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The health costs of the industrial leap forward in China: Evidence from the sulfur dioxide emissions of coal-fired power stations[☆]

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ABSTRACT

In this study, we attempt to empirically test the effects of air pollution on public health in China. Using three-stage least squares (3SLS) to solve the potential endogeneity problem in sulfur dioxide (SO_2) emissions, we find that air pollution has significant negative effects on public health. Specifically, a 1% increase in SO_2 emissions is found to lead to 0.067 and 0.004 more deaths per 100,000 population due to respiratory diseases and lung cancer, respectively. In terms of absolute magnitude, every one million ton increase in SO_2 emissions results in 0.735 and 0.052 extra deaths due to respiratory diseases and lung cancer per 100,000 population, respectively. Moreover, SO_2 emissions result in 230,000 extra deaths every year and the related economic costs over the study period amount to RMB 8.179 billion.

1. Introduction

After three decades of unbridled and rampant economic growth, China now faces numerous serious environmental challenges. Notably, China is the world's largest source of sulfur dioxide (SO_2) emissions, 16 of the 20 cities with the world's worst air quality are situated in China, and around 54% of the water in the seven main rivers in China is unsafe for human consumption (World Bank, 2007). This intense environmental degradation is beginning to have serious effects on the health of the Chinese population.[1] For example, air pollution is now the fourth biggest contributor to disease in China (Murray et al., 2012). The mortality of lung cancer, which is the most common form of cancer in China and is closely associated with air quality, has increased by 465% since 1978 and accounted for 25% of the total deaths due to cancers in 2008 (Chen, 2008). It is well known that a decline in the level of public health leads to reduced life expectancy, which in turn harms the long term economic development of a country. Accordingly, the health effects of environmental degradation and pollution are a significant government and public concern, and therefore warrant substantial research attention.

Most existing research on the health effects of environmental degradation has been based on routinely collected survey data from developed countries, and few systematic investigations of such effects have been conducted in developing countries.[2] Considering the

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[1] The rapid economic development in China has increased incomes and life expectancy (it reached an average of 75 years in 2012) (WHO, 2012). However, the World Bank predicted that if the related non-communicable diseases such as lung cancer are not addressed, China could see a fall in life expectancy: the average span of healthy life, at 66 years, is already proportionally lower than in many other countries (Wang, Marquez, & Langenbrunner, 2011, p.8).

[2] See Brunekreef and Holgate (2002). For details, see Section 2.

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severe environmental pollution due to the rapid industrialization in such countries, exploring the health effects of air pollution becomes critical for policy design. Moreover, the previous literature has not adequately addressed the endogeneity of gas exposure factors and hence has not convincingly established the causal relationship between poisonous gas emissions and health. The measurement bias and the potential omitted variables would underestimate the effects. Such problems to some extent explain why the existing evidence on the effects of SO_2 emissions on public health is inconsistent. Given that SO_2 pollution is the focus of many air quality regulations, this paper attempts to use the systematic data to examine the relationship between SO_2 emissions and public health in China.[3]

In order to meet the increasing demands of energy, the Chinese government planned to achieve an electric power balance by 2006. In 2005, the local government launched an industrial leap forward program: 1056 new power plants were built and total nameplate capacities increased by 63 million kilowatts. The resulting 11.76% increase of SO_2 emissions made China become one of the few countries with highest SO_2 emissions in the world (Su, Li, Cui, & Tao, 2011). Thus, the variation in power plants emissions due to the new industrial policy offers a good opportunity for us to identify the effects of environment changes on public health. To address the above-mentioned endogeneity problem, we exploit a three-stage least squares (3SLS) method, using coal reserves as an indicator of the scale of power plants, to examine the effects of SO_2 emissions on public health.

Based on a nationwide panel dataset covering 161 prefectural-level municipalities in China from 2004 to 2010, we find that the emissions of poisonous gases significantly affect the health of the population. Specifically, a 1% increase in SO_2 emissions results in around 0.067 and 0.004 extra deaths due to respiratory diseases and lung cancer, respectively, per 100, 000 population. Moreover, 1 million ton increase in SO_2 emissions will lead to 0.735 and 0.052 extra deaths from respiratory diseases and lung cancer, respectively, which means that the conventional ordinary least squares (OLS) estimation underestimates the effects by about 16.42%. In terms of lives lost and treatment costs, around 230,000 people are predicted to die prematurely each year as a result of excessive SO_2 emissions and the related treatment costs will be in excess of RMB 8.179 billion.[4] This effect is relatively more significant during the first three years after a plant is built.

Our findings have significant implications. First, estimating the magnitude of the costs of air pollution is a fundamental question which is still less studied in developing countries (Greenstone & Hanna, 2014). Note that China is the biggest emitter of greenhouse gases and developing country (Harvey, 2017; Hornby & Buck, 2017), estimating the effects of air pollution and related treatment costs of China could provide new insights into this question. Second, calculating the treatment costs may not only help the government design the optimal regulatory policy and the thresholds for particulates pollution, but also provide valuable references to environmental compensation and health insurance. Finally and more importantly, our findings prompt us to rethink China's personnel control system. The existing incentive of central government's personnel control overemphasizes the officials' performance in promoting economic growth (Li & Zhou, 2005; Yao & Zhang, 2015), which causes the conflict among industrial leap, environmental quality and human health. The results of this paper clearly demonstrate the environmental and health costs of such system, and more generally, the inadequate and unbalanced development of China. We believe that introducing the environmental factors into the performance evaluation system would undoubtedly drive officials concerning the potential environmental risks in the process of policy making. Doing so not only helps us improve the public decision-making qualities, but also reduce the risks of social instability, and finally promote the sustainable economic growth of China.[5]

The remainder of this study is organized as follows. In Section 2, we review the related literature on the relationship between air pollution and public health. Section 3 provides an overview of the power plant policies and public health in China. The data and measurements are described in Section 4. We present our empirical analysis in Section 5. Section 6 concludes the study.

2. Literature review

Epidemiologic studies over the past 40 years have shown that the concentrations of poisonous gases in the air are associated with worse health outcomes (Brunekreef & Holgate, 2002; Cohen & Pope III, 1995; Hoek, Brunekreef, Goldbohm, Fischer, & van den Brandt, 2002; Lee et al., 2000; Nemery, Hoet, & Nemmar, 2001; Ware et al., 1986). A large amount of time-series or cohort studies on air pollution effects use fine particles as exposure pollutant. In the U.S., for instance, two prospective cohort studies have corroborated the claim of the higher ambient levels of fine particles and sulfate resulted in excess mortality (Dockery et al., 1993; Pope III et al., 1995). These studies indicated that each 10 $\mu g/m^3$ elevation in fine particulate air pollution was associated with approximately a 6.2%–16% increased risk of all-cause mortality (Abbey et al., 1995; Krewski et al., 2000; Laden, Schwartz, Speizer, & Dockery, 2006; Pope III et al., 2002). Similar results were also found in studies based on data from Europe, Australia, and Japan etc. (Hoek et al., 2002; Omori, Fujimoto, Yoshimura, Nitta, & Ono, 2003; Raaschou-Nielsen et al., 2013; Simpson et al., 2005).

Recently, a blossoming number of epidemiologic literature turned to focus on the health effects of specific pollutants such as SO_2 emissions on the respiratory diseases and lung cancer. For instance, based on the multi-city time-series data of Europe, Katsouyanni and Pershagen (1997) estimated that an increase of 50 $\mu g/m^3$ in SO_2 was associated with about 0.8%–3% increase in daily mortality. In a large cohort of nonsmoking Californians followed over the period 1977–1992, SO_2 showed a strong relationship with lung cancer mortality (Abbey et al., 1999). Research on cities of China also estimated an increase of 10 mg/m^3 in SO_2 corresponded to a

[3] Only a few studies of SO_2 emissions have been conducted in China and the data of such studies were often from one city or just a few cities of China (Kan & Chen, 2003; Venners et al., 2003).

