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Estimation of the PM$_{2.5}$ health effects in China during 2000–2011

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Abstract Exposure to fine particulate matter (PM$_{2.5}$) has been associated with mortality, but the extent of the adverse impacts differs across various regions. A quantitative estimation of health effects attributed to PM$_{2.5}$ in China is urgently required, particularly because it has the largest population and high air pollution levels. Based on the remote sensing-derived PM$_{2.5}$ and grid population data, we estimated the acute health effects of PM$_{2.5}$ in China using an exposure-response function. The results suggest the following: (1) The proportion of the population exposed to high PM$_{2.5}$ concentrations (>35 μg/m^3) increased consistently from 2000 to 2011, and the population exposed to concentrations above the threshold defined by World Health Organization (WHO) (>10 μg/m^3) rose from 1,191,191,943 to 1,290,562,965. (2) The number of deaths associated with PM$_{2.5}$ exposure increased steadily from 107,608 in 2000 to 173,560 in 2010, with larger numbers in the eastern region. (3) PM$_{2.5}$ health effects decreased in three pollution control scenarios estimated for 2017, i.e., the Air Pollution Prevention and Control Action Plan (APPCAP) scenario, the APPCAP under WHO IT-1 scenario (35 μg/m^3), and the APPCAP under WHO IT-3 scenario (15 μg/m^3), which indicates that pollution control can effectively reduce PM$_{2.5}$ effects on mortality.

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Introduction

China has experienced rapid growth in urbanization and industrialization since 1978. The economic and social development in the country has been accompanied by environmental pollution, especially air pollution that has adverse impacts on human health (Li et al. 2016; Wei et al. 2016; Zheng et al. 2015). In 2013, the annual PM$_{2.5}$ concentration in Beijing was 96.5 μg/m^3, about two to three times higher than the World Health Organization (WHO) interim target-1 level (PM$_{2.5}$ <35 μg/m^3) (Wu et al. 2015). According to the China Environmental Status Bulletin, in 2014, the annual mean PM$_{2.5}$ concentrations in 90.1% of all cities (145 of 161 cities)

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exceeded the secondary standard of the Ambient Air Quality Standard (AAQS) ($PM_{2.5}$ <35 μg/m^3) (GB3095-2012).

Atmospheric pollutants are a complex mixture including total suspended particles (TSP), particulate matter (PM), sulfur dioxide (SO_2), nitrogen dioxide (NO_2), etc. PM contributes to decreased life expectancy as confirmed by many epidemiological studies on the exposure to fine ($PM_{2.5}$, less than 2.5 μm in diameter) and coarse particulate matter ($PM_{2.5–10}$, a diameter between 2.5 and 10 μm) (Fantke et al. 2015; Franklin et al. 2007; Fuentes et al. 2006). A majority of previous studies used PM_{10} as the exposure indicator for PM to estimate the health effects of air pollution (Keuken et al. 2011; Kunzli et al. 2000; Medina et al. 2004; Pope et al. 1991; Pope and Dockery 1992). As $PM_{2.5}$ pollution is aggravated, epidemiologists investigated different methods of evaluating the health effects of $PM_{2.5}$ (Ballester et al. 2006; Fann et al. 2012; Kesanurm et al. 2014; Kheirbek et al. 2013; Orru et al. 2009), and the Global Burden of Disease used $PM_{2.5}$ as indicator for PM-related health impacts (Lim et al. 2012).

Exposure to $PM_{2.5}$ adversely affects the cardiopulmonary, respiratory, cardiovascular, nervous, and immune system (Brook et al. 2010; Guo et al. 2009; Guo and Wei 2013; Hoek et al. 2013; Mehta et al. 2013; Schwartz 2000; Sloss and Smith 2000) and significantly increases premature deaths (Boldo et al. 2006; Huang et al. 2012; Ma et al. 2011; Pope et al. 2002; Shang et al. 2013; Yang et al. 2012; Zanobetti et al. 2009). However, humans spent most of their time indoors (around 85–90% on a global average) and that the exposure to $PM_{2.5}$ happens mostly indoors, and it is mainly driven by outdoor sources of $PM_{2.5}$ (Hodas et al. 2016). There have been a large number of epidemiological studies on atmospheric pollution in which mortality, number of hospital admissions, emergency treatment, lung disease symptoms, and activity limitations are defined as the health endpoints based on outdoor $PM_{2.5}$ concentration (Guo et al. 2009; Guo and Wei 2013; Schwartz 2000). Numerous epidemiologists demonstrated that short-term exposure to $PM_{2.5}$ was associated with increased risk of mortality and morbidity (Franklin et al. 2007; Fuentes et al. 2006; Pope et al. 1991). Some studies presented their results as the relative risk (RR) of mortality due to short-term $PM_{2.5}$ exposures (Cao et al. 2011; Chen 2013; Dai et al. 2004; Pelucchi et al. 2009; Zhang et al. 2011; Zhou et al. 2014). The estimated RR coefficient was used to assess the $PM_{2.5}$ health effects by using PM-attributable deaths as the endpoint. Most of the studies on $PM_{2.5}$ health effects are from North America and Europe (Ballester et al. 2006; Fann et al. 2012; Kesanurm et al. 2014; Pelucchi et al. 2009) with very few publications for China (Cao et al. 2011; Chen 2013; Dai et al. 2004; Zhang et al. 2011). This study uses a time-series analysis method to investigate the acute health effects of $PM_{2.5}$ in China. To the best of our knowledge, 11 publications have investigated the short-term exposure-response relationships between PM and acute mortality in Chinese cities, 6 of

which focused on $PM_{2.5}$ (Huang et al. 2009, 2012; Yang et al. 2012; Chen 2013; Dai et al. 2004; Chen et al. 2011;) while others were on PM_{10} (Chen et al. 2013; Kan et al. 2008; Qian et al. 2007; Wong et al. 2008; Zhang et al. 2007). Three of the six studies on the acute effects of $PM_{2.5}$ on mortality were conducted in Shanghai.

Quantitative estimation of the health effects attributable to $PM_{2.5}$ is critical for designing policy to reduce air pollution. However, thus far, research on the health effects of $PM_{2.5}$ in China has mostly been carried out at cities (Chen 2013; Dai et al. 2004; Zhang et al. 2011); a nationwide study in China is lacking. Furthermore, these studies lack time-series analysis, which is important to study changing trends in recent years. In addition, previous studies relied on statistical and site-monitoring data and did not have any intuitive visualization of the results (Chen et al. 2013; Kan et al. 2008; Qian et al. 2007). Our study however is based on the remote sensing-derived $PM_{2.5}$ and grid population data, using international quantitative risk evaluation methods to evaluate $PM_{2.5}$ health effects in China on a national scale. The study aims to fill the gaps in the evaluation of nationwide health effects and spatial visualization research. The research was conducted in Mainland China (31 provinces) excluding Hong Kong, Macao, Taiwan, and the South China Sea Islands due to the abundance of extant studies in these areas.

