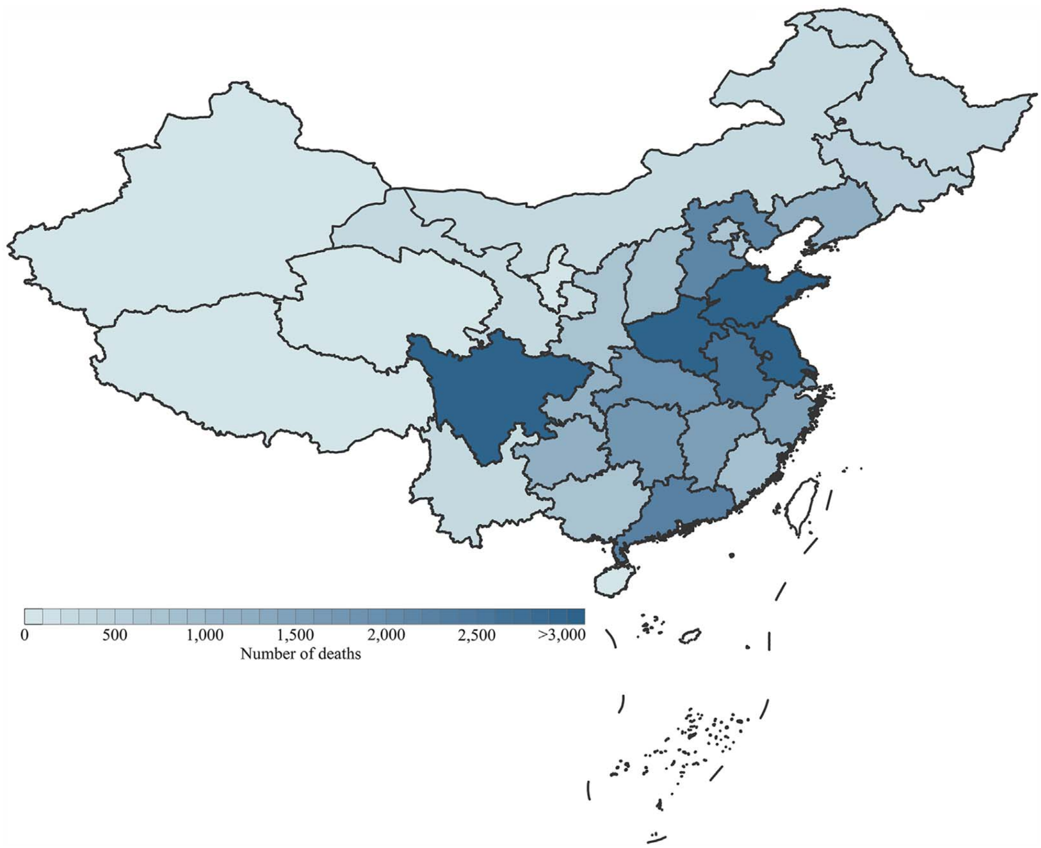

Fig. 2. Avoided $PM_{2.5}$-related premature deaths from the national implementation of SSSR by 2020 (at provincial level. Unit: number of deaths).

Table 1
Avoided deaths associated with reduced exposure to ambient $PM_{2.5}$ resulting from the national implementation of SSSR in 2020 (at provincial level, unit: number of death).

Province	Lung cancer	Ischemic heart disease	Cerebrovascular disease	Total mortality
Beijing	299	454	38	792
Tianjin	287	435	36	758
Hebei	812	1235	103	2150
Shanxi	287	437	36	760
Inner Mongolia	78	120	10	207
Liaoning	440	671	55	1166
Jilin	173	265	22	460
Heilongjiang	123	188	15	325
Shanghai	553	841	70	1464
Jiangsu	1170	1775	148	3093
Zhejiang	567	862	71	1501
Anhui	1000	1517	127	2643
Fujian	314	479	40	834
Jiangxi	594	903	75	1572
Shandong	1451	2204	183	3838
Henan	1145	1739	145	3029
Hubei	722	1096	92	1910
Hunan	668	1016	85	1768
Guangdong	850	1294	108	2252
Guangxi	269	411	34	715
Hainan	22	33	3	57
Chongqing	421	639	53	1113
Sichuan	1268	1952	160	3380
Guizhou	445	677	56	1179
Yunnan	113	172	14	299
Tibet	0	0	0	0
Shaanxi	290	441	36	767
Gansu	95	146	12	254
Qinghai	8	12	1	22
Ningxia	22	32	3	56
Xinjiang	1	2	0	3

second moment analysis; 3) point estimate method; 4) Monte Carlo simulation; and 5) Mellin transform. Giang and Selin (2016) used bounding assumptions along the policies-to-impacts pathway to assess the relative influence of various uncertainties. In some other studies, uncertainties were investigated through parameters variability in exposure–response functions (Rice et al., 2010).