[4] It has been deflated to 2010 RMB using CPI index (China Statistical Yearbook, 2016).

[5] For instance, a series of high-profile protests against PX (paraxylene) projects has occurred in the cities of Xiamen (2007), Dalian (2011), Ningbo (2012), and Kunming (2013).

respective increase in relative risk of mortality from all causes of 1.014 (Kan & Chen, 2003) and 0.01% increase in respiratory mortality (Aunan & Pan, 2004).

However, the research design of cohort-based and time-series studies may face endogeneity problem (Graff Zivin & Neidell, 2013). Some economic studies turned to employ quasi-experimental techniques or 2SLS to develop the causal estimates of the effects of air pollution on health. For instance, Chay and Greenstone (2003a) estimated the impact of total suspended particulates (TSPs) on infant mortality by using the 1981–1982 recession in US as a quasi-experiment. They found that a 1% reduction in TSPs results in a 0.35% decline in the infant mortality rate at the county level. Chay and Greenstone (2003b) analyzed the effects of Clean Air Act Amendments on infant health. By using nonattainment status as an instrumental variable for TSPs changes, they estimated that a 1% decline in TSPs results in a 0.5% decline in infant mortality rate. Chen, Ebenstein, Greenstone, and Li (2013) exploited a regression discontinuity design and showed that the winter heating policy in northern China greatly increased the TSPs and decreased life expectancy: on average, chronic exposure to an additional 100 $\mu g/m^3$ of TSPs was found to lead to a reduction in life expectancy of about three years.

Some other economic studies uncovered the different pattern. For instance, similarly using the Clean Air Act of 1970 as an experiment, Chay, Dobkin, and Greenstone (2003) found that, regulatory status was associated with large reductions in TSPs pollution but has little association with reductions in either adult or elderly mortality. Currie and Neidell (2005) based on within zip code variation in pollution levels to estimate that the reductions in carbon monoxide and particulates over the 1990s in California saved over 1000 infant lives but little evidence was observed of pollution effects on fetal deaths, low birth weight or short gestation.

The controversial discussion also existed between SO_2 emissions and public health. For example, after controlling for other pollutants, several studies have reported statistically significant associations between ambient SO_2 and factors such as respiratory admissions and lung cancer (Burnett et al., 2000; Cao & Gao, 2012; Chen et al., 2012; Hajat, Armstrong, Wilkinson, Busby, & Dolk, 2007; Lee et al., 2002; Sunyer, Ballester, et al., 2003). However, Sunyer, Atkinson, et al., 2003 showed that the statistical significance between SO_2 emissions and respiratory admissions disappeared after adjusting for particulates. Nyberg et al. (2000) observed no association between the SO_2 and lung cancer based on the long-term individual data from Stockholm.

Given the fact of the inconsistency of effects of SO_2, it is worthy to continue to explore its relationship with public health. More importantly, existing literature of either cohort studies or economic studies based on exogenous shocks mainly focuses on the developed countries with the relatively low ambient levels of pollution. Research on China is still limited.[6] Note that China is experiencing rapid industrialization and is one of the few countries with highest SO_2 emissions in the world (Su et al., 2011). Thus, examining the relationship between air pollution and public health at high pollutant levels would be valuable.

3. Background

In China, 70% of energy is supplied by coal combustion, which is the main source of airborne SO_2 (Lu et al., 2010; Zhao et al., 2009; Zhao et al., 2012). The rise in the level of coal combustion can be further attributed to national industrial restructuring since 2000, along with rapid urbanization and the increasing demand for energy (Li, Zhang, Wang, Zhao, & Ma, 2012). To meet the soaring energy demand, China set a goal to achieve an electric power balance by 2006 in its Tenth Five-Year Plan. In response, the local governments launched an industrial leap forward program in 2005. In that year, the total nameplate capacities increased by 63 million kilowatts, which directly led to SO_2 emissions rising to 25,494 thousand tons, an increase of 3000 thousand tons over the previous year (Fig. 1). Accordingly, the variation in power plant emissions due to the new industrial policy offers a good opportunity for us to identify the effects of environment changes on public health.

Fig. 1 plots the nameplate capacity of power plants and SO_2 emissions in China over time. Fig. 2 shows the close spatial association between the number of power plants and SO_2 emissions; in general, the areas with greater SO_2 pollution have more power plants. Therefore, these spatial and temporal variations allow us to identify the causal mechanism of the increased SO_2 pollution.

4. Data

We assemble a panel dataset covering 135 of 284 prefecture-level municipalities in China in 2004, 2006, 2008, 2009, and 2010.[7] There are two proxies for the level of public health: the number of deaths due to respiratory diseases per 100,000 population (*DEATH-RES*) and the number of deaths due to lung cancer per 100,000 population (*DEATH-CAN*). The first proxy is the sum of all the deaths due to diseases of the respiratory system, which include influenza, pneumonia, other acute lower respiratory infections, chronic lower respiratory diseases, and other diseases of the respiratory system. The second proxy, lung cancer, is not only closely related to the air quality but also has the highest cancer mortality rate, with 0.18 deaths per 100,000 population, which is twice the average number (Center for Disease Control, CDC hereafter, 2010). The data are extracted from the annual reports of the disease surveillance points (DSPs) system held by the CDC. The key explanatory variable is the amount of SO_2 emissions. Fig. 3 plots the spatial distribution of SO_2 concentrations and the mortality rates of respiratory diseases and lung cancer in our sample.

In addition to the air quality, existing literature has indicated that a number of other important variables influence public health,

[6] The only exception is Tanaka (2015) which explores the impact of environmental regulations in China on infant mortality.

[7] The Disease Surveillance Points (DSP) contains mortality and morbidity data for China. Surveillance conducted at 161 voluntary sites in urban and rural areas. Our database consists of all the 135 municipalities covered in the surveillance. For those municipalities with more than one site, we calculate the mean of the mortality in different sites.

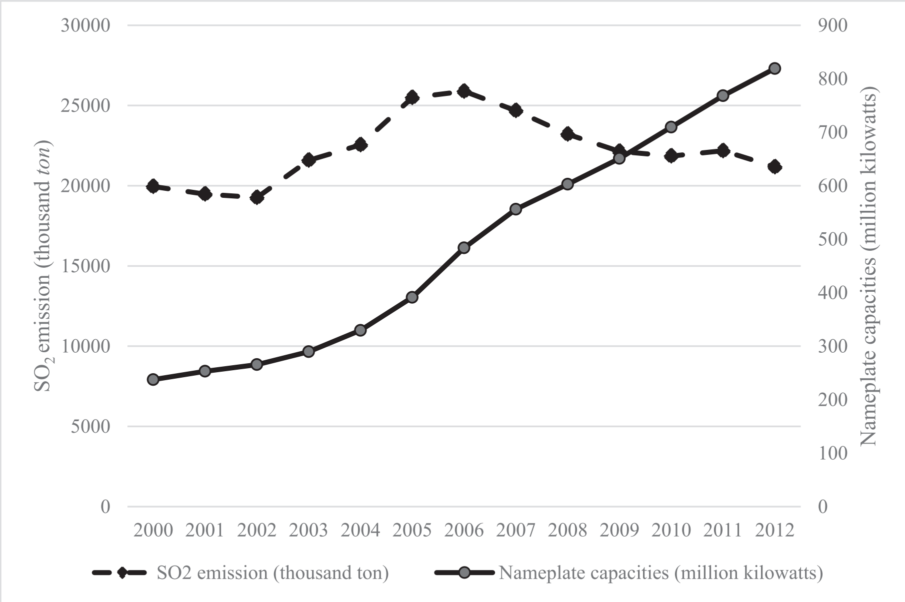


Fig. 1. SO_2 emissions and the nameplate capacity of power plants in China.
Source: China Statistical Yearbook; China Electric Power Yearbook; Compilation of Statistical Materials of Electric Power Industry.