With significant increases in air pollution levels, the state governments have been taking steps to control the air quality. On February 29, 2012, the China Ministry of Environmental Protection and the State Administration for Quality Supervision and Inspection and Quarantine (AQSIQ) jointly issued a new edition of the AAQS (GB3095-2012). Moreover, in 2013, China proclaimed an Air Pollution Prevention and Control Action Plan, which was its largest prevention program, proposing overall goals to improve air quality for 2017. Other provinces subsequently proposed their 2017 $PM_{2.5}$ concentration goals to improve the air quality as well as human habitat. Therefore, we created three scenarios to quantitatively estimate the health effect in China, i.e., the Air Pollution Prevention and Control Action Plan (APPCAP) scenario, the APPCAP under WHO IT-1 scenario (35 μg/m^3), and the APPCAP under WHO IT-3 scenario (15 μg/m^3).

This study has three research objectives. The first objective is to reveal spatiotemporal dynamics of population exposure to $PM_{2.5}$ from 2000 to 2011 based on the Geographic Information System (GIS) technology, i.e., ArcGIS10.2. The second objective is to estimate health effects of short-term exposures to $PM_{2.5}$ in China in 2000, 2006, 2008, and 2010, characterized by PM-attributable deaths, using an exposure-response function. The third objective is to develop three different scenarios for 2017 to simulate the health effects of $PM_{2.5}$ at different target concentrations, i.e., the APPCAP scenario, the APPCAP under WHO IT-1 scenario (35 μg/m^3), and the APPCAP under WHO IT-3 scenario (15 μg/m^3).

Methods and data

Data collection and preparation

We used global PM$_{2.5}$ concentrations for 2000–2011 estimated using remote sensing data from the Atmospheric Composition Analysis Group, which was retrieved by van Donkelaar et al. (2015) to analyze the spatiotemporal dynamics of PM$_{2.5}$ concentrations in China. The remote sensing data are based on the statistical method of using a 3-year average as the annual average with a resolution of 10 km. Demographic data used to analyze the population exposure to PM$_{2.5}$ was acquired from the LandScan Dynamic Global Population Database with a 1-km resolution, provided by the US Oak Ridge National Laboratory (ORNL). In addition, we collected mortality data from the National Statistical Yearbook, and mortality in this study refers to all-cause mortality.

Exposure-response function

The exposure-response function is used to quantitatively evaluate the change in the death rate caused by a unit increase in a pollutant's concentration level. Epidemiologists have done many researches to establish the exposure-response function (Chen 2013; Dai et al. 2004; Zhou et al. 2014; WHO 1999). Our study used the exposure-response function to evaluate acute health effects of PM$_{2.5}$ in 2000, 2006, 2008, and 2010. In addition, the spatial distribution of health effects was evaluated using hot spot analysis.

The occurrence of a disease or death is a low-probability event and is thereby in accordance with the statistical Poisson distribution. The risk ratio model in the Poisson regression model forms the basis of the current epidemiological studies on air pollution (Hong et al. 2005). In this study, we selected PM$_{2.5}$ as the main indicator of air pollution and defined mortality as the endpoint. The exposure level of atmospheric pollution is ultimately decided by the degree of pollution, which is defined as the ambient air concentration of pollutants. The value of health effects (premature deaths attributable to PM$_{2.5}$) was calculated using Eq. (1).

$$E = e^{\beta(C-C_0)} \times E_0 \qquad (1)$$

where β is the exposure-reaction coefficient, C is the actual concentration of pollutants, C$_0$ is the reference level, E is the value of health effects caused by the actual concentration of pollutants, and E$_0$ is the value of health effects caused by the reference level.

The health effects attributed to atmospheric pollution were calculated using Eq. (2).

$$\Delta E = (E-E_0)Pop = E_0\left[e^{\beta(C-C_0)}-1\right]Pop \qquad (2)$$

where Pop is the population influenced by the change in air quality.

The exposure-reaction coefficient β in Eq. (2) is the RR coefficient in our study. Based on the research by Lei (2013), mass concentrations of PM$_{2.5}$ and PM$_{10}$ have a strong correlation with 60% PM$_{2.5}$ in PM$_{10}$ according to the conversion rate of aerosol concentrations. Therefore, we converted PM$_{10}$ to PM$_{2.5}$ using a ratio of 0.6 so that we could reflect the results to wider regions in China and observe the RR coefficient in detail. According to the 11 studies summarized in Table 1, the exposure-response coefficient was 0.0054 (95% CI 0.0010, 0.0096) based on the meta-analysis, which suggests a 0.54% rise in risk for each 10 μg/m^3 increase in PM$_{2.5}$ concentrations. These numbers are in agreement with the average risk estimated from the existing studies conducted in China. Given the rising number of empirical studies in China, the risk function may be re-fit when alternative formulae are used.

National air quality standards are generally set in consideration of the local social environment, technical level of the locals, and the support capacity of the economy. However, the air quality guidelines (AQGs) set by the WHO are based on the impact of a pollutant on human health only and thus they consider the health assessment reference levels. Based on the AQGs, we chose 10 μg/m^3 as the reference level, i.e., C$_0$ in Eq. (2).

In order to explore the spatial agglomeration, we used Getis-Ord Gi* local statistics to identify the hot spots of the health effects caused by PM$_{2.5}$. The formula of Getis-Ord Gi* local statistics is described in Eq. (3) below.

$$G_i^* = \frac{\sum_{j=1}^{n}w_{i,j}x_j - X\sum_{j=1}^{n}w_{i,j}}{s\sqrt{\frac{n\sum_{j=1}^{n}w_{i,j}^2 - \left(\sum_{j=1}^{n}w_{i,j}\right)^2}{n-1}}} \qquad (3)$$

where x_j is the property value of element j, $w_{i,j}$ is the space weight between element i and element j, and n is the number of all elements. In this article, the elements i and j both referred to provinces.