The present study combines the bound assumptions and parameters variability to identify the uncertainties in the environmental and health benefits. In the path from policies to emission inventories, key assumptions include: 1) the sulfur conversion rate and desulfurization efficiency for SO_2 emission; 2) empirical coefficient for NO_x emission; and 3) proportion of ash into bottom ash, percentage of $PM_{2.5}$ in the total particulate matter produced, removal efficiency of APCDs for $PM_{2.5}$ emission. The lower and higher bounding estimations of the reduction of air pollutant concentrations were listed in Table 2. In the assessment of the public health impacts of $PM_{2.5}$, the key parameters mainly included the concentration-response coefficients linking $PM_{2.5}$ exposure to health effects. According to Burnett et al., the concentration-response coefficients follow normal distribution, with $0.33 \pm 0.18\%$ for lung cancer, $0.21 \pm 0.13\%$ for ischemic heart disease, and $0.33 \pm 0.11\%$ for cerebrovascular disease (Burnett et al., 2014). We conducted Monte Carlo simulations with 10,000 iterations. The result of the uncertainty analysis is summarized in Table 3.

It is noteworthy that this study only estimates environmental health impacts of part of SSSR – measures to reduce overcapacity in the coal, steel, and iron sectors, specifically. Due to lack of data on other SSSR measures, we were unable to develop a complete picture of the reform impacts. Based on our modeling, all the selected SSSR measures, intended to make China's economy more competitive and healthier, would potentially improve air quality and the associated public health. This assessment unveils only partial effects of SSSR, providing quantitative evidence supporting projections of the transregional distribution of such effects. We believe that good understanding of such effects of

Table 2
The average, lower and higher bounding estimations of the reduction of air pollutant concentrations in 2020 compared with 2014.

Province	NO_x (μg/m^3)			SO_2 (μg/m^3)			$PM_{2.5}$ (μg/m^3)		
	Average	High	Low	Average	High	Low	Average	High	Low
Beijing	2.54	3.05	2.04	9.19	11.81	6.57	7.29	9.10	5.52
Tianjin	3.30	3.99	2.63	14.15	18.17	10.11	9.61	12.03	7.24
Hebei	2.55	3.07	2.03	10.30	13.23	7.36	7.28	9.12	5.50
Shanxi	2.18	2.60	1.75	8.41	10.81	6.01	5.28	6.58	4.01
Inner Mongolia	0.31	0.37	0.25	1.05	1.35	0.75	0.96	1.21	0.71
Liaoning	2.22	2.65	1.78	6.93	8.90	4.95	6.27	7.87	4.72
Jilin	1.74	2.09	1.40	4.36	5.59	3.12	3.95	4.94	3.00
Heilongjiang	0.80	0.96	0.65	1.37	1.76	0.98	2.06	2.54	1.58
Shanghai	3.53	4.29	2.82	14.46	18.57	10.34	9.23	11.70	6.89
Jiangsu	2.99	3.61	2.39	11.38	14.59	8.16	10.16	12.75	7.67
Zhejiang	1.42	1.70	1.15	4.80	6.14	3.45	6.29	7.91	4.74
Anhui	2.07	2.48	1.67	6.61	8.46	4.75	10.23	12.72	7.80
Fujian	0.77	0.91	0.62	2.62	3.33	1.89	4.76	6.02	3.55
Jiangxi	1.27	1.52	1.03	5.56	7.11	4.00	7.84	9.75	5.99
Shandong	2.98	3.59	2.38	10.78	13.84	7.72	9.38	11.77	7.08
Henan	0.92	1.09	0.75	5.51	7.07	3.94	8.35	10.36	6.37
Hubei	1.09	1.30	0.88	7.46	9.55	5.35	8.76	10.90	6.67
Hunan	0.36	0.42	0.30	3.22	4.11	2.32	6.66	8.23	5.12
Guangdong	0.73	0.87	0.59	3.09	3.95	2.21	4.47	5.58	3.38
Guangxi	0.10	0.11	0.08	1.53	1.96	1.10	3.75	4.63	2.89
Hainan	0.05	0.05	0.04	0.32	0.41	0.23	1.65	2.07	1.24
Chongqing	0.93	1.06	0.81	7.81	9.96	5.62	9.04	11.20	7.03
Sichuan	0.69	0.83	0.56	4.24	5.40	3.06	4.17	5.20	3.19
Guizhou	2.75	2.98	2.53	13.78	17.27	10.00	8.87	11.83	6.70
Yunnan	0.42	0.49	0.34	1.52	1.92	1.11	1.47	1.87	1.10
Tibet	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.02	0.01
Shaanxi	1.08	1.29	0.87	4.70	6.03	3.36	4.89	6.06	3.73
Gansu	0.45	0.54	0.36	1.38	1.77	0.99	1.38	1.72	1.04
Qinghai	0.04	0.05	0.03	0.09	0.11	0.06	0.15	0.19	0.11
Ningxia	1.30	1.55	1.05	3.56	4.56	2.55	2.18	2.75	1.63
Xinjiang	0.00	0.00	0.00	0.01	0.01	0.00	0.04	0.05	0.03
Jing-Jin-Ji Region	2.71	3.26	2.17	10.57	13.57	7.55	7.77	9.72	5.87
Yangtze River Delta	2.22	2.68	1.78	8.94	11.46	6.41	8.94	11.20	6.76
Chuan-Gui Region	1.46	1.62	1.30	8.61	10.88	6.23	7.36	9.41	5.64