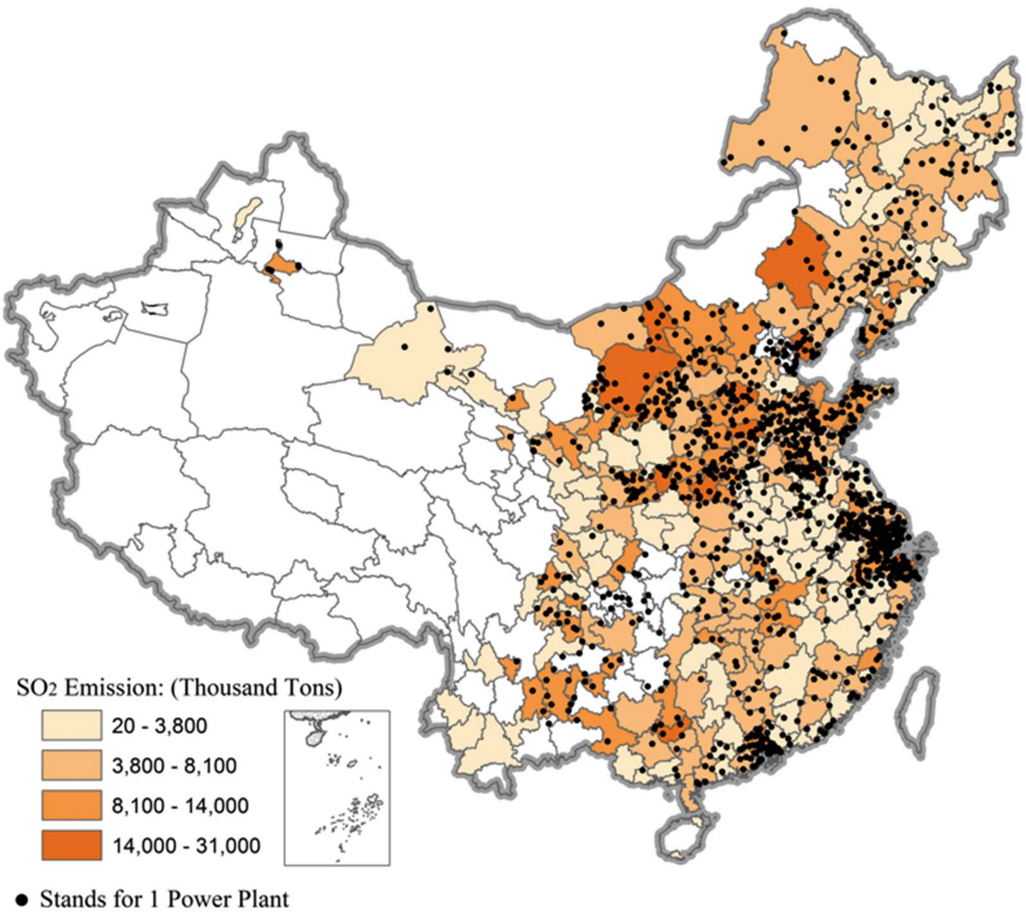
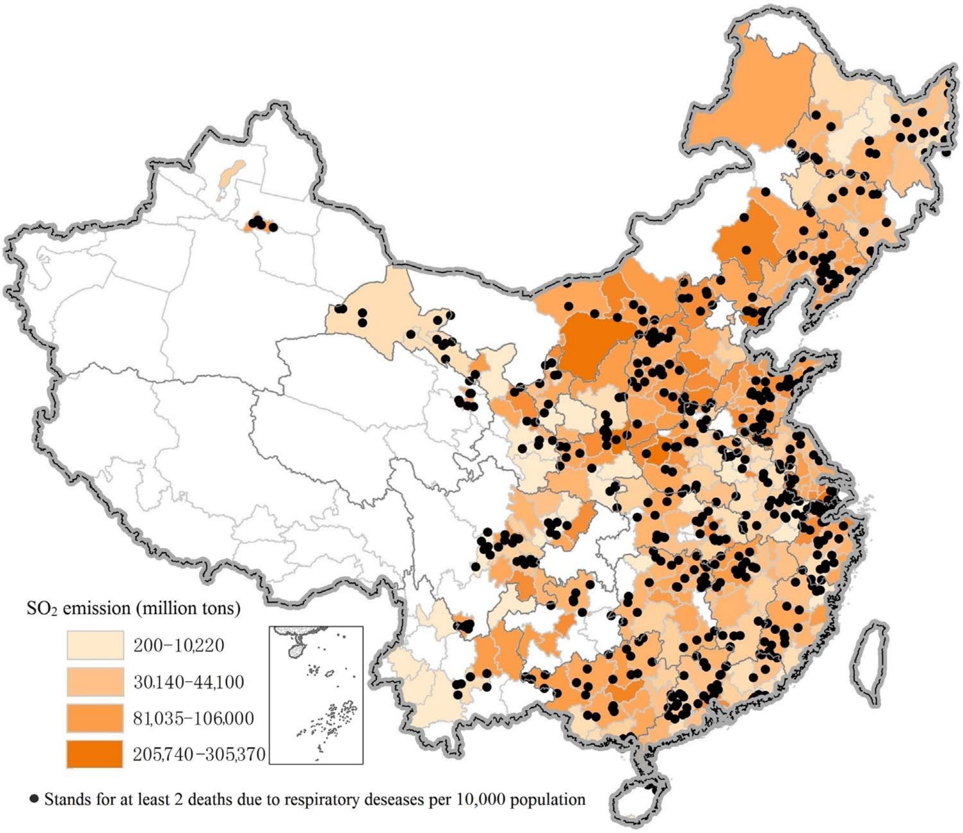


Fig. 2. Geographic distribution of power plants and SO_2 emissions.Note: The blank means there is no related SO_2 emission data.
Source: China Industrial Census Database (2004–2010) and China City Statistical Yearbook.