The working principle of the tool is to calculate the Gi* statistics of each element and learn the spatial agglomeration of high- or low-value elements according to the Z score and the P value. When we analyze a high-value element, we must simultaneously observe the adjacent values around the element. The result has a statistical significance only when the adjacent values are also high. The Gi* statistics of each element includes the Z score, which is a multiple of the standard deviation. When Z is

Table 1 Relative risk (RR) of all-cause mortality and detailed information for each study

Study city/year	Species	Average concentration	Excess risks (%) of all-cause mortality	95% CI	Adapted from
A district in Shanghai 2002	$PM_{2.5}$	68	0.85	(0.32, 1.39)	(Dai et al. 2004)
17 Chinese cities	$PM_{2.5}$	87	0.40	(0.18, 0.61)	(Chen 2013)
17 Chinese cities	PM_{10}	109	0.35	(0.13, 0.56)	(Chen et al. 2013)
Wuhan 2001–2004	PM_{10}	141.8	0.36	(0.19, 0.53)	(Qian et al. 2007)
Shanghai 2001–2004	PM_{10}	102	0.25	(0.14, 0.37)	(Kan et al. 2008)
Hong Kong 1996–2002	PM_{10}	51.6	0.53	(0.26, 0.81)	(Wong et al. 2008)
Taiyuan 2004–2005	PM_{10}	155.54	0.25	(0.04, 0.46)	(Zhang et al. 2007)
Shanghai 2004–2005	$PM_{2.5}$	56.4	0.30	(0.06, 0.54)	(Huang et al. 2009)
Guangzhou 2007–2008	$PM_{2.5}$	70.1	0.90	(0.55, 1.26)	(Yang et al. 2012)
Xi'an 2004–2008	$PM_{2.5}$	177	0.20	(0.07, 0.33)	(Huang et al. 2012)
Beijing 2007–2008	$PM_{2.5}$	82	0.53	(0.37, 0.69)	(Chen et al. 2011)
Shanghai 2004–2008	$PM_{2.5}$	55	0.47	(0.22, 0.72)	
Shenyang 2006–2008	$PM_{2.5}$	94	0.35	(0.17, 0.53)	

The RR for mortality was assessed for each 10 $\mu g/m^3$ increase in PM_{10} and $PM_{2.5}$

positive, a higher Z score is an indication of closer clustering of high values. When Z is negative, a lower Z score indicates closer clustering of low values. The Z score must also satisfy the statistical significance, which is evaluated by the P value. Z scores and P values are associated with the standard normal distribution. We set the confidence level at 95%. We defined the hot spot area of high values as HH (the high-value spot is adjacent to the high-value spot), the hot spot area of low values as LL (the low-value spot is adjacent to the low-value spot), and other areas as transitional region (TR) (they have no statistical significance).

Scenario simulations

According to AAQS (GB3095-2012), the $PM_{2.5}$ concentration limit rules are divided into two levels. The primary standard is consistent with the interim target-3 (IT-3) of the World Health Organization (WHO) air quality guidelines for $PM_{2.5}$ (annual average value of 15 $\mu g/m^3$, 24-h average of 35 $\mu g/m^3$), and the secondary standard is consistent with the interim target-1 (IT-1) (annual average value of 35 $\mu g/m^3$, 24-h average of 75 $\mu g/m^3$). In this context, we created three different scenarios to analyze the possible changes in health effects in 2017. The Air Pollution Prevention and Control Action Plan (APPCAP) scenario was created from the province documents, based on the concentration target of 2017. In the APPCAP under WHO IT-1 scenario, the target concentration for areas with concentration levels above 35 $\mu g/m^3$ was decreased to 35 $\mu g/m^3$. In the APPCAP under WHO IT-3 scenario, the target concentration for areas still above the threshold was reduced to 15 $\mu g/m^3$.

In this study, we assumed that mortality rate in 2017 was the same as 2010 and calculated the future population growth based on the average coefficient of the natural population growth rate from 2011 to 2013 using Eq. (4).

$$Pop = (1 + Gr)^n \times Pop_0 \qquad (4)$$

where Pop is the predicted population, Gr is the natural population growth rate, Pop_0 is the population base value, and n is the difference between the base year and the forecast year.

Results

Descriptive statistics

PM$_{2.5}$ concentration, population, and mortality data in 31 provinces for 2000, 2006, 2008, and 2010 are shown in Fig. 1a–c, respectively. There was an overall upward trend in PM$_{2.5}$ concentrations since 2000. While the concentrations in Xizang, Yunnan, and Neimenggu had been staying steady

in lower levels, with an average of 4.86, 15.7, and 17.24 μg/m^3, respectively, the concentrations in Shandong, Henan, and Jiangsu remained consistently higher with an average of 71.34, 77.12, and 70.01 μg/m^3, respectively. PM$_{2.5}$ pollution was the most serious in the Northern China region and the least serious in the Qinghai-Xizang region. There were many similarities in the distributions of population and PM$_{2.5}$ levels. Beijing, Shanghai, and Tianjin were the only three cities that had inflection points because they are all municipalities, not
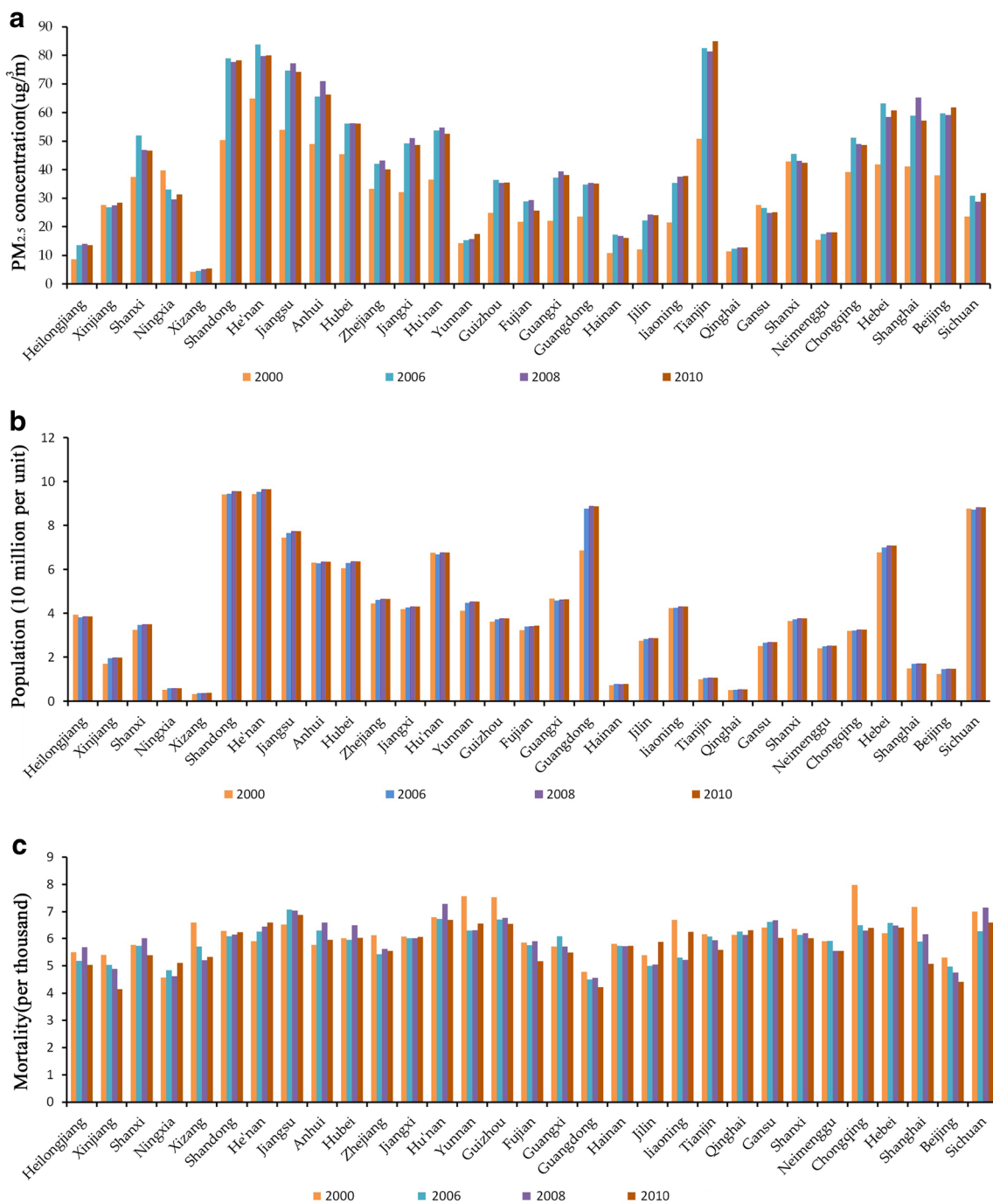