Table 3
The uncertainty of the estimated avoided mortality burdens associated with reduced exposure to ambient $PM_{2.5}$.

Disease	Average	High	Low
Lung cancer	14,486 (95% CI: 2518, 28,318)	18,182 (95% CI: 3131, 35,661)	10,966 (95% CI: 1929, 21,364)
Ischemic heart disease	22,047 (95% CI: 3629, 44,479)	27,640 (95% CI: 4515, 55,896)	16,708 (95% CI: 2779, 33,623)
Cerebrovascular disease	1833 (95% CI: 793, 2914)	2299 (95% CI: 991, 3664)	1388 (95% CI: 603, 2202)
Total mortality	38,365 (95% CI: 14996, 64,918)	48,122 (95% CI: 18739, 81,586)	29,062 (95% CI: 11407, 49,075)

economic policies and their temporal spatial distributions is crucial for evidence-based precise air quality and health management.

3.4. Implications for SSSR policy and environmental health management

China's "Airpocalypse" has made air quality the top environmental concern in recent years for both the general public and the government. $PM_{2.5}$ has become the fourth prominent threat to the health of Chinese people (next to unhealthy dietary, hypertension, and smoking) (Zhang et al., 2016). Air quality management is well known as a wicked environmental problem. The fact that air quality in a given location can be substantially affected by atmospheric transport of pollution from distant sources (Zhang et al., 2017) has implications to China's environmental health policies and management.

In the current central-local relationship in China, the pressure-based environmental regulatory approach has been stretched to its limits when it comes to air quality management. Initial monitoring of $PM_{2.5}$ was conducted in 74 key cities; today it covers most prefectural cities. Cities are the major implementers of national and local environmental regulations in China, thus are under pressure from both upper level government and the public. The revised Air Law 2016 makes it mandatory for cities that fail to meet air quality standards to make time-bound air quality improvement plans. According to the timetable proposed by the Ministry of Environmental Protection in 2013, all Chinese cities are expected to meet the Class II annual limit of the National Ambient Air Quality Standard by 2030; this is currently in a pilot phase. Many Chinese cities lack capacity to produce science-based and cost-effective planning that can effectively cope with transregional effects. Even cities with more effective environmental controls worry about pollution from distant sources – including Shenzhen, the first metropolitan city in China to meet the new national standards and which announced at the end of 2015 its goal of further improvement toward the WHO air quality standard by 2020. This raises questions about scales and integration of various planning efforts; about what kinds of monitoring, supervision and coordination should be developed to have a fair assessment of the air quality management performance of cities; and about what types of policy instruments (e.g. incentives, subsidies and penalties) are most effective in achieving intended results.

This study's findings suggest several policy recommendations: 1) short-term and long terms benefits and distributions of SSSR measures should be monitored and assessed at the national level to support decision making at that level and coordination/negotiation between regions, provinces and cities; 2) local SSSR efforts should be fully integrated, both planning procedure and contents, into cities' air quality improvement attainment planning and management performance evaluation; 3) incentives, such as cross regional payment/compensation for environmental services, should be provided to motivate local governments to take actions where overall benefits are maximized; and 4) costs and benefits assessment to public health should be enhanced to attract broader participation of non-governmental actors. As with other SSSR measures, reduced production and consumption of coal, steel, and iron has co-benefits of improving air quality, reducing emission of greenhouse gases, and improving public health; such co-benefits deserve further studies.

4. Conclusion

Supply-side structural reform is the most important ongoing economic reform in China. Outside of China, SSSR has been either understudied or mistaken as deindustrialization. Its environmental and health effects have been less studied than its social and economic effects, This study provides quantitative evidence of the projected environmental health impacts of China's industrial policy reform. Cumulative effects of SSSR from 2014 to 2020 on air quality and public health are projected across cities and regions of China. Transregional distribution of such effects may further complicate the air quality management and health risk management in China. The results challenge current approaches that rely solely on cities to improve air quality. To minimize offsetting effects, evaluation of cities' performance in air quality management needs to take into consideration the uneven distribution of costs and benefits of both the SSSR measures and social, economic, and environmental health benefits.

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Appendix A. Supplementary data

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