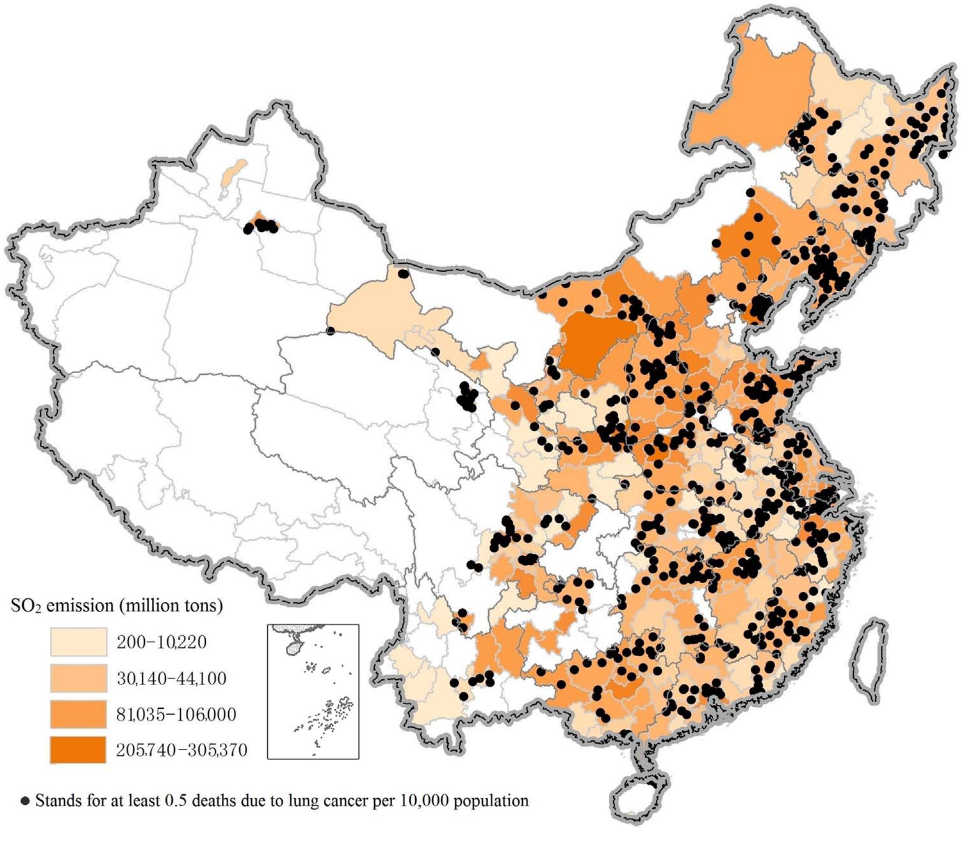


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Fig. 3. SO_2 concentrations and public health.Note: Each point represents 2 deaths due to respiratory diseases per 100,000 population (upper panel) and 0.5 death due to lung cancer per 100,000 population (lower panel). More points in one municipality means more deaths due to related diseases.
Source:China City Statistical Yearbook and DSPs (2004, 2006, 2008–10).

Table 1
Descriptive statistics.

	Data Source	Obs	Mean	S.D
Dependent variables				
Number of deaths due to respiratory diseases per 100,000 population *(DEATH-RES)*	DSPs (2004, 2006, 2008–10)	675	0.51	0.44
Number of deaths due to lung cancer per 100,000 population *(DEATH-CAN)*	DSPs (2004, 2006, 2008–10)	675	0.18	0.18
Key explanatory variables				
Amount of SO_2 emissions (10,000 tons)	China City Statistical Yearbook	675	6.99	5.81
Number of employees of power plants	China Industrial Census Database	675	4017.42	8169.03
Total assets of power plants (10,000 Yuan)	China Industrial Census Database	675	702.79	932.18
Instrumental variable				
Coal reserves	*Chinese Natural Resources Series (Volume Mineral Products)*	675	1.56	1.62
Other control variables				
Health expenditure per capita (Yuan)	*Financial Statistics of Cities and Counties*	675	304.08	219.49
Population density per km^2	*China County Statistical Yearbook*	675	379.59	260.61
GDP per capita (Yuan)	*China County Statistical Yearbook*	675	24,612.44	17,667.84

Note: The data are reported as absolute values, whereas the logrithm form is used in the regressions. The sample is a panel dataset covering the national prefecture-level cities in 2004, 2006, 2008, 2009, and 2010. *DEATH-RES* and *DEATH-CAN* are prefecture-level data for 2004 and 2006 and provincial-level data for 2008, 2009, and 2010. *DEATH-RES* is the sum of all the deaths due to diseases of the respiratory system, which include influenza, pneumonia, other acute lower respiratory infections, chronic lower respiratory diseases, and other diseases of the respiratory system. The health expenditure per capita and GDP per capita are deflated by the 2000 CPI index.

such as public health infrastructure (Crémieux et al., 2005), population density (Cassel, 1971, Ch. 12; Levy & Herzog, 1974), and level of economic development (Marmot, 2005; Smith, 1998; Wilkinson & Marmot, 2003). Thus, we separately use health expenditure per capita, population density per square kilometer, and GDP per capita as proxies. A statistical summary of all of the variables is presented in Table 1.

5. Empirical results

In this section, we empirically examine the effects of SO_2 emissions on public health. Section 5.1 provides event study evidence. The OLS estimation results are reported in Section 5.2. Section 5.3 shows the estimation results of the 3SLS, which provide a convincing solution to the potential endogeneity problem. Section 5.4 estimates the lives lost and related treatment costs due to SO_2 emissions national wide.

5.1. Event study

We find that 63% of the cities in our sample built new power plants between 2004 and 2006. Thus, a differences-in-differences (DID) method can be used to compare the changes in public health between the pre- and post-policy periods and between the cities that built new power plants and those that did not. Fig. 4 mainly plots the *DEATH-RES* variation with 95% confidence intervals between the different types of cities (treatment group and control group) over time. No substantial increase or decrease is evident during the period before the first new plant was built, suggesting that the two groups are comparable. However, this pattern changes after the plants were built: the number of deaths due to respiratory diseases in the cities with newly-built plants is significantly higher than that of the control group. Given that the survival duration for respiratory diseases after diagnosis is always shorter than the lung cancer,[8] it is reasonable to attribute this variation to the air pollution from the power plants. Nonetheless, the event study only provides prima facie evidence and we cannot rule out the possibility of omitted variable bias. Hence, we provide further evidence of the correlation in the next section.

5.2. OLS

The following estimation equation is used to test the relationship between the SO_2 emissions from power plants and public health:

$$health_{it} = \alpha_1 SO_{2it} + X'\beta_1 + prefecture_i + year_t + PT_{pt} + \varepsilon_{1it} \tag{1}$$

[8] Wu et al. (2009) showed that the average survival time of lung cancer was 13.65 months, based on data from Beijing.

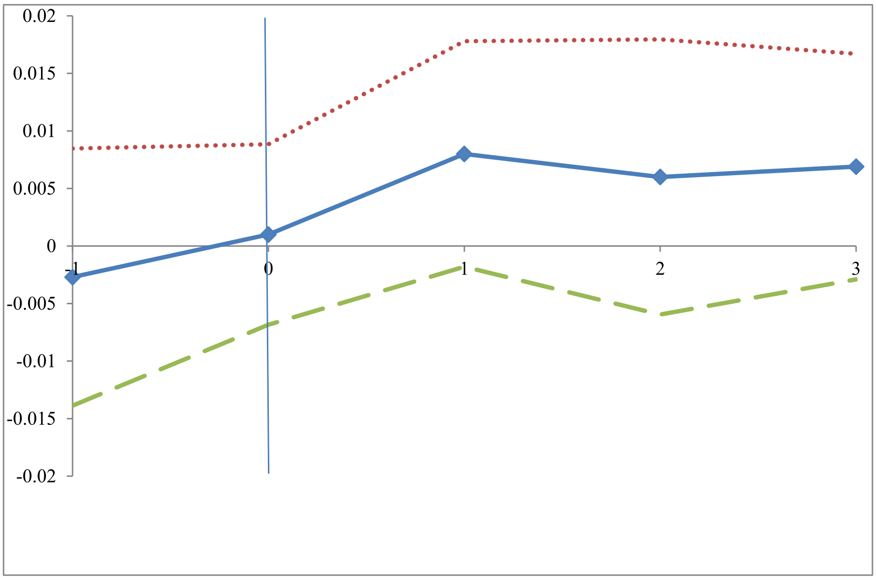

Fig. 4. The dynamic effect of SO_2 emissions on public health before and after the building of new power plants.
Note: The horizontal axis measures the number of years since the first new power plant was built. The plots connected by the solid line indicate changes in *DEATH-RES* relative to cities not built the new power plants conditional on GDP per capita (ln), health expenditure per capita (ln), population density (ln), year fixed effects, and prefecture fixed effects. The dotted lines indicate the 95% confidence intervals.

Table 2
SO_2 emissions from power plants and public health (OLS).