Fig. 1 **a** PM$_{2.5}$ concentrations, **b** population, and **c** mortality in 31 provinces in 2000, 2006, 2008, and 2010

provinces (Fig. 1b). Although in general smaller areas have lower population, these three regions have high population density that is correlated with high PM$_{2.5}$ concentrations (>35 μg/m^3) (Fig. 1a). There was no obvious regular distribution of mortality (Fig. 1c).

Population exposure to PM$_{2.5}$

The population exposure to PM$_{2.5}$ is based on the distribution of PM$_{2.5}$ concentrations and population. According to the Air Quality Index (AQI), AQG, and AAQS, we divided the exposed population into six groups by PM$_{2.5}$ concentrations. Levels I, II, III, IV, V, and VI correspond to less than 10, 10–15, 15–35, 35–75, 75–115, and more than 115 μg/m^3, respectively. The result for 2000–2011 is displayed in Fig. 2. The proportion of the population exposed to level I declined from 4.04 to 1.46%; level II declined from 6.94 to 2.01%; level III declined from 31.64 to 21.78%; level IV declined from 52.27 to 49.44%; level V increased from 5.11 to 24.41%; and level VI increased from 0.01 to 0.89%. The proportion of population exposed to levels above the primary AAQS of China (>15 μg/m^3) (i.e., levels III, IV, V, VI) increased from 89.02 to 96.52%. The proportion of population exposed to concentrations above the basic level fixed by the WHO (>10 μg/m^3) (i.e., II, III, IV, V) rose from 95.96 to 98.54% and covered almost the entire population. Overall, the proportion of population exposed to high PM$_{2.5}$ concentrations (>35 μg/m^3) in China increased significantly from 2000 to 2011.

Estimation of health effects

PM-attributable deaths rose from 107,608 (20,078, 189,936) in 2000 to 173,560 (32,490, 305,400) in 2010, increasing by 61.3%, and the compound annual growth rate of PM-attributable deaths was 4.9%. Figure 3 displays the distribution of the health effects of PM$_{2.5}$ in 2000, 2006, 2008, and 2010 and the change in the spatial distribution over the years.

PM-attributable deaths in each province in the map characterize the health effect for the area in that year. The high-value areas displayed annual similarities and concentrated in the central and eastern regions.

The results of the hot spot analysis (95% CI) (Fig. 4a) were consistent with the distribution of the health effects of PM$_{2.5}$, and the degree of spatial agglomeration in the hot spot analysis was more intuitive and meticulous. The HH areas were consistent in 2000, 2006, 2008, and 2010 and were concentrated in the central and eastern regions. The HH areas included Beijing, Tianjin, Hebei, Shanxi, Shandong, Shanxi, Henan, Jiangsu, Anhui, Hubei, Zhejiang, Jiangxi, and Fujian. In addition, three kinds of HH areas were determined by setting the confidence levels to 99, 95, and 90%. Thus, the research area was divided into four categories; strict control area, primary control area, secondary control area, and third control area (Fig. 4b). The first three categories corresponded to the confidence levels of 99, 95, and 90%, respectively (each inferior region excluded the regions of the superior confidence level), and the third control area referred to the rest of the research area.

Figure 5a depicts the numbers of the PM-attributable deaths in 2000, 2006, 2008, and 2010. The health effects of PM$_{2.5}$ sequentially increased from 2000 to 2010. The top three provinces of the PM-attributable deaths were Henan, Shandong, and Jiangsu, and the mortality rates due to PM$_{2.5}$ pollution in these three provinces had always been higher than the other provinces. The number of deaths in Heilongjiang, Jilin, Neimenggu, Ningxia, Gansu, Xinjiang, Qinghai, Xizang, Yunnan, Fujian, and Hainan provinces were generally low in these years. The numbers in Hebei, Shandong, Henan, Jiangsu, and Anhui were higher than the aforementioned provinces. In Liaoning, Chongqing, Hunan, Guangxi, and Guangdong, the number of deaths showed a significant increase over the years.

The variations of provincial PM-attributable excess deaths for the periods of 2000–2006, 2006–2008, and 2008–2010 are displayed in Fig. 5b. For the period of 2000 to 2006, the increments of the PM-attributable excess deaths were