	DEATH-RES	*DEATH-RES*	*DEATH-CAN*	*DEATH-CAN*
	(1)	(2)	(3)	(4)
SO_2 emissions (ln)	0.046*	0.056*	0.002	0.004
	(0.025)	(0.027)	(0.012)	(0.078)
Health expenditure per capita (ln)		−0.335*		−0.223**
		(0.186)		(0.110)
Population density (ln)		0.086*		0.045***
		(0.049)		(0.011)
GDP per capita (ln)		−0.076***		−0.047***
		(0.021)		(0.015)
Constant	0.647***	2.364***	0.034	1.368***
	(0.112)	(0.147)	(0.049)	(0.231)
Provincial specific time trend	No	Yes	No	Yes
Prefecture fixed effects	Yes	Yes	Yes	Yes
Year fixed effects	Yes	Yes	Yes	Yes
Observations	675	671	675	671
R^2	0.149	0.429	0.165	0.387

Note: Huber robust standard errors are in parentheses, clustered by province. The regression disturbance term is weighted by the area of the jurisdiction (square kilometers).
*** Significant at the 1% level.
** Significant at the 5% level.
* Significant at the 10% level.

where i refers to prefecture, t is the year, and *health* is the dependent variable including *DEATH-RES* and *DEATH-CAN*. The key explanatory variable, SO_2, denotes the amount of SO_2 emissions. The vector X' includes the proxies for the other abovementioned variables. The prefecture fixed effect *prefecture* captures the time-invariant factors that may simultaneously influence both SO_2 emissions and public health such as local governance. *Year* is the time fixed effects that capture the factors that influence all of the samples over time, such as the economic cycle and macroeconomic policies. PT_{pt} is to capture the technology advances and the other factors that change over time and across regions and simultaneously influence the public health such as the motor vehicle exhaust and cigarette smoking etc. Given the possibility of omitted variables, ε_1 represents other possible omitted variables that may affect public health but are not captured.

Table 2 presents the OLS regression results. The key dependent variables are *DEATH-RES* in columns (1) and (2) and *DEATH-CAN* in columns (3) and (4). Although not all the coefficients of the terms for SO_2 emissions are significant, the results suggest that the SO_2

emissions predict damages in public health: a 1% increase in emissions results in increases of 0.056 and 0.004 deaths per 100,000 people due to respiratory diseases and lung cancer, respectively. Since the SO_2 emissions in 2006 were 14.8% higher than in 2004, SO_2 emissions resulted in 0.829 and 0.059 extra deaths per 100,000 people due to the respective diseases between 2004 and 2006.[9] Also, the SO_2 emissions from power plants could explain 57.9% of the increase in deaths due to respiratory diseases and 5.5% due to lung cancer between 2004 and 2006.[10] Considering the potential lagged effects of SO_2 emissions on public health, we also add the lagged term of SO_2 in columns (2) and (4) and find that the estimated coefficients are positive but statistically insignificant. The results for the control variables show that both health expenditure per capita and economic development are significantly positively related to respiratory morbidity, whereas the population density has the opposite effects.

The findings in Table 2 indicate that the SO_2 emissions have negative effects on public health. However, the OLS estimation may suffer endogeneity problems. First, the key explanatory variable may contain measurement bias. The existing studies have shown that air quality has been included in the performance evaluation criteria of local officials in China. The local governments have incentives to manipulate their air pollution data rather than to report the correct data (Chen, Jin, Kumar, & Shi, 2013). Therefore, the measurement error of the air quality data may be closely related to the officials' incentives, which may further affect public health and underestimate the effect of air pollution. Second, except for the measurement bias, recent evidence has suggested that public health is affected by other factors, such as weather conditions (McMichael, Woodruff, & Hales, 2006; Patz, Campbell-Lendrum, Holloway, & Foley, 2005), motor vehicle exhaust (Kagawa, 2002; Sydbom et al., 2001), and cigarette smoking (U.S. Department of Health and Human Services, 2006; Warner, 1986). If the distribution of power plants and their nameplate capacities are also related to the local socio-economic development, which may simultaneously influence these factors, the simple fixed effects model would underestimate the effects.[11] Thus, to further address this problem, we apply the 3SLS estimation in Section 5.3.

5.3. 3SLS

The empirical analysis in this section aims to solve the endogeneity problems related to gaseous pollutants. SO_2 emissions are affected by the scale of the power plants and their location. However, these factors are endogenous. Thus, in this section, we use the exogenous coal reserves as an instrumental variable for the scale of the power plants and their location. Since the specific data are unavailable, we count it as categorical variable: a coal mine is regarded as large and if its annual production capacity is over 3 million tons, as middle and if the capacity is between 0.9 and 3 million tons, and as small if the capacity is below 0.9 million tons.[12] Then we calculate the sum of total coal reserves of each coal mine in one prefecture to get the related value for certain prefecture. Each type is given a weight on a scale of 1 to 3 (small = 1, middle = 2, large = 3). Fig. 5 displays the geographic distribution of coal abundance and the number of employees of power plants. It shows the high spatial correlation between the scale of power plants and the coal reserves. For instance, the coal-abundant areas like the northeast and Shanxi province have higher density of the large-scale power plants.[13] In addition, to capture the time variation, we exploit its interaction with the dummy variable of year 2005 and after as an instrumental variable in the empirical estimation.[14]

The estimation equations are as follows:

$$health_{it} = \alpha_2 \widehat{SO}_{2it} + X'\beta_2 + prefecture_i + year_t + PT_{pt} + \varepsilon_{2it}, \tag{2}$$

$$SO_{2it} = \alpha_3 \widehat{plant_{it}} + X'\beta_3 + prefecture_i + year_t + PT_{pt} + \varepsilon_{3it}, \tag{3}$$

$$plant_{it} = \alpha_4 coal_{it}{}^{*}time_t + X'\beta_4 + prefecture_i + year_t + PT_{pt} + \varepsilon_{4it}. \tag{4}$$

We first use the interaction between coal reserves and time year dummy to predict the scale of the power plants in the first-stage regression (Eq. (4)). Since there is no data about specific nameplate capacity or yearly generation capacity, two measurements in the Industrial Census serve as proxies for the scale of the power plants in this stage: the number of employees and total assets. Then we put the estimated values into the second-stage regression (Eq. (3)), which examines the effects of power plants on SO_2 emissions. The variation in

[9] The emissions of coal-fired power plants accounts for over 62.1% of the total SO_2 emissions (Li, 2005; Liu, 2005), while the transportation is responsible for 2%–5% of the total (OECD IEA, 2016; Streets & Waldhoff, 2000). Since data on regional SO_2 emissions from vehicles were not available and the national vehicles parc over time increased smoothly, we added the provincial specific time trend to capture the effects of vehicle emissions. Thus, the estimated coefficients of 0.657 and 0.027 could be regarded as the upper bound of the effects of SO_2 emissions from power plants. We thank the anonymous referee's suggestions.

[10] The direct contribution of SO_2 emission to the public health equals the estimated elasticity of the emission (0.056 and 0.004, respectively) multiplied by the change in the SO_2 emission (153% and 203%, respectively), and divided by the change in public health outcomes (14.8%).

[11] Here we assume that positive association exists between economic growth and public health (Barro, 2013; Bloom & Canning, 2000; Bloom, Canning, & Sevilla, 2004) and the air pollution in developing countries always increases with the economic development (Grossman & Krueger, 1995; Stern & Common, 2001).

[12] The thresholds 0.9 and 3 million ton are chosen based on the regulation of data source. We also divide the coal mines into two categories based on whether its production capacity is larger than the average (1.2 million tons) or not to re-estimate the coal abundance indicators. The results are similar with our 3SLS results.

[13] Figure 5 indicates that there still exists mismatch between the coal reserves and geographic distribution of the scale of power plants. This means there might be the geographic variations of the predicted values derived from instrumental variable. Thus, we drop the observations whose correlations of coal reserves and the scales of power plants are last 5, 10, and 20 lowest and then re-estimate the columns (3) and (6) of Table 4. The results are still positive and significant, indicating that the geographic variations of predictive power of instrumental variable would not affect our argument. We thank the anonymous referee for the helpful suggestion.

[14] The value of time year dummy equals 0 if it is before 2005, equals 1 if it is 2005 and after. The interaction term can not only capture the time variation, but also isolate the possibilities that the coal reserves have effects on local economy or industrial policies and further affect the public health. We choose the year of 2005 as threshold to capture the sharp changes of SO_2 emissions due to government policy adjustment in this year. To test whether the estimated results are driven by the certain year, we also make two placebo tests: assuming that the policy adjustment happened in 2008 or 2009 (assuming the result of 2006 is same as 2005), we generate the new time year dummy and get the quite similar results.

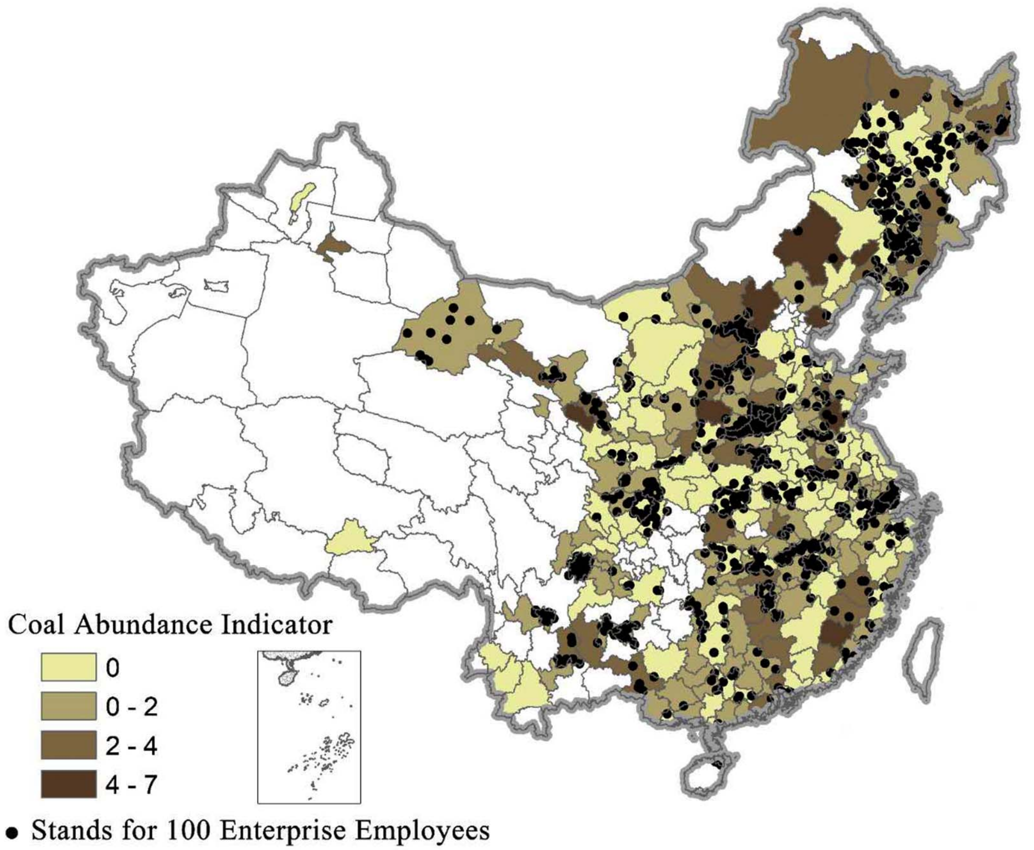

Fig. 5. Geographic Distribution of Coal Abundance and the Number of Employees of Power Plants.

the scale of the plants in this stage stems from the exogenous variable–coal reserves. Then, in Eq. (2), we use the estimated values of SO_2 emissions from Eq. (3) as the key explanatory variable and obtain the final estimation. As this process involves two instrumental variables, we use a 3SLS estimation method to systematically optimize the correlation between the different error terms of the equations. Note that here we employ a panel data model so that we could control the fixed effects—namely, we use the 3SLS with controls of prefectural and year fixed effects. Thus, the X in Eqs. (2) to (4) are also from the corresponding years, 2004, 2006, 2008, 2009 and 2010. In this sense, the 3SLS estimation method can be seen as a combination of the 2SLS and SUR (seemly unrelated regression) approaches and the estimated results are more effective. The consistent estimator for Eq. (4) reflects whether the coal reserves affect the SO_2 emissions solely by influencing the scale of the power plants. However, it is possible that the coal reserves also affect SO_2 emissions through affecting the iron and steel industries. If this is the case, the use of coal reserves as an instrumental variable would underestimate the effects. Some scholars have also noted that the coal-rich regions are generally located in developed areas with more fiscal revenue and more public health infrastructure, which may lead to higher levels of public health (Caselli & Michaels, 2013). To address this concern, we control the related economic development variables and public health expenditures in Eq. (3).

The results of the first and second stage regressions are reported in Table 3. The first-stage relationship between coal reserves and two proxies of the scale of the power plants are strongly positive: both the number of employees and total assets are significantly related to coal reserves at over 95% confidence. Each F-value is considerably greater than 10, suggesting that there is no weak instrument problem. In the second-stage regression, the estimated scale of the power plants is used as the key explanatory variable to estimate the amount of SO_2 emissions. In this case, a 1% increase in the scale of the power plants results in a 0.7%–2% rise in SO_2 emissions.

Following the same logic, the third-stage regression uses the predicted amount of SO_2 emissions derived from the number of employees as the explanatory variable since the total assets may contain other businesses.[15] The results presented in Table 4 show

[15] We believe even this situation were true, it would not affect the estimated results. Since there are no systematic data on non-power-generation businesses of power plants, we randomly select ten power plants and visit their websites to make sure whether they contain other businesses. These plants are as followed: Guizhou Qiandong Coal-fired Power Plant, Guizhou Yaxi Power Generation Limited Company, Luoyang Shouyangshan Power Plant, Huaneng Hanfeng Power Plant, Guodian Fuzhou Jiangyin Power Plant, Huaneng Huaiyin Power Plant, Suizhong Power Generation Limited Company, Huaneng Yangluo Power Plant, Mahanshan Dangtu Power Generation Limited Company, and Huanan Fengtai Power Plant. We find that only one power plant contains other non-power-generation businesses. Despite this, we still use the total assets to predict the SO_2 emissions and get the similar results. We thank the anonymous referee for the helpful suggestion.

Table 3
SO_2 emissions from power plants and public health (the first and second stage regressions).

	First-stage regression			
	Number of employees (*ln*) (1)	Number of employees (*ln*) (2)	Total assets (*ln*) (3)	Total assets (ln) (4)
IV	0.030**	0.121***	0.008**	0.026***
	(0.014)	(0.020)	(0.004)	(0.004)
Observations	675	671	675	671
R^2	0.052	0.075	0.153	0.335
	Second-Stage Regression SO_2 emission (*ln*)			
Number of employees (*ln*)	0.007**	0.024***		
	(0.003)	(0.005)		
Total assets (*ln*)			0.017***	0.017***
			(0.004)	(0.003)
Provincial specific time trend	No	Yes	No	Yes
Prefecture fixed effects	Yes	Yes	Yes	Yes
Year fixed effects	Yes	Yes	Yes	Yes
Observations	675	671	675	671
R^2	0.083	0.261	0.073	0.478

Note: Huber robust standard errors are in parentheses, clustered by province. The regression disturbance term is weighted by the area of the jurisdiction (square kilometers).The control variables in the second stage are the same as in the first stage regression except for the instrumental variables.
*** Significant at the 1% level, ** Significant at the 5% level, * Significant at the 10% level.

Table 4
SO_2 Emissions from Power Plants and Public Health (The Third-stage Regression).