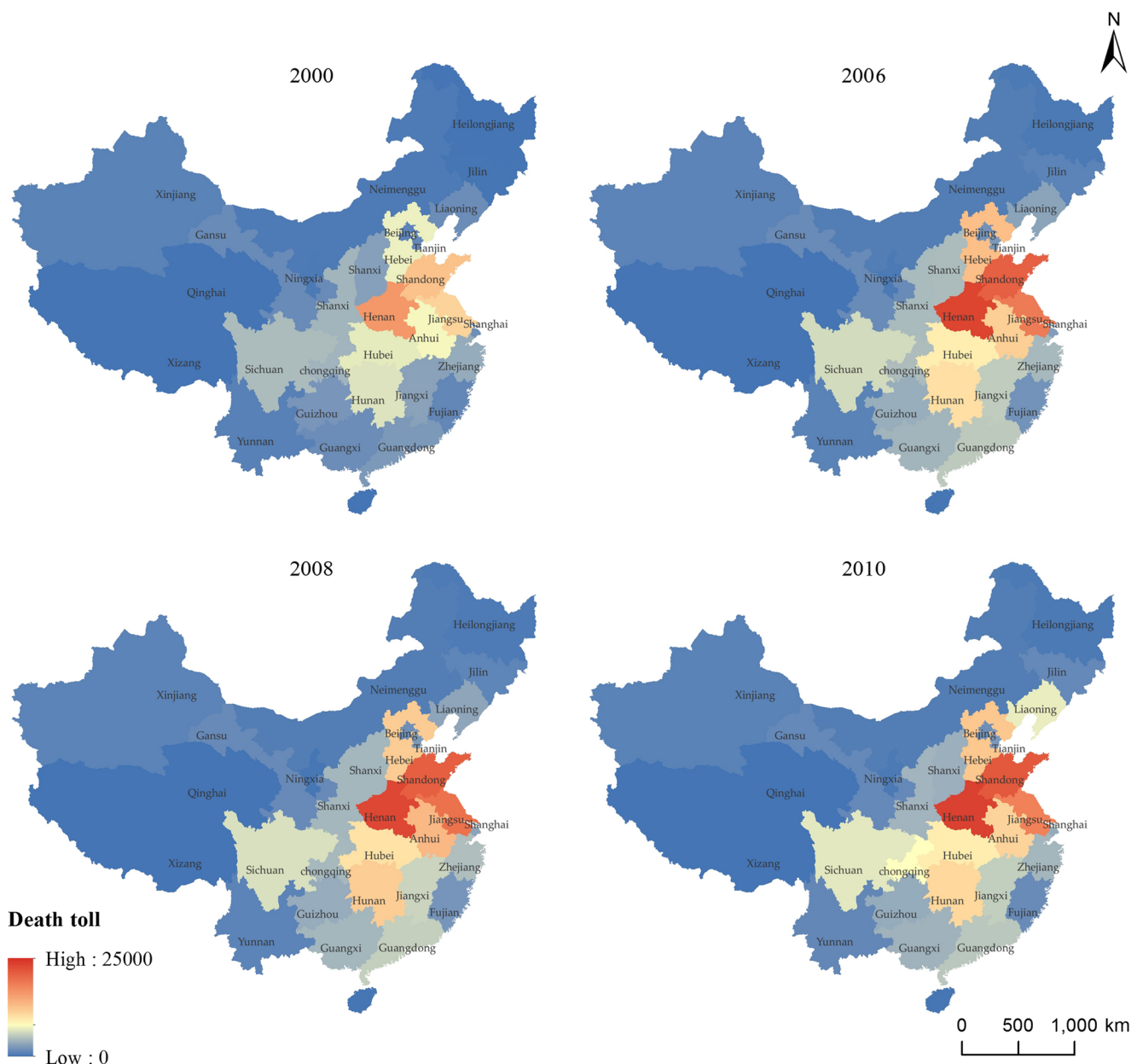**Fig. 2** Cumulative histogram of China's population exposure to PM$_{2.5}$ in the period 2000–2011
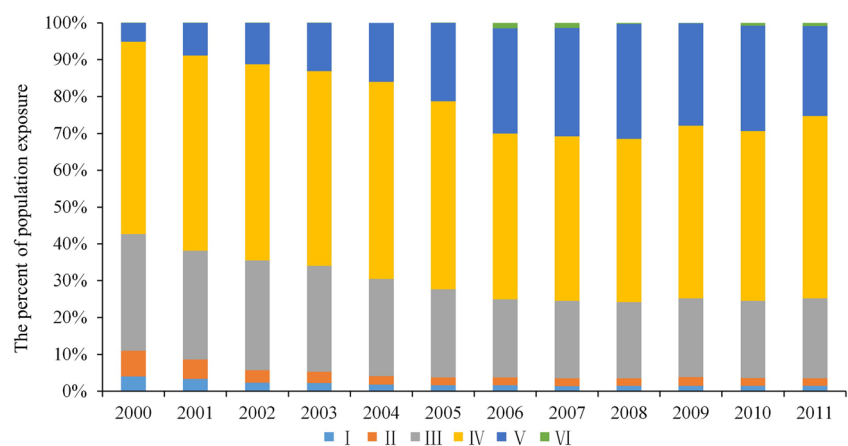
Fig. 3 Distribution of PM$_{2.5}$ health effects in 2000, 2006, 2008, and 2010

significant in Shandong, Jiangsu, Henan, and Hebei, accounting for 47.83% of the 6 years' variation. However, Ningxia was the only province that had a decline in PM-attributable excess deaths. For the period of 2006 to 2008, significant increases, which were almost 1.05 times the total periodical variation, took place in Anhui, Hunan, Hubei, and Jiangsu. Besides, PM-attributable excess deaths declined in 11 provinces, among which Hebei had the largest decrease. For the period of 2008 to 2010, increases in Liaoning, Chongqing, Henan, and Shandong were 5.19 times the total periodical variation. The overall health effects in China worsened during the research period, despite the improvements in some of the provinces.

To explore the regional differences in the study area, the mortality rates in four major economic zones in the northeastern, central, eastern, and western regions of China were compared (Fig. 6). PM-attributable excess deaths were the highest in the eastern region and lowest in the northeast over the years. However, the rate of growth in the number of deaths for 2000–2006, 2006–2008, and 2008–2010 was the highest in the northeast region, while it was lowest in the eastern region.

Scenario simulation

In the 2017 three scenarios, APPCAP, the APPCAP under WHO IT-1 (35 μg/m^3), and the APPCAP under WHO IT-3
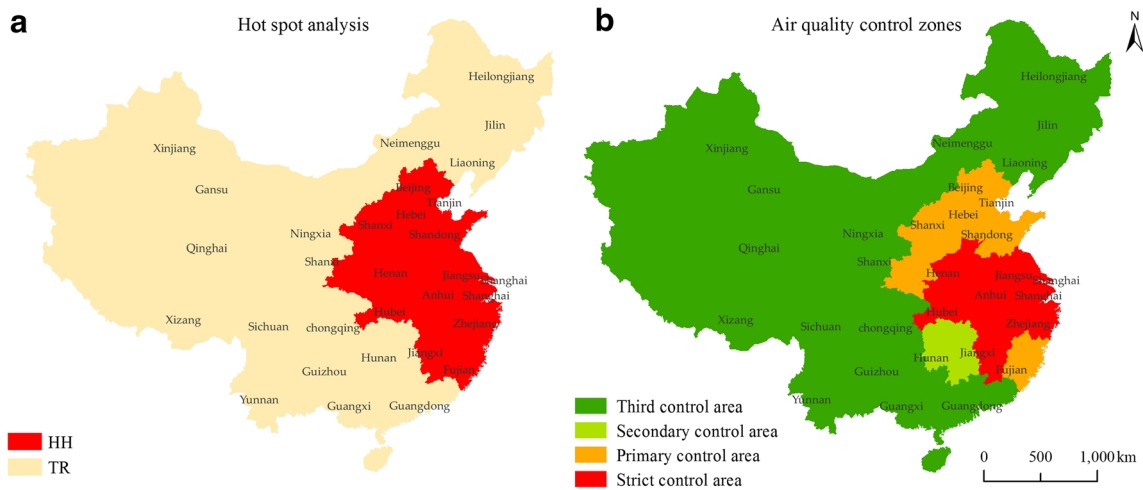
Fig. 4 **a** Hot spot analysis of PM$_{2.5}$ health effects. **b** Delineation of air quality control zones based on hot spot analysis

(15 μg/m³), the PM$_{2.5}$ health effects were found to be very different (Fig. 7). The high-value areas were still in the central and eastern regions and in Chongqing, but the high-value areas in eastern Sichuan were more scattered. In the APPCAP scenario, high-value areas were in Shandong, Henan, and Jiangsu, and the PM$_{2.5}$ concentrations in these

provinces in the base year were higher than in the other areas. In the APPCAP under WHO IT-1 scenario, the high-value areas were Hunan, Henan, Shandong, and Jiangsu. In the APPCAP under WHO IT-3 scenario, the PM$_{2.5}$ concentrations in each province exhibited very small differences; therefore, the differences in health effects in each province were also
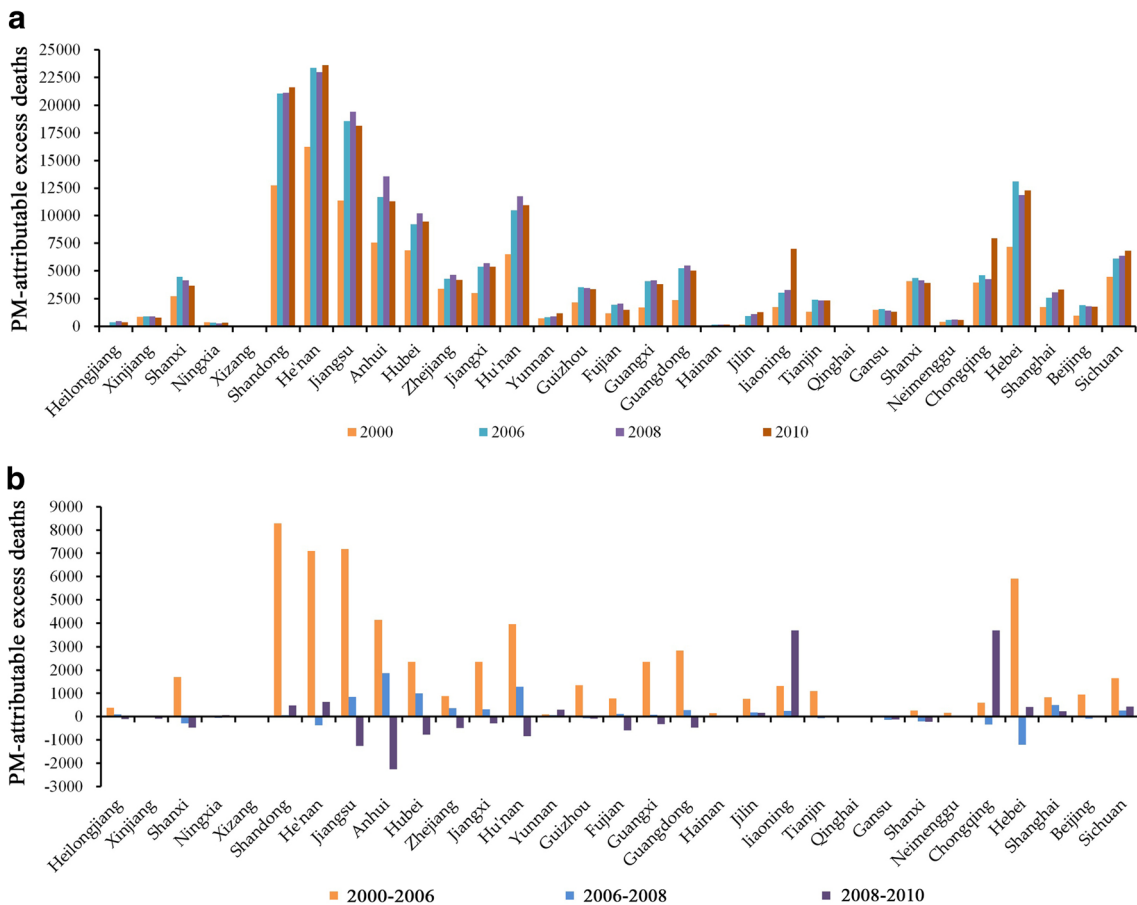

Fig. 5 **a** Numbers of the PM-attributable excess deaths in 2000, 2006, 2008, and 2010. **b** Variation of PM-attributable excess deaths for 2000 to 2006, 2006 to 2008, and 2008 to 2010
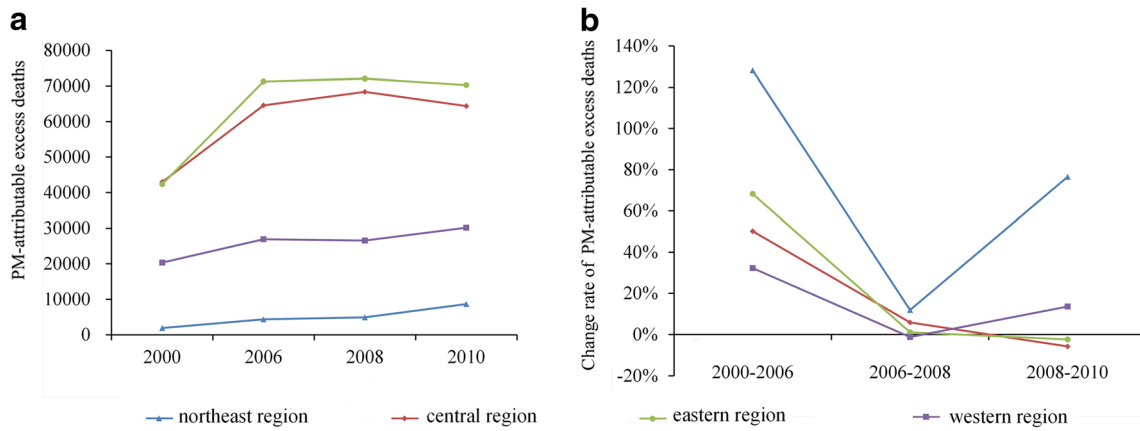
Fig. 6 **a** Numbers of the PM-attributable excess deaths in four major economic zones. **b** The rate of change in PM-attributable excess deaths in four major economic zones (northeast region: Liaoning, Jilin, Heilongjiang; central region: Shanxi, Anhui, Jiangxi, Henan, Hubei, Hunan; eastern region: Beijing, Tianjin, Hebei, Shanghai, Jiangsu, Zhejiang, Fujian, Shandong, Guangdong, Hainan; western region: Neimenggu, Guangxi, Chongqing, Sichuan, Guizhou, Yunnan, Xizang, Shanxi, Gansu, Qinghai, Ningxia, Xinjiang)

minimal. This indicates that when the concentration falls down to 15 μg/m^3, the variations in health effects among the provinces also decline, suggesting that the PM$_{2.5}$ concentrations are directly correlated with the adverse health effects.