	DEATH-RES (1)	*DEATH-RES* (2)	*DEATH-RES* (3)	*DEATH-CAN* (4)	*DEATH-CAN* (5)	*DEATH-CAN* (6)
SO_2 emissions (ln)	0.069**	0.113**	0.067**	0.006**	0.008***	0.004*
	(0.031)	(0.047)	(0.030)	(0.003)	(0.002)	(0.002)
Health expenditure per capita (ln)			−0.307			−0.002**
			(0.256)			(0.001)
Population density (ln)			0.123			0.413
			(0.743)			(1.039)
GDP per capita (ln)			−0.083***			−0.075***
			(0.021)			(0.013)
Constant	0.168***	0.520***	0.487***	−1.039	−3.001	2.039
	(0.012)	(0.026)	(0.026)	(1.774)	(2.489)	(3.265)
Provincial specific time trend	No	No	Yes	No	No	Yes
Prefecture fixed effects	No	Yes	Yes	No	Yes	Yes
Year fixed effects	Yes	Yes	Yes	Yes	Yes	Yes
Observations	675	675	671	675	675	671
R^2	0.056	0.072	0.459	0.031	0.047	0.326

Note: Huber robust standard errors are in parentheses, clustered by province. The regression disturbance term is weighted by the area of the jurisdiction (square kilometers). The instrumental variable in columns (1) and (4) is the coal reserves. Since this variable is time-invariant, we do not control the prefecture fixed effects. The instrumental variable in the other columns are the interactions between the coal reserves and the time year dummy of 2005.
*** Significant at the 1% level.
** Significant at the 5% level.
* Significant at the 10% level.

that the SO_2 emissions have significant negative effects on public health.[16] Specifically, a 1% increase in SO_2 emissions results in 0.067 and 0.004 extra deaths due to respiratory diseases and lung cancer per 100,000 population, respectively, suggesting that the OLS estimation method underestimates the effects of air pollution on public health. Moreover, the dramatic increase in SO_2 emissions in 2006 associated with the policy change resulted in an extra 0.992 and 0.059 deaths per 100,000 population, respectively, and can explains 69.3% and 5.5% of the mortality increases during the sample period.

[16] In Table 4, the instrumental variable in columns (1) and (4) is the coal reserves. Since this variable is time-invariant, we do not control the prefecture fixed effects. The instrumental variable in the other columns are the interaction between the coal reserves and the time year dummy of 2005.

Table 5
SO_2 emissions from power plants (absolute values) and public health (the third-stage regression).

	DEATH-RES	*DEATH-RES*	*DEATH-CAN*	*DEATH-CAN*
	(1)	(2)	(3)	(4)
SO_2 emissions	1.203***	0.735***	0.091***	0.052***
	(0. 311)	(0.220)	(0.025)	(0.004)
Other controls and constants	No	Yes	No	Yes
Prefecture fixed effects	Yes	Yes	Yes	Yes
Year fixed effects	Yes	Yes	Yes	Yes
Constant	Yes	Yes	Yes	Yes
Observations	675	671	675	671
R^2	0.063	0.175	0.117	0.235

Note: Huber robust standard errors are in parentheses, clustered by province. The regression disturbance.
term is weighted by the area of the jurisdiction (square kilometers).
*** Significant at the 1% level, ** Significant at the 5% level, * Significant at the 10% level.

Table 6
SO_2 emissions and public health (estimation of lives lost and the related treatment costs).

Year	SO_2 emissions	The number of respiratory deaths	The number of lung cancer deaths	Cost of respiratory diseases	Cost of lung cancer	Total costs
2000	19.951	185,856	13,149	6344.9	111.3	6456.2
2001	19.478	182,715	12,927	6281.3	110.2	6391.5
2002	19.266	181,896	12,869	6203.2	108.8	6312.0
2003	21.585	205,018	14,505	7075.6	124.1	7199.7
2004	22.549	215,436	15,242	7725.1	135.5	7860.6
2005	25.494	245,008	17,334	8943.6	156.9	9100.5
2006	25.888	250,115	17,695	9267.0	162.6	9429.6
2007	24.681	239,689	16,958	9307.0	163.3	9470.3
2008	23.212	226,571	16,030	9316.7	163.5	9480.1
2009	22.144	217,201	15,367	8868.9	155.6	9024.5
2010	21.851	215,357	15,236	9083.7	159.4	9243.1

Note: The unit of all costs is 1 million Yuan which has been deflated to 2010 RMB using CPI index. The unit of SO_2 emission is million tons. The number of respiratory deaths and lung cancer deaths are calculated based on the results of Table 5. The total treatment cost of respiratory diseases = (The number of respiratory deaths / The case fatality rate of respiratory diseases) *Average treatment cost of respiratory diseases. The average treatment cost of respiratory diseases in 2009 is 2109 Yuan which is the means of the treatment costs of influenza (Xia et al., 2012), pneumonia (Zhou, Wu, & Wu, 2010), and lower respiratory diseases (Hou, Zhang, & Li, 2010). The case fatality rate of respiratory diseases was about 5% (Guan et al., 2010). The total treatment cost of lung cancer = The number of deaths due to lung cancer *Average treatment cost of lung cancer. Here we ignore the case fatality rate of lung cancer due to its high mortality rate. The average treatment cost of lung cancer in 2001 is 10,460 Yuan based on an analysis of 379 patients with lung cancer (Tian, Yu, & Han, 2003).

In terms of the other control variables, the health expenditure per capita and GDP per capita have positive effects on public health. Specifically, a 1% increase in per capita health expenditure results in 0.307 and 0.002 fewer deaths per 100,000 population due to respiratory diseases and lung cancer, whereas the equivalent values for GDP per capita are 0.083 and 0.075. Note that these results are consistent with the existing literature (Nixon & Ulmann, 2006) highlighting that increased public health expenditure as the economy develops can improve public health.

Thus far, our findings indicate that SO_2 emissions could trigger a serious public health crisis. However, the extent of the problem is not obvious from the percentages. In Table 5, we change the logarithm of SO_2 emissions into absolute values and re-estimate Eq. (4) to obtain the increases in SO_2 emissions for certain diseases. Table 5 reports the new results, which indicate that every 1 million tons increase in SO_2 emissions contributes to 0.735 and 0.052 more deaths per 100,000 population due to respiratory diseases and lung cancer, respectively.[17]

5.4. Treatment costs estimation

Table 6 shows the lives lost and related treatment costs due to the SO_2 emissions reported in Table 5. Overall, SO_2 emissions lead to about 230,000 deaths every year and the average annual resulting treatment costs reach about RMB 8.179 billion between 2000 and 2010. While the total number of deaths due to respiratory diseases is 1,097,633 every year and the corresponding figure for lung

[17] Based on the estimated coefficients and the SO_2 emissions after 2010, we could calculate the deaths due to the respiratory diseases and lung cancer over time. The predicted number of deaths due to the two diseases after 2010 is 16.08 per 100,000 population which is quite close to the corresponding figure of 17.61 before 2010. The resulting deaths and treatment costs also reach about 218,000 and RMB 9.669 billion, respectively.

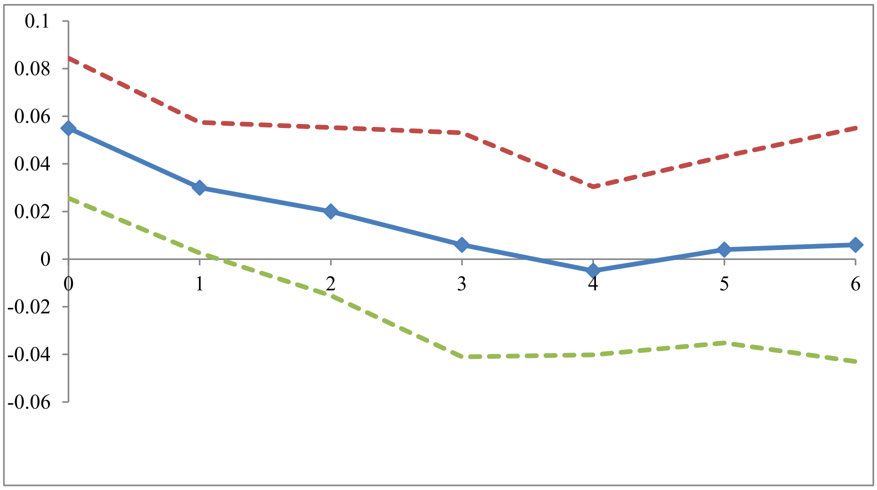

Fig. 6. The SO_2 emissions and public health by year relative to the time of first new power plant built
Note: The horizontal axis measures the number of years since the first new power plant was built. The plots connected by the solid line indicate changes in *DEATH-RES* of cities relative to the time of first new power plant built conditional on GDP per capita (ln), health expenditure per capita (ln), population density (ln), year fixed effects, and prefecture fixed effects. The dotted lines indicate the 95% confidence intervals.

cancer is 566,184. Thus, about 19.58% of the respiratory deaths and 2.7% of the lung cancer deaths are due to the SO_2 emissions every year. Note that although lung cancer patients generally die within 13.64 months (Wu et al., 2009), our estimation does not consider this lagged effect because it has little effect on the costs due to the high mortality rate of lung cancer.