Figure 8 depicts the numbers of the PM-attributable deaths for the same three scenarios. Compared to the health effects in 2000, 2006, 2008, and 2010 shown in Fig. 5a, the corresponding numbers for the three scenarios significantly reduced (Fig. 8). The PM-attributable deaths in the three scenarios displayed a decreasing trend, from 141,742 (26,582, 248,980) in the APPCAP scenario to 96,304 (18,084, 168,962) in the APPCAP under WHO IT-1 scenario, and finally to 22,068 (4154, 38,633) in the APPCAP under WHO IT-3 scenario, the total decrease being 84.4%. As noted in the "Estimation of health effects" section of this paper, the number of PM$_{2.5}$-related deaths in 2010 was 173,560 (32,490, 305,400). Therefore, considering only the population growth and assuming no changes in other conditions, 31,818 deaths

could have been avoided by controlling PM$_{2.5}$ concentrations in various provinces. This result suggests that controlling PM$_{2.5}$ concentrations is the most effective way to reduce the PM-related mortality rates.

Discussion

According to Eq. (1), the health effects are determined by PM$_{2.5}$ concentrations, population, and mortality data, which are all positively correlated with each other. In addition, the distributions of PM$_{2.5}$ and population show that high concentrations and population are mostly located in the central and eastern regions of China, where the fertile land and abundance of products had been attracting people and commercial activities since ancient times (Fang et al. 2012). However, the high-density population and economic activity bring a higher level of urbanization and industrialization, leading to dangerous
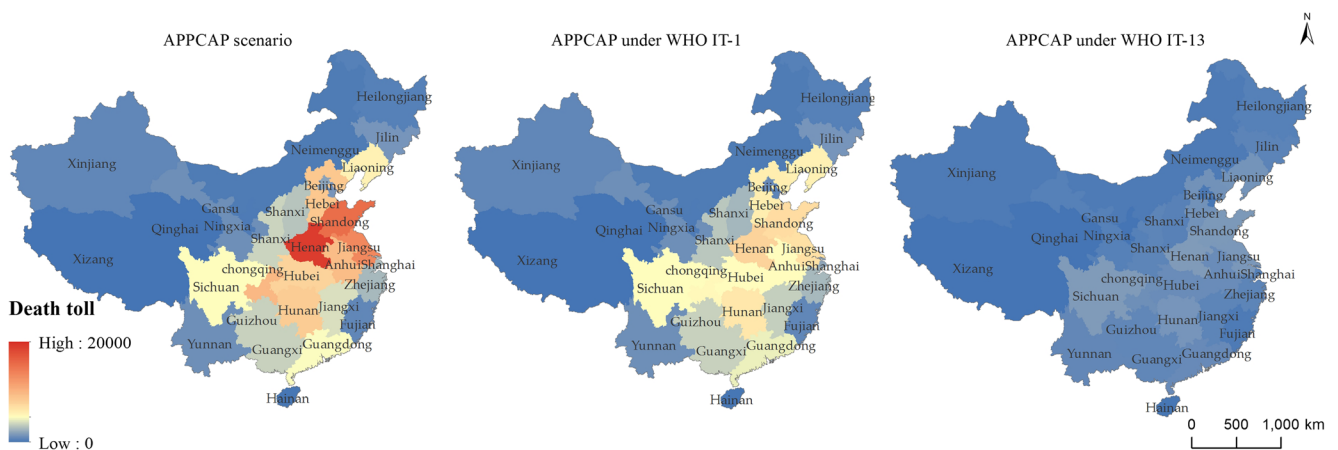

Fig. 7 Density distribution of PM$_{2.5}$ health effects in the APPCAP scenario, the APPCAP under WHO IT-1 scenario, and the APPCAP under WHO IT-3 scenario
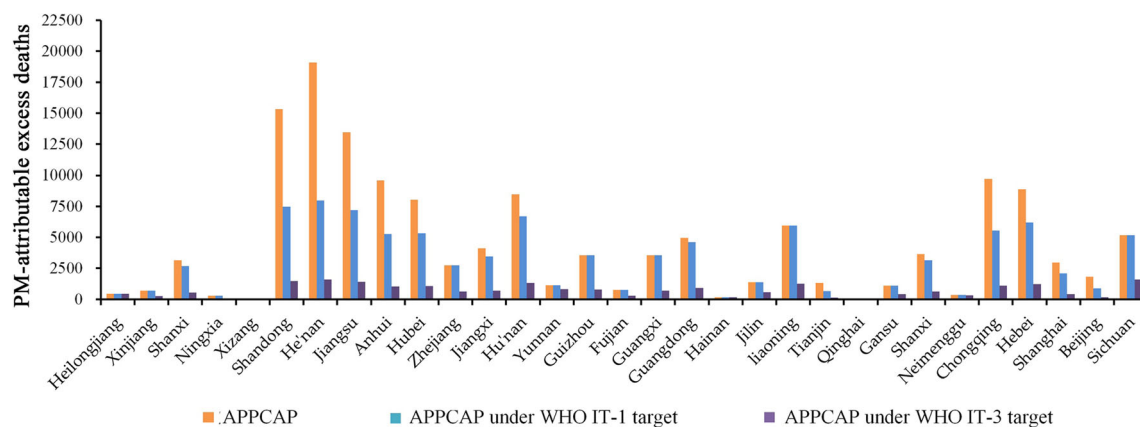
Fig. 8 Numbers of PM-attributable excess deaths in the APPCAP, APPCAP under WHO IT-1, and APPCAP under WHO IT-3 scenarios

levels of air pollution (especially PM$_{2.5}$) (Hu et al. 2014; Quan et al. 2011).

The average proportion of population exposure to high PM$_{2.5}$ concentrations (PM$_{2.5}$ >35 μg/m^3) during 2000–2011 was 70.24%, implying that over half the population was exposed to concentrations of more than 35 μg/m^3. Several studies showed that the main sources of PM$_{2.5}$ were motor vehicles, coal combustion, industrial production, and dust (Li and Luo 2014; Huang et al. 2006; Huang et al. 2015). During the 12-year period, car ownership increased from 16,090,000 to 64,670,000 and urbanization rate increased from 36.2 to 57.0%. These factors led to a higher proportion of population exposed to high PM$_{2.5}$ concentrations (Chen et al. 2014; Zhou et al. 2006; Zhou et al. 2016).

The PM-attributable death spatial distribution was consistent with the result in the research of Apte et al. (2015), while the PM-attributable deaths in our study were lower due to the difference in exposure-response values. The exposure-response values in Apte's research were predominantly based on cohort studies from European and American countries and extrapolations for high-concentration regions, and this was higher than the value we used based on the studies in China. The results of the health effects analysis showed that PM-attributable deaths increased with PM$_{2.5}$ annual average concentrations and population. The top three provinces of PM-attributable deaths in 2000, 2006, 2008, and 2010 were Henan, Shandong, and Jiangsu. Henan had the highest population and the highest PM$_{2.5}$ annual average concentration among the 31 provinces. The population of Shandong was the second among the 31 provinces, and its PM$_{2.5}$ annual average concentration was within the top three; the population of Jiangsu was the fifth among the 31 provinces, and its PM$_{2.5}$ annual average concentration was in the top four. The annual average PM$_{2.5}$ concentrations of Henan and Shandong were always elevated because of the heavy industry that contributed consistently to air pollution. The PM-attributable death in Xizang during 2000–2011 was zero, and the PM$_{2.5}$ annual average concentration was under 10 μg/m^3. The ranking of

Beijing's health effects was 22nd among all the provinces in 2000, 21st in 2006, 21st in 2008, and 20th in 2010, as its population was less than that of Henan and Shandong. Additionally, the mortality in Beijing rate was much smaller than in the other provinces; in 2010, it was 33.2% less than the rate in Henan.

Concerning the differences in health effects between the northeast and the eastern regions, the speed of heavy industrialization and economic growth in the Northeast led to the increase in pollutant emissions (Cao 2012). By contrast, the eastern areas vigorously developed their tertiary industry, limited the heavy-polluting industries, and gradually improved the environmental conditions (Qi and Zhang 2015).

According to the results of the hot spot analysis, the HH areas were the same in 2000, 2006, 2008, and 2010. The HH areas included Beijing, Tianjin, Hebei, Shanxi, Shandong, Shanxi, Henan, Jiangsu, Anhui, Hubei, Zhejiang, Jiangxi, and Fujian. The large area of high concentrations in the spatial clustering is likely related to the mobility of air (Meng et al. 2006). PM$_{2.5}$ pollution has negative impacts on the surrounding area, not solely on the area around the emission sources (Han et al. 2015). Therefore, pollution control policies should be implemented on a larger scale than at the provincial level. In this paper, we divide the research area (Mainland China) into four categories—strict control area, primary control area, secondary control area, and third control area—according to the results of the hot spot analysis on the health effects data. Based on the results, we recommend the following policy guidance: (1) The government should set rational control objectives considering the differences in health effects among provinces, rather than setting a unified standard. (2) The provinces required to adopt control rules and regulations should be provided with financial support. In addition to closing high-polluting industries or increasing their tax and loan requests, it is also necessary to provide low-polluting industries with tax breaks and other financial incentives. (3) The zone defense guideline on the national level is needed. Considering the diffusivity of PM$_{2.5}$, it is feasible to make joint efforts on the

regional scale rather than concentrate on the interests of a single province. The best results can be achieved only with the co-governance on a regional scale.

In the scenario simulations, the gaps of health effects in each province narrowed. Furthermore, there was a step-down in the degree of spatial agglomeration of the high values and the distribution was more dispersed. The PM-attributable deaths decreased from 141,742 in the APPCAP scenario to 96,304 in the APPCAP under WHO IT-1 scenario, and finally to 22,068 in the APPCAP under WHO IT-3 scenario. When $PM_{2.5}$ concentrations reduced from the APPCAP scenario to the APPCAP under WHO IT-1 scenario and from the APPCAP under WHO IT-1 scenario to the APPCAP under WHO IT-3 scenario, 45,438 (8498, 80,018) and 74,236 (13,930, 130,330) deaths could have been avoided, respectively. In order to reduce $PM_{2.5}$ concentrations, the government introduced a series of policies related to the emission sources of the pollutants. For vehicles, the government enforced fuel economy standards and raised motor vehicle emission standards. In addition, they provided road usage limitations and accelerated the elimination process of the vehicles that have not passed the emission tests. For coal combustion, they made great efforts on the optimization of the energy structure and the reduction of coal use and the increase in the use of oil and gas. Moreover, the treatment of coal waste benefited the reduction of air pollution. As to the $PM_{2.5}$ emissions from industrial production, they adjusted the industrial structure and promoted the use of clean energy to decrease $PM_{2.5}$ emissions.

There are some limitations in our study. Firstly, the aerosol data predicted by van Donkelaar et al. (2015) was used to represent $PM_{2.5}$ concentrations in our study. However, the $PM_{2.5}$ concentrations retrieved from the remote sensing data had deviations from the observed concentrations (Peng et al. 2016), which may affect the results of the quantitative evaluation of health effects. Secondly, the three scenarios are simulated based on the mortality data in 2010 at the provincial level. However, mortality is also affected by varying population structure, adaptability, living habits, socio-economic conditions, and sanitary conditions. Lastly, due to the lack of empirical studies on the $PM_{2.5}$ exposure in certain cities in China, our study did not focus on different regional characteristics such as living habits, economic environment, and environmental sanitation.

This work is of vital importance considering the serious $PM_{2.5}$ pollution problems in China. However, there were some shortcomings owing to the limitations of data sources and restrictions in the parameter selection. Future research efforts should focus on the following aspects: Firstly, in addition to mortality, the endpoint for health effects should be based on factors such as number of hospital admissions, emergency room visits, respiratory symptoms, lung disease symptoms, and activity limitations. Each endpoint may affect the research results considerably, providing a theoretical basis for related studies and policymaking. Moreover, the process of health effect evaluation should also include refining the population according to different human attributes such as gender, age, and health conditions, which will affect $PM_{2.5}$-related health effects. Under these circumstances, the results will be more accurate and targeted towards certain groups of the population.

Conclusions

We have conducted a quantitative evaluation of the health effects of $PM_{2.5}$ and simulated the health effects in different scenarios, which could greatly influence the policymaking for controlling air quality. The results indicate the following: (1) The proportion of the population exposed to high concentrations of $PM_{2.5}$ increased consistently from 2000 to 2011, and the population exposed to concentrations above the threshold defined by the WHO (10 $\mu g/m^3$) rose from 96.31 to 98.93%. (2) The number of deaths attributed to $PM_{2.5}$ exposure increased steadily from 107,608 in 2000 to 173,560 in 2010, with higher numbers in the eastern region. The top three provinces with the highest number of PM-attributable deaths in 2000, 2006, 2008, and 2010 were He'nan, Shandong, and Jiangsu, and the HH areas included Beijing, Tianjin, Hebei, Shanxi, Shandong, Shanxi, He'nan, Jiangsu, Anhui, Hubei, Zhejiang, Jiangxi, and Fujian. It is important to control $PM_{2.5}$ pollution in these provinces. (3) The mortality rates associated with $PM_{2.5}$ exposure decreased in all three scenarios for 2017, the Air Pollution Prevention and Control Action Plan (APPCAP) scenario, the APPCAP under WHO IT-1 scenario (35 $\mu g/m3$), and the APPCAP under WHO IT-3 scenario (15 $\mu g/m3$). This suggests that $PM_{2.5}$ control can effectively reduce the risk of mortality. (4) The government should set rational but differential PM control objectives taking into account the differences among provinces.

Author contributions J. Wu and W. Li designed the study, performed data analysis, and reviewed and approved the manuscript. J. Zhu and D. Xu participated in data analysis and manuscript writing. J. Liu participated in data analysis.

Compliance with ethical standards

Conflicts of interest The authors declare that they have no conflict of interest.

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