To further explore the distribution of the effects across time, we add an interaction term between the amount of SO_2 emissions and the number of years since the first new power plant was built in Eq. (24). Fig. 6 plots the estimated coefficients and the 95% confidence intervals. These are interpreted as the changes in public health relative to the period since the first new power plants were built. The control group is the effects of SO_2 emissions on public health before the first new plant was built. The point estimates show that the SO_2 emissions result in 0.05–0.06 extra deaths per 100,000 population due to respiratory diseases in the first year compared with the period before the first plant was built. This effect is relatively more significant during the first three years after the plant was built. Such pattern is consistent with the existing literature about the saturation effect, in which underlying biochemical processes became saturated when exposed to a high level of a toxic component and the dose-response effects would be flattened out at higher exposure (Pope III et al., 2009; Yin et al., 2017).

We then roughly calculate the magnitude. Our estimation shows that an increase of 10 $\mu g/m^3$ SO_2 is associated with 0.79% and 0.16% increase of the respiratory mortality and lung cancer mortality, respectively.[18] The appendix has listed the related research. This magnitude is a little bit lower than the health effects of SO_2 on respiratory mortality of 1.25% estimated by Chen et al. (2012) and of 1.36% reported by Yu et al. (2012). But it is larger than the health effects of SO_2 on total mortality of 0.51% in Asian (HEI International Oversight Committee, 2004) and of 0.6% in Europe (Katsouyanni & Pershagen, 1997). This may be because pollutants differ in their toxicity and the differences across regions regarding health care utilization and many other aspects may influence the relationship between air pollution and mortality.

6. Conclusion

Based on a nationwide panel dataset covering 161 prefectural-level municipalities in China from 2004 to 2010, in this study we use the 3SLS estimation method to address the endogeneity problem caused by SO_2 emissions. We find that SO_2 emissions have significantly negative effects on public health. Generally, a 1% increase in SO_2 emissions results in a 0.085 increase in the number of deaths due to respiratory diseases and a 0.006 increase in deaths from lung cancer per 100,000 population. Meanwhile, 1 million additional tons of SO_2 emissions leads to 0.735 and 0.052 extra deaths per 100,000 population, respectively. This effect is relatively more significant during the first three years after a plant is built.

The findings of this study have implications for both academics and policy makers. Few literature employs the quasi-experiment to explore the relationship between SO_2 emissions and public health and the association between the two is still inconsistent. This study would enhance our understanding of this issue. In terms of the policy implication, the exploration of the health effects of pollution and its economic costs would benefit the government to make optimal industrial and public health policies, especially on environmental compensation and health insurance.

[18] We estimated the relationship between SO_2 concentration and emissions by using the data from 2004 to 2010.

Appendix A

Table A1
Results of studies analyzing the association between SO_2 and public health.

Authors (year of publication)	Period of study	Site	Age group	Outcome	Description
China					
Chen et al. (2012)	Periods of varying duration between the years 1996 and 2010	17 Chinese cities	All ages	Total mortality	An increase of 10 μg/m^3 of two-day moving averaged SO_2 was associated with 0.75% increase of total mortality.
				Cardiovascular mortality	An increase of 10 μg/m^3 of two-day moving averaged SO_2 was associated with 0.83% increase of cardiovascular mortality.
				Respiratory mortality	An increase of 10 μg/m^3 of two-day moving averaged SO_2 was associated with 1.25% increase of respiratory mortality.
Yu et al. (2012)	2006–2009	Guangzhou, China	All ages	Cardiovascular mortality	The increments of 10 μg/m^3 in SO_2 were associated with excessive risks of 2.28% for cardiovascular deaths.
				Respiratory mortality	The excessive risks for SO_2 were 1.36% with 10 μg/m^3 increase.
Kan and Chen (2003)	2000/06–2001/11	Shanghai, China	All ages	Total mortality	An increase of 10 mg/m^3 in SO_2 corresponded to a respective increase in relative risk of mortality from all causes of 1.014.
Venners et al. (2003)	1995	Chongqing, China	All ages	Total mortality	The relative risk of mortality of the second lag day associated with a 100 μg/m^3 increase in mean daily SO_2 was 1.04.
				Cardiovascular mortality	The relative risk of cardiovascular mortality on the second day after a 100 μg/m^3 increase in mean daily SO_2 was 1.10.
				Respiratory mortality	The relative risk of respiratory mortality on the second day after a 100 μg/m^3 increase in mean daily SO_2 was 1.11.
Wong, Ma, Hedley, and Lam (2001)	1995–1997	Hong Kong, China	All ages	Cardiovascular mortality	No significant positive exposure-response relationships during the warm season were observed.
				Respiratory mortality	No significant positive exposure-response relationships were observed.
Other countries					
Sunyer, Ballester, et al. (2003)	Periods of varying duration between the years 1988 and 1997	Birmingham, London, Milan, Paris, Rome, Stockholm, and Netherlands	All ages	Cardiovascular diseases admissions	An increase in 10 μg/m^3 of daily average of SO_2 was associated with an increase of 0.7% of all cardiovascular admissions of the same and the next day.
Katsouyanni and Pershagen (1997)	1991–1994	12 European cities in the APHEA project	All ages	Total mortality	In various European cities, an increase of 50 μg/m^3 in SO_2 was associated with about 0.8%–3% increase in daily mortality.
Hajat et al. (2007)	1990–2000	10 major cities of England	Infant	Infant mortality	A 10 mg/m^3 increase of SO_2 was associated with a relative risk of 1.02 in all infant deaths.

Zmirou et al. (1996)	1985–1990	Lyon, France	All ages	Respiratory mortality	The relative risk of respiratory deaths associated with a 50 mg/m^3 increment of mean daily SO_2 over the whole period was 1.22.
				Cardiovascular mortality	The relative risk of cardiovascular deaths associated with a 50 mg/m^3 increment of mean daily SO_2 over the whole period was 1.54.
Luechinger (2014)	1985–2003	Germany	Infant	Infant mortality	0.032 infant lives are lost per 1000 live births for every 1 $\mu g/m^3$ increase in SO_2 concentration.
Nyberg et al. (2000)	1985–1990	Stockholm, Sweden	40–75 years	Lung cancer	Little association was observed between SO_2 and lung cancer.
Sunyer, Atkinson, et al. (2003)	Periods of varying duration between the years 1988 and 1997	Birmingham, London, Milan, Paris, Rome, Stockholm, and Netherlands	All ages	Respiratory diseases admissions	The SO_2 association with respiratory admissions disappeared after adjustment for PM10.
			0–14 years	Asthma admissions	For an increase of 10 $\mu g/m^3$ of SO_2, the daily number of admissions for asthma in children increased 1.3%.

Note: The reviewed literature is listed based on the estimated effects of SO_2 on public health. Results for pollutants other than SO_2 is not shown here.